U.S. Environmental Protection Agency
Region IX

Los Angeles Area Lakes Total Maximum Daily Loads for Nitrogen, Phosphorus, Mercury, Trash, Organochlorine Pesticides and PCBs

Photo: Puddingstone Reservoir

Approved by:

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26 March 2012

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Executive Summary

The Los Angeles Regional Board identified 10 lakes in the Los Angeles region as impaired by algae, ammonia, chlordane, copper, DDT, eutrophication, lead, organic enrichment/low dissolved oxygen, mercury, odor, PCBs, pH and/or trash and placed them on California's 303(d) list of impaired waters requiring a Total Maximum Daily Load (TMDL) (LARWCQB, 1998). The United States Environmental Protection Agency (USEPA) Region IX subsequently entered into a consent decree with several environmental groups on March 22, 1999 that required development of TMDLs for these waterbody pollutant combinations by March 2012 (Heal the Bay Inc., et al. v. Browner C 98-4825 SBA). To meet the consent decree deadline, USEPA is establishing Total Maximum Daily Loads (TMDLs) in nine of these lakes in the Los Angeles region. For several lakes, USEPA concluded that ammonia, pH, copper and/or lead are currently meeting water quality standards and TMDLs are not required at this time. In other lakes, recent chlordane and dieldrin data indicate additional impairment. USEPA is establishing 33 TMDLs in all, as follows:

NITROGEN AND PHOSPHORUS TMDLS

EPA is establishing eight total nitrogen and eight total phosphorus TMDLs for Peck Road Park Lake, Lincoln Park Lake, Echo Park Lake, Lake Calabasas, El Dorado Park Lakes, Legg Lakes, Puddingstone Reservoir and Santa Fe Dam Park Lake. The Los Angeles Regional Board identified eight lakes as impaired by algae, ammonia, eutrophication, organic enrichment/low dissolved oxygen, odor and/or pH. These various impairments stem from excess nitrogen and phosphorus in the lake, causing excess algae growth, which then impairs aquatic life and recreation uses. Chlorophyll *a* is used as an indicator of algal density and a target of 20 micrograms per liter was set in these TMDLs to protect beneficial uses. The impacts of nutrient loading on each impaired lake were estimated through scientific modeling of lake-specific conditions. This model generates site-specific nutrient loadings required to attain the chlorophyll *a* target at each lake. Data currently indicate Echo Park Lake, Peck Road Park Lake, Santa Fe Dam Park and the southern lake system of El Dorado Park Lakes are meeting the chlorophyll *a* target. In these lakes, USEPA is therefore assigning wasteload and load allocations to the responsible jurisdictions based on existing loading of nitrogen and phosphorus to each lake. Lake Calabasas, Legg Lakes, Lincoln Park Lake, Puddingstone Reservoir and the northern lake system of El Dorado Park Lakes are assigned wasteload and load allocations based on model outputs. To allow flexibility in implementing the nutrient TMDLs, responsible jurisdictions receiving required reductions have the option to submit a request to the Regional Board for alternative concentration-based wasteload allocations, with a Lake Management Plan to show how the water quality standards, chlorophyll *a* target and the concentration-based wasteload allocations will be achieved by improved lake management practices. These jurisdictions can receive alternative concentration-based wasteload allocations not to exceed 1.0 and 0.1 milligrams per liter total nitrogen and total phosphorus, respectively. For lakes not currently attaining the chlorophyll *a* target, this TMDL includes required reductions in total loading of 45 percent to 71 percent for total nitrogen and 23 percent to 62 percent for total phosphorus, depending on the lake.

MERCURY TMDLS

EPA is establishing three mercury TMDLs for El Dorado Park Lakes, Puddingstone Reservoir and Lake Sherwood. Elevated fish tissue concentrations of methylmercury are impairing beneficial uses at Lake Sherwood, El Dorado Park Lakes and Puddingstone Reservoir. The concentrations of these pollutants in fish tissue exceed the State of California's Fish Contaminant Goals (FCGs) to protect human health. Mercury is a heavy metal that bioaccumulates and biomagnifies up the food chain. As fish grow, they accumulate more methylmercury in their tissue such that older and larger fish have higher concentrations of methylmercury than younger and smaller fish. The fish tissue target for these TMDLs, 0.22 parts per million methylmercury, is based on a 350 mm largemouth bass which is the most common size and the most common species caught by anglers in these lakes. These TMDLs assign wasteload and load allocations to responsible jurisdictions for total mercury as a mass per year. These TMDLs include a dissolved methylmercury target of 0.081 nanograms per liter based on a calculation of the maximum allowable concentration in the water column to attain the largemouth bass fish tissue target using nationally derived bioaccumulation factors. Required reductions in total mercury loading range from 47 percent to 72 percent, depending on the lake.

CHLORDANE, DIELDRIN, TOTAL DDTS, AND TOTAL PCBS TMDLS

EPA is establishing 11 TMDLs for chlordane, dieldrin, total DDTs and total PCBs at Peck Road Park Lake, Echo Park Lake and Puddingstone Reservoir. Elevated fish tissue concentrations of organochlorine pesticides and PCBs are impairing the beneficial uses at Echo Park Lake, Peck Road Park Lake and Puddingstone Reservoir. The concentrations of these pollutants in fish tissue exceed the State of California's FCG targets. These types of pollutants have low solubility and a high affinity for organic solids and lipids, and tend to bioaccumulate and biomagnify up the food chain from sediment to fish tissue. Water column concentrations of these pollutants are extremely low and currently attaining water quality criteria. Wasteload and load allocations are therefore assigned as a concentration of a pollutant associated with suspended sediments. USEPA set sediment targets by calculating the maximum allowable concentrations in sediment to attain the fish tissue targets and choosing the lower of this value or a target to protect benthic organisms. In all but one case, the sediment value calculated to attain the fish tissue targets is lower and wasteload and load allocations are assigned to responsible jurisdictions based on that calculated value. Additionally, if responsible jurisdictions demonstrate that fish tissue targets are being attained, alternative sediment wasteload allocations, based on the target used to protect benthic organisms, go into effect. Required reductions in pollutant concentrations in sediment range from 5.2 percent to 99 percent depending on the particular pollutant and lake.

TRASH TMDLS

EPA is establishing three trash TMDLs in Peck Road Park Lake, Lincoln Park Lake and Echo Park Lake. Trash in lakes causes water quality problems including reduced habitat for aquatic life, direct harm to wildlife from ingestion or entanglement, and health impacts to people recreating near trash potentially contaminated with human or pet wastes. Since any amount of trash causes impairment, wasteload and load allocations assigned to responsible jurisdictions are set at zero trash.

The following TMDLs are included in this document:

- Peck Road Park Lake: nitrogen, phosphorus, chlordane, DDT, dieldrin, PCBs, trash
- Lincoln Park Lake: nitrogen, phosphorus, trash
- Echo Park Lake: nitrogen, phosphorus, chlordane, dieldrin, PCBs, trash
- Lake Calabasas: nitrogen, phosphorus
- El Dorado Park Lakes: nitrogen, phosphorus, mercury
- Legg Lakes (North, Center and Legg): nitrogen, phosphorus
- Puddingstone Reservoir: nitrogen, phosphorus, chlordane, DDT, PCBs, mercury, dieldrin
- Santa Fe Dam Park: nitrogen, phosphorus
- Lake Sherwood: mercury

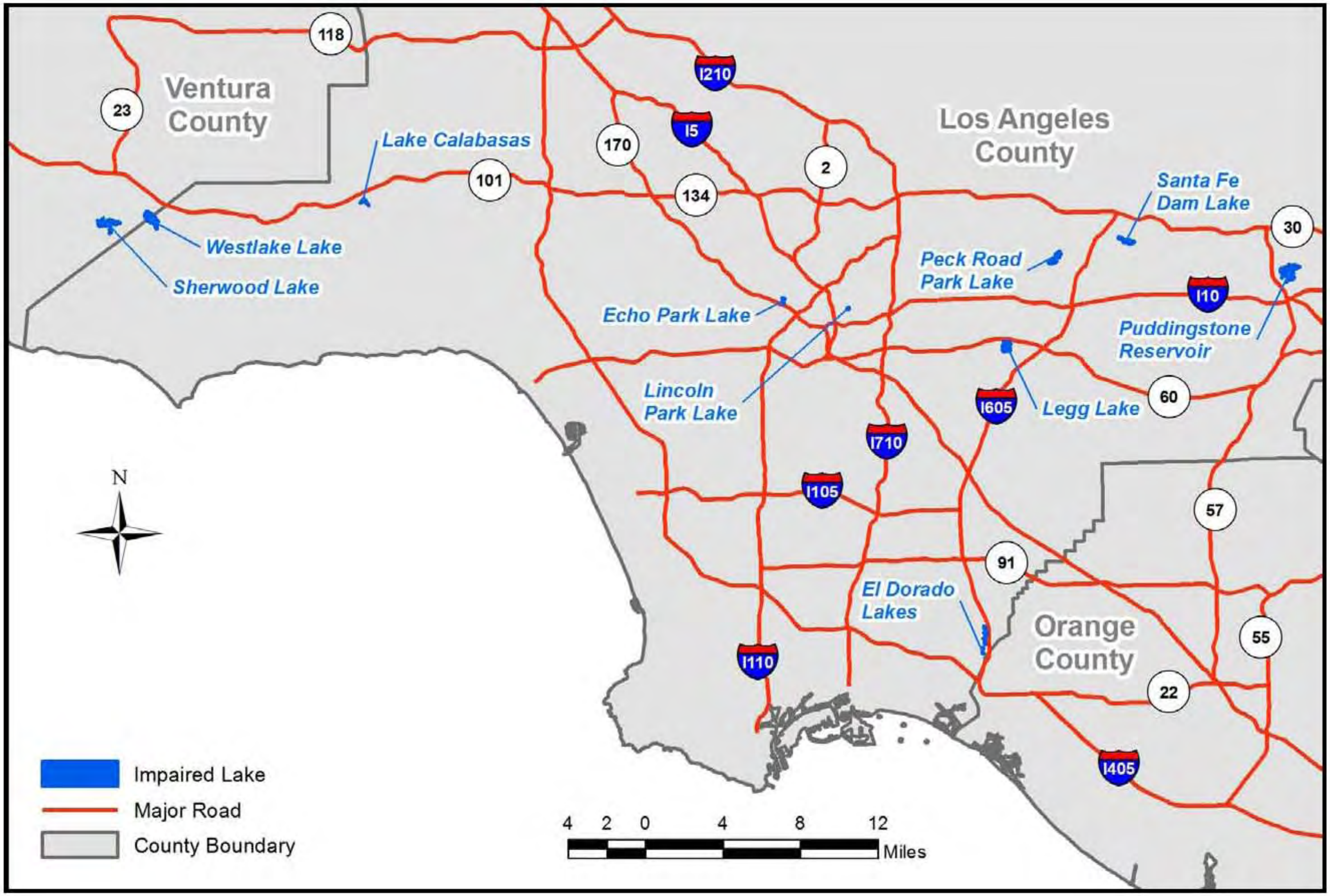

Figure ES-1. Location of Ten Lakes in the Los Angeles Region

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1 Introduction

The United States Environmental Protection Agency (USEPA) Region IX is establishing Total Maximum Daily Loads (TMDLs) in nine lakes in the Los Angeles Region. USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Tetra Tech produced the Technical Support Document to aid in the development of these TMDLs.

Numerous impaired lakes are addressed by these TMDLs. Each lake is located in the Los Angeles River Basin, San Gabriel River Basin, or Santa Monica Bay Basin (Figure 1-1). The identified pollutants are either categorized or individual; e.g., trash or mercury. Chlordane, dieldrin and DDT are organochlorine (OC) pesticides and have been grouped together with PCBs. Nutrient TMDLs are defined to address: algae, ammonia, eutrophication, low dissolved oxygen/organic enrichment, odor, and/or pH.

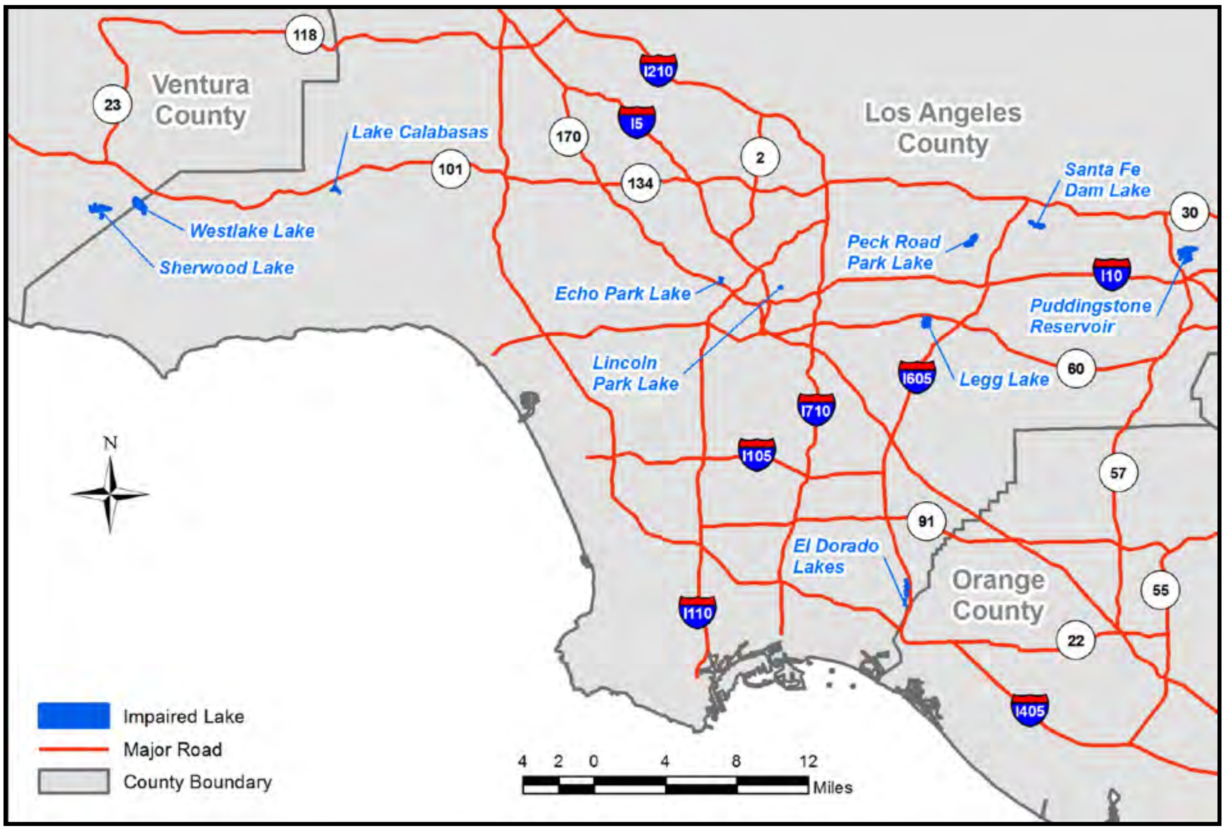

Figure 1-1. **Location of Ten Lakes in the Los Angeles Region**

The TMDLs included in this document are summarized below:

- Peck Road Park Lake: nitrogen, phosphorus, chlordane, DDT, dieldrin, PCBs, trash
- Lincoln Park Lake: nitrogen, phosphorus, trash
- Echo Park Lake: nitrogen, phosphorus, chlordane, dieldrin, PCBs, trash
- Lake Calabasas: nitrogen, phosphorus
- El Dorado Park Lakes: nitrogen, phosphorus, mercury

- Legg Lakes (North, Center and Legg): nitrogen, phosphorus
- Puddingstone Reservoir: nitrogen, phosphorus, chlordane, DDT, PCBs, mercury, dieldrin
- Santa Fe Dam Park: nitrogen, phosphorus
- Lake Sherwood: mercury

USEPA determined some lakes were not impaired for copper or lead, therefore we did not develop TMDLs for those metals. Information related to our findings of non-impairment is included within the lake specific sections as well as Appendix G (Monitoring Data). A full list of specific waterbody-pollutant combinations addressed by this document is included in Table 2-31.

This document is organized into the following sections and appendices to address the multiple lake/impairment combinations included in these TMDLs:

- Section 1 contains the introductory material, regulatory background, and description of the elements of a TMDL.
- Section 2 describes the problem statement in terms of water quality standards, beneficial uses, water quality objectives, and numeric targets. The 1998 basis of 303(d) listing and summary of impairments for each lake are also included in this section.
- Section 3 summarizes the approach that was used for the source assessment and linkage analysis for each impairment.
- Sections 4 through 13 contain the lake specific TMDL information including the environmental setting and the summaries of impairments, monitoring data, pollutant loading, and TMDL allocations.
- Section 14 contains references for this document.
- Appendix A (Nutrient TMDL Development) describes the model input and output for application of the NNE BATHTUB model in relation to the nutrient impairments.
- Appendix B (Internal Loading) describes the processes of internal loading, wind mixing, and bioturbation of the lake sediments.
- Appendix C (Mercury TMDL Development) explains the load allocation determinations for the mercury impairments.
- Appendix D (Wet Weather Loading) describes wet weather pollutant loading.
- Appendix E (Atmospheric Deposition) describes the estimation of pollutant loading from atmospheric deposition.
- Appendix F (Dry Weather Loading) describes dry weather pollutant loading.
- Appendix G (Monitoring Data) contains the monitoring data relevant to each lake and impairment.
- Appendix H (Organochlorine Compounds TMDL Development) describes the steady-state model for Organochlorine (OC) Pesticides (including chlordane, DDT, and dieldrin) and PCBs.

1.1 Regulatory Background

Section 303(d) of the Clean Water Act (CWA) requires that each state "shall identify those waters within its boundaries for which the effluent limitations are not stringent enough to implement any water quality standard applicable to such waters." The CWA also requires states to establish a priority ranking for waters on the 303(d) list of impaired waters and establish TMDLs for such waters.

The elements of a TMDL are described in 40 CFR 130.2 and 130.7 and Section 303(d) of the CWA, as well as in U.S. Environmental Protection Agency (USEPA) guidance (USEPA, 2000b). A TMDL is defined as the "sum of the individual waste load allocations (WLAs) for point sources and load allocations (LAs) for nonpoint sources and natural background" (40 CFR 130.2) such that the capacity of the waterbody to assimilate pollutant loads (the Loading Capacity) is not exceeded. A TMDL is also required to account for seasonal variations and include a margin of safety to address uncertainty in the analysis.

The USEPA has oversight authority for the 303(d) program and is required to review and either approve or disapprove the TMDLs submitted by states. In California, the State Water Resources Control Board (State Board) and the nine Regional Boards are responsible for preparing lists of impaired waterbodies under the 303(d) program and for preparing TMDLs, both subject to USEPA approval. If USEPA does not approve a TMDL submitted by a state, USEPA is required to establish a TMDL for that waterbody. The Regional Boards also hold regulatory authority for many of the instruments used to implement the TMDLs, such as National Pollutant Discharge Elimination System (NPDES) permits and state-specified Waste Discharge Requirements (WDRs).

As part of its 1998 regional water quality assessments, the Regional Board identified over 700 waterbody-pollutant combinations in the Los Angeles Region where TMDLs would be required (LARWCQB, 1998). These are referred to as "listed" or "303(d) listed" waterbodies. A 13-year schedule for development of TMDLs in the Los Angeles Region was established in a consent decree approved between USEPA and several environmental groups on March 22, 1999 (Heal the Bay Inc., et al. v. Browner C 98-4825 SBA). For the purpose of scheduling TMDL development, the decree combined the more than 700 waterbody-pollutant combinations into 92 TMDL analytical units.

This report addresses waterbody impairment combinations identified in Analytical Units 16, 17, 19, 20, 41, 42, 44, and 68 of the Consent Decree. Under the consent decree, USEPA must approve or establish these TMDLs by March 2012. The State is unlikely to complete adoption of these TMDLs in time to meet the consent decree deadline; therefore, USEPA is establishing these TMDLs.

USEPA performed a review and analysis of available monitoring data and information for pollutants and waterbodies within the analytical units in the consent decree described above. Historic data related to the 1998 list and current data related to the current 303(d) list were evaluated to determine if any water quality conditions had changed (either from impaired to non-impaired or vice versa). In certain cases, USEPA concluded that ammonia, pH, and metals (copper and lead) are currently achieving numeric targets and TMDLs are not required for these pollutants. These analyses and determinations of non-impairment are presented in the lake-specific chapters. Establishment of the TMDLs in this document thereby completes the requirement in the consent decree to address Analytical Units 16, 17, 19, 20, 41, and 42. It also partially addresses analytical units 44 and 68. In addition, these TMDLs incorporate impairments not included in the consent decree. There are several impairments for these waterbodies included on the 2008-2010 303(d) list (SWRCB, 2010), which was developed after the consent decree, as well as newly identified impairments not currently on the 303(d) list. USEPA is including TMDLs to address these additional impairments to more efficiently use agency resources and encourage expediency of restoration of water quality in these lakes.

Overall, this report includes an evaluation of available data to either confirm, establish, or refute impairment(s) for each waterbody. TMDLs have been developed to address the impairments. Table 2-31 summarizes the waterbody impairment combinations addressed by this report.

1.2 Elements of a TMDL

Guidance from USEPA (2000b) identifies seven elements of a TMDL. This report contains these seven elements in the following Sections or Appendices:

1. Problem Statement. Section 2 reviews the evidence used to include each waterbody on the 303(d) list. A description of the water quality standards, beneficial uses, water quality objectives, and numeric targets that form the basis for each listing was reviewed.

2. Numeric Targets. Section 2 also includes the numeric targets based on the numeric and narrative water quality objectives stated in the Basin Plan as well fish tissue guidelines and sediment quality guidelines. These targets are used for confirmation of impairments and calculation of TMDLs for mercury, OC Pesticides and PCBs, and trash. For the nutrient impairments, lake specific total nitrogen and total phosphorus targets are developed using the NNE BATHTUB model (described in Appendix A, Nutrient TMDL Development). Appendix C (Mercury TMDL Development) and Appendix H (Organochlorine Compounds TMDL Development) include additional details on the mercury and OC Pesticides and PCBs targets. Load reductions and pollutant allocations in these TMDLs are developed to ensure that these numeric targets for the impaired waterbodies are met.

3. Source Assessment. This step is a quantitative estimate of point sources and nonpoint sources of pollutant loading in each watershed. The source assessment considers seasonality and flow. The general approach for determining source assessments by pollutant is summarized in Section 3. Lake specific loading summaries by pollutant are included in the individual lake sections (Sections 4 through 13). More detailed information regarding modeling input and data sets used to quantify pollutant loading are described in Appendices B, C, D, F, and H.

4. Linkage Analysis. This analysis demonstrates how the sources of pollutant compounds in each waterbody are linked to the observed conditions in the impaired waterbody. The linkage analysis includes an assessment of critical conditions, which are periods when the changing pollutant sources and changing assimilative capacity of the waterbody combine to produce either extreme impairment conditions or conditions especially resistant to improvement. Section 3 describes the linkage analysis for each impairment, and more details are provided in the appendices.

5. TMDLs and Pollutant Allocations. The total loading capacity for each waterbody is determined as the amount of pollutant loading a waterbody can receive without causing impairment. A Margin of Safety (MOS) is set aside to account for inherent variability in modeling assumptions and datasets. The TMDL is set as the loading capacity minus the MOS. Each pollutant source is allocated an allowed quantity of pollutant loading that it may discharge. Allocations are designed such that the waterbody will not exceed numeric targets for any of the compounds or effects in any of its reaches. Point sources and areas draining to municipal separate stormwater systems (MS4s) are given waste load allocations, and nonpoint sources are given load allocations. TMDLs and pollutant allocations are described for each lake and impairment in Sections 4 through 13.

6. Implementation Recommendations. This element describes the plans, regulatory tools, or other mechanisms by which the waste load allocations and load allocations may be achieved. The Regional Board has responsibility to implement these TMDLs and incorporate them into permits. They may choose to develop implementation plans in a separate document(s) in the future.

7. Monitoring Recommendations. Monitoring each waterbody is recommended to ensure that the wasteload allocations and load allocations are achieved, that numeric targets are no longer exceeded, and that the secondary effects intended to be addressed by these TMDLs are being addressed.

2 Problem Statement

The lakes covered by this document are impacted by numerous impairments including nutrient-related impairments (algae, ammonia, eutrophication, low dissolved oxygen/organic enrichment, odor, pH), metals (copper and lead), mercury, trash, and OC Pesticides (chlordane, DDT, and dieldrin) and PCBs. This section describes the beneficial uses identified in the Water Quality Control Plan (Basin Plan) for each waterbody and discusses the applicable numeric targets for each beneficial use. It also includes water quality information (wherever possible) to describe the basis for each listing as provided by the Regional Board for the 1998 303(d) list. The reader will find discussion and summary of more recent monitoring data for each waterbody in the lake-specific chapters.

2.1 WATER QUALITY STANDARDS

California state water quality standards include of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives and numeric water quality criteria, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Boards in the Basin Plans. Numeric and narrative objectives are specified in each region's Basin Plan and numeric criteria are included in the California Toxics Rule (CTR), designed to be protective of the beneficial uses.

2.1.1 Beneficial Uses

The Water Quality Control Plan for the Los Angeles Region (LARWQCB, 1994) defines 11 beneficial uses for the 10 lakes addressed by this report:

AGR - Agricultural Supply. Uses of water for farming, horticulture, or ranching including, but not limited to, irrigation, stock watering, or support of vegetation for range grazing.

COLD - Cold Freshwater Habitat. Uses of water that support cold water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish, or wildlife, including invertebrates.

GWR - Ground Water Recharge. Uses of water for natural or artificial recharge of ground water for purposes of future extraction, maintenance of water quality, or halting of saltwater intrusion into freshwater aquifers.

MUN - Municipal and Domestic Supply. Uses of water for community, military, or individual water supply systems including, but not limited to, drinking water supply.

NAV - Navigation. Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels.

RARE - Rare, Threatened, or Endangered Species. Uses of water that support habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened, or endangered.

REC1 - Water Contact Recreation. Uses of water for recreational activities involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, waterskiing, skin and scuba diving, surfing, white water activities, fishing, or use of natural hot springs.

REC2 - Non-contact Water Recreation. Uses of water for recreational activities involving proximity to water, but not normally involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tidepool and marine life study, hunting, sightseeing, or aesthetic enjoyment in conjunction with the above activities.

WARM - Warm Freshwater Habitat. Uses of water that support warm water ecosystems including, but not limited to, preservation or enhancement of aquatic habitats, vegetation, fish, or wildlife, including invertebrates.

WET - Wetland Habitat. Uses of water that support wetland ecosystems, including, but not limited to, preservation or enhancement of wetland habitats, vegetation, fish, shellfish, or wildlife, and other unique wetland functions which enhance water quality, such as providing flood and erosion control, streambank stabilization, and filtration and purification of naturally occurring contaminants.

WILD - Wildlife Habitat. Uses of water that support terrestrial ecosystems including, but not limited to, preservation and enhancement of terrestrial habitats, vegetation, wildlife (e.g., mammals, birds, reptiles, amphibians, invertebrates), or wildlife water and food sources.

These uses are identified as existing (E), potential (P), or intermittent (I) uses. Table 2-1 contains the beneficial use designations relevant to this report (LARWQCB, 1994). All 10 lakes are designated REC1, REC2, and WARM. The majority are also designated WILD and MUN. Other uses include WET, GWR, COLD, RARE, AGR, and NAV. Potential beneficial uses marked with an asterisk (P*) in the Basin Plan (and in the table below) are indicted as a conditional use. Conditional designations are not recognized under federal law and are not water quality standards requiring TMDL development at this time. (See letter from Alexis Strauss [US EPA] to Celeste Cantú [State Board], Feb. 15, 2002.)

Table 2-1. Beneficial Uses Designations for the Ten Lakes

Lake/Reservoir	REC1	REC2	WARM	WILD	MUN	WET	GWR	COLD	RARE	AGR	NAV
Peck Road Park Lake[1]	Pm	E	P	I	P*		I				
Lincoln Park Lake	P	E	P	E	P*						
Echo Park Lake	P	E	P	E	P*						
Lake Calabasas[2]	Pm	I	P	P	P*						
El Dorado Park Lakes	E	E	P	E	P*	E					
North, Center, and Legg Lakes	E	E	E	E	P*	E	E	E			
Puddingstone Reservoir	E	E	E	E	E*		E	E	E	E	
Santa Fe Dam Park Lake	P	I	I	E	P*	E	I				
Lake Sherwood	E	E	E	E	P*	E	E				E
Westlake Lake	E	E	E	E	P*						E

[1] Beneficial uses were not identified in the Basin Plan for Peck Road Park Lake. Therefore, the downstream segment's uses (Rio Hondo below Spreading Grounds) apply (Regional Board, personal communication, 12/22/2009).

[2] Beneficial uses were not identified in the Basin Plan for Lake Calabasas. Therefore, the downstream segment's uses (Arroyo Calabasas) apply (Regional Board, personal communication, 2/24/2009).

* Asterisked MUN designations are designated under SB 88-63 and RB 89-03. Some designations may be considered for exemptions at a later date.

m Access prohibited by Los Angeles County DPW in concrete-channelized areas.

E - Existing; P - Potential; I- Intermittent

2.1.2 Water Quality Objectives and Criteria

The Basin Plan describes numeric and narrative water quality objectives for beneficial uses in the Los Angeles Region (LARWQCB, 1994). The California Toxics Rule (CTR) includes numeric water quality criteria for certain human health and aquatic life designated uses. The objectives and criteria for the impairments addressed in this document are described below.

2.1.2.1 Ammonia

The Basin Plan establishes numeric objectives for ammonia which are protective of fish (COLD and WARM), and wildlife (WILD) (see Basin Plan Tables 3-1 through 3-4). The objective for chronic exposure is based on a four-day average concentration while the objective for acute toxicity is based on a one-hour average concentration. These objectives are expressed as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life.

2.1.2.2 Bioaccumulation

The Basin Plan states that “toxic pollutants shall not be present at levels that will accumulate in aquatic life to levels which are harmful to aquatic life or human health.” To implement this narrative objective, the fish contaminant goals defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) were used to set numeric targets for mercury, chlordane, DDTs, dieldrin, and PCBs.

2.1.2.3 Biostimulatory Substances (nutrients)

The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient (e.g., nitrogen and phosphorous) concentrations in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, “waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses.” To implement this narrative objective, the Numeric Nutrient Endpoint (NNE) BATHTUB model was used to define nitrogen and phosphorus target concentrations on a site specific basis that will not lead to nuisance conditions in the waterbody, such as excessive chlorophyll *a* concentrations.

2.1.2.4 Chemical Constituents

The Basin Plan states that “chemical constituents in excessive amounts in drinking water are harmful to human health” and “surface waters shall not contain concentrations of chemical constituents in amounts that adversely affect any designated beneficial use.” Specifically, waters designated MUN shall not have concentrations exceeding the following maximum contaminant levels: mercury, 0.002 mg/L; nitrate as NO_3, 45 mg/L; nitrate plus nitrite as N, 10 mg/L; nitrite as nitrogen, 1 mg/L; chlordane, 0.0001 mg/L; PCBs, 0.0005 mg/L. The Basin Plan provides maximum contaminant levels for additional pollutants; however, no others are relevant for these TMDLs. The CTR also includes criteria for some of these pollutants (see Section 2.1.2.5).

2.1.2.5 California Toxics Rule

The CTR includes numeric water quality criteria for certain human health and aquatic life designated uses. The strictest applicable targets from those identified in the Basin Plan and CTR apply to the waterbodies in this report. The CTR includes criteria applicable to these lakes for: chlordane, copper, dieldrin, DDT, lead, mercury and PCBs. The specific criteria are described in Section 2.2.

2.1.2.6 Dissolved Oxygen

Adequate dissolved oxygen levels are required to support aquatic life. Dissolved oxygen requirements are dependent on the beneficial uses of the waterbody. The Basin Plan states "At a minimum (see specifics below) the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges" and "the dissolved oxygen content of all surface waters designated as COLD shall not be depressed below 6 mg/L as a result of waste discharges."

2.1.2.7 Floating Material (trash)

The Basin Plan specifies that "waters shall not contain floating materials including solids, liquids, foams, and scum, in concentrations that cause nuisance or adversely affect beneficial uses."

2.1.2.8 Pesticides

The Basin Plan states that "no individual pesticide or combination of pesticides shall be present in concentrations that adversely affect beneficial uses. There shall be no increase in pesticide concentrations found in bottom sediments or aquatic life." To implement this narrative objective, the fish contaminant goals defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) were used to set numeric targets for chlordane, DDTs, and dieldrin. The CTR also includes criteria for some of these pollutants (see Section 2.1.2.5).

2.1.2.9 pH

The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." This narrative objective will be achieved, in nutrient- impaired lakes, by applying the Numeric Nutrient Endpoint (NNE) BATHTUB model, which was used to define nitrogen and phosphorus target concentrations on a site specific basis that will not lead to fluctuations of pH due to excessive algal growth in the waterbody.

2.1.2.10 Polychlorinated Biphenyls (PCBs)

The Basin Plan states that "the purposeful discharge of PCBs to waters of the Region, or at locations where the waste can subsequently reach waters of the Region, is prohibited. Pass-through or uncontrollable discharges to waters of the Region, or at locations where the waste can subsequently reach water of the Region, are limited to 70 pg/L (30-day average) for protection of human health and 14 ng/L and 30 ng/L (daily average) to protect aquatic life in inland fresh waters and estuarine waters respectively." In addition, OEHHA (2008) has published fish consumption guidelines for PCBs that were used to set fish tissue targets. The CTR also includes a criterion for PCBs (see Section 2.1.2.5).

2.1.2.11 Taste and Odor

The Basin Plan states that "waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses." This narrative objective will be achieved, as it relates to nutrient-related odor impairments, by applying the Numeric Nutrient Endpoint (NNE) BATHTUB model, which was used to define nitrogen and phosphorus target concentrations on a site specific basis that will not lead to nuisance algal growth in the waterbody. Additionally, trash TMDLs will further address this impairment in applicable lakes.

2.1.2.12 Toxicity

The Basin Plan states that "all waters shall be maintained free of toxic substances in concentrations that are toxic to, or that produce detrimental physiological response in human, plant, animal, or aquatic life."

2.1.2.13 Antidegradation

State Board Resolution 68-16, "Statement of Policy with Respect to Maintaining High Quality Water in California," known as the "Antidegradation Policy," protects surface and ground waters from degradation. Any actions that can adversely affect water quality in all surface and ground waters must be consistent with the maximum benefit to the people of the state, must not unreasonably affect present and anticipated beneficial use of such water, and must not result in water quality less than that prescribed in water quality plans and policies. Furthermore, any actions that can adversely affect surface waters are also subject to the federal Antidegradation Policy (40 CFR 131.12). The proposed TMDLs will not degrade water quality, and will in fact improve water quality as they will lead to meeting the numeric water quality standards.

2.2 NUMERIC TARGETS

Numeric targets represent water column, sediment, or fish tissue concentrations that result in attainment of the water quality standards. For the TMDLs in this document, the targets are assigned based on either: 1) numeric water quality objectives outlined in the Basin Plan, 2) fish contaminant goals (FCG) defined by the Office of Environmental Health Hazard Assessment, 3) water concentrations defined by the California Toxics Rule (CTR), 4) consensus-based sediment quality guidelines defined by MacDonald et al. (2000), 5) bioaccumulation factor (BAF) or biota-sediment accumulation factor (BSAF) calculations to translate the FCGs into water and sediment targets respectively, or 6) interpretation of the Regional Board regarding narrative water quality objectives.

2.2.1 Ammonia

The Basin Plan expresses ammonia targets as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life. In order to assess compliance with the standard, pH, temperature, and ammonia must be determined at the same time. The toxicity of ammonia increases with increasing pH and temperature; therefore, ammonia targets depend on the site specific pH and temperature as well as the presence or absence of early life stages (ELS) of aquatic life. For the purpose of this report, pH and temperature samples at the surface (less than 0.5 meters of depth) were used to determine the median temperature and 95th percentile pH, which were then used to calculate chronic targets. Acute values were based entirely on the 95th percentile pH. Any single day sample without a depth was assumed to be sampled at the surface and included within the target calculation.

A December 2005 Amendment to the Basin Plan assumes that ELS are present in any waterbody designated as COLD. Designated uses applied in the calculation of site-specific ammonia targets are presented in Table 2-2. The 30-day average target concentrations (criterion continuous concentration (CCC)) of ammonia for waterbodies with and without ELS can be calculated using Equations 2-1 and 2-2, respectively. Concentration targets are also presented in Tables 3-1 through 3-4 of the Basin Plan (LARWQCB, 1994). The four-day maximum average concentrations shall not exceed 2.5 times the 30-day average objective, while the one-hour acute level, with and without ELS, can be calculated with Equations 2-3 and 2-4, respectively (USEPA, 1999).

Table 2-2. Temperature and pH Dependent Acute and Chronic Total Ammonia Targets (un-ionized ammonia target)

Lake (designated use)	Median Temperature (n = number of samples)	95th% pH Values (n = number of samples)	Acute (1-hr Maximum Concentration) (mg-N/L)[1]	Four-day Ammonia Max Average (mg-N/L)[2]	Chronic Ammonia Target (mg-N/L)[3]
Lincoln Park (WARM, WILD)	19.0 (n=8)	9 (n=22)	1.32	0.91	0.36
Echo Park (WARM, WILD)	19.7 (n=44)	9.1 (n=60)	1.14	0.76	0.30
Calabasas (WARM)	21.8 (n=144)	9.4 (n=172)	0.78	0.46	0.19
El Dorado Park (WARM, WILD)	16.2 (n=46)	8.5 (n=46)	3.20	2.44	0.98
Legg (COLD)**	16 (n=14)	9.6 (n=30)	0.42**	0.56**	0.23**

Note: The median temperature and 95th percentile pH values were calculated from the observed surface depth data and used in the calculation of ammonia targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan.

[1] The acute criterion represents a short term one-hour maximum concentration.

[2] The four-day criterion is the maximum average concentration allowed in a four-day period.

[3] The chronic criterion is the maximum 30 day average.

**ELS assumed to be present.

Equation 2-1: 30-day average total ammonia concentration for waterbodies with ELS present.

$$\text{30-day Average Concentration} = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) * MIN\left(2.85, 1.45 * 10^{0.028*(25-T)}\right)$$

Equation 2-2: 30-day average total ammonia concentration for waterbodies with ELS absent.

$$\text{30-day Average Concentration} = \left(\frac{0.0577}{1+10^{7.688-pH}} + \frac{2.487}{1+10^{pH-7.688}}\right) * 1.45 * 10^{0.028*(25-MAX(T,7))}$$

Equation 2-3: Acute criteria for total ammonia-nitrogen for waterbodies with ELS absent (USEPA, 1999).

$$\text{Acute Limit} = \left(\frac{0.41}{1+10^{7.204-pH}}\right) + \left(\frac{58.4}{1+10^{pH-7.204}}\right)$$

Equation 2-4: Acute criteria for total ammonia-nitrogen for waterbodies with ELS present (USEPA, 1999).

$$\text{Acute Limit} = \left(\frac{0.267}{1+10^{7.204-pH}}\right) + \left(\frac{39.0}{1+10^{pH-7.204}}\right)$$

2.2.2 Chlordane

Targets associated with OC Pesticides and PCBs are provided to ensure protection of both human health and wildlife, consistent with the beneficial uses associated with the OC Pesticides and PCBs-impaired waterbodies. The OC Pesticides and PCBs targets considered for use in calculating the TMDLs are discussed below by media.

2.2.2.1 Selection of Water Quality Targets

Water column targets for OC Pesticides and PCBs are based on beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level associated with chlordane and PCBs. The Basin Plan also requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this document is designated WARM, at a minimum, and must meet this requirement. The WQOs intended to protect these beneficial uses defer to numeric water quality criteria included in the California Toxics Rule (CTR) (USEPA, 2000a). To meet the designated beneficial uses, the aquatic life and human health criteria must be met. Acute and chronic criterion in freshwater systems are considered protective of aquatic life. However, the most stringent water column targets are the criteria for protection of human health. The “water and organisms” criterion is applicable to Puddingstone Reservoir, where there is an existing MUN use, while the “organisms only” criterion is applicable to Echo Park Lake and Peck Road Park Lake. The CTR criteria for “water and organisms” or “organisms only” both account for human health risk associated with bioaccumulation directly from the water column.

2.2.2.2 Selection of Sediment Quality Targets

OC Pesticides and PCBs have an affinity for organic matter and will partition from water to organic substances such as sediment, benthic organisms, and fish. The levels of contamination in sediment are important because they are a crucial pathway for pollutant accumulation in fish and other edible species (such as clams and mussels). Partitioning of OC Pesticides and PCBs from water through fish skin is also important, but does not result in the high accumulation caused by the continuous ingestion of contaminated organisms in most fish species. Two target sediment concentrations have been identified that consider the protection of sediment biota and the potential for bioaccumulation in aquatic organisms, as well as the associated hazards to the species that consume aquatic organisms. Consensus-based threshold effect levels are described in Section 2.2.2.2.1 and are designed to protect benthic biota from excessive toxic pollutants. These sediment targets have been used in similar freshwater OC Pesticides and PCBs TMDLs in the Los Angeles region. The other type of sediment targets, included in section 2.2.2.2.2, were calculated to attain the fish tissue target based on a biota-sediment accumulation factor (BSAF). The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target for each lake. Additionally, these TMDLs include alternative wasteload allocations to be applied when a sufficient demonstration has been made that the fish tissue targets are met. These targets are based on the consensus-based TEC values described below. Details on when each set of targets apply are included in the wasteload allocation section of each relevant lake chapter.

2.2.2.2.1 Consensus-Based Sediment Quality Guidelines Threshold Effects Concentrations (consensus-based TECs)

There are no WQOs in the Basin Plan for OC Pesticides and PCBs in sediments. Instead, the Regional Board assesses the quality of the lake sediments using the Probable Effects Concentration (PEC) values for the consensus-based sediment quality guidelines published by MacDonald et al. (2000). The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008). Sediment quality guidelines (SQGs) are developed from field and laboratory studies to predict the toxicity of pollutants on sediment-dwelling organisms. MacDonald et al. (2000) compiled a set of all published SQGs and used the resulting geometric mean value to establish CBSQGs for threshold and probable effect concentrations of individual contaminants. The PEC is the concentration at which harmful effects on sediment-dwelling organisms are expected to occur, whereas the threshold effect concentration (TECs) describes the level of contaminant that is not expected to have harmful effects on sediment-dwelling organisms. PECs are appropriate when assessing impairments, while TECs are more conservative and best used as the targets for the TMDLs. The consensus-based sediment quality guidelines are designed to protect benthic dwelling organisms.

2.2.2.2.2 Biota-Sediment Accumulation Factor (BSAF)

To ensure protection of both human health and wildlife, it is also important to consider the potential for bioaccumulation in aquatic organisms and the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans). Thus a separate target calculation was conducted to ensure that fish tissue concentration goals are supported by sediment concentration. The fish goals may be translated through biota-sediment accumulation factor (BSAF) calculations to estimate associated sediment targets. This is done on a site-specific basis.

Specifically, a sediment target to achieve FCGs (see Selection of Fish Targets below) can be calculated based on biota-sediment bioaccumulation (a BSAF approach), using the ratio of the FCG to existing fish tissue concentrations. This ratio is applied to the observed in-lake sediment concentration to obtain the site-specific sediment target concentration to achieve fish tissue goals. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations are likely to have declined steadily since the cessation of production and use of the OC Pesticides and PCBs.

2.2.2.3 Selection of Fish Tissue Targets

Beneficial uses may also be impaired if concentrations of OC Pesticides and PCBs in fish tissue are sufficiently high to pose potential adverse health impacts from the ingestion of sport-caught or local fish. Tissue concentrations of OC Pesticides and PCBs biomagnify in the food chain. OC Pesticides and PCBs levels increase with the species' trophic level and organisms at the top of a food chain system will have the highest accumulation of OC Pesticides and PCBs (note: trophic levels describe the position an organism occupies in the food chain [i.e., what the organism eats and what eats the organism] and are described in greater detail below). The OC Pesticides and PCBs accumulation also increases with the age of the organisms and resides mostly in the lipid portions of the fish. The top predators and fatty fish species in a given lake system tend to have the highest concentrations of OC Pesticides and PCBs, but concentrations are also elevated in fish that feed directly in contaminated sediment. Top predators (such as bass) are often target species for sport fishermen. Risks to human health from the consumption of contaminated fish are based on long-term, cumulative effects, rather than concentrations in individual fish. Therefore, the criterion should not be applied to the extreme case of the most-contaminated fish within a target species; instead, the criterion is most applicable to average concentrations in top predator species and fish that are popular for consumption.

The Office of Environmental Health Hazard Assessment (OEHHA) describes fish contaminant goals (FCGs) as pollutant levels in fish that "pose no significant health risk to individuals consuming sport fish at a standard consumption rate of eight ounces per week (32 g/day), prior to cooking, over a lifetime…" OEHHA also states that FCGs provide a reasonable starting point for criteria development (OEHHA, 2008).

FCGs for OC Pesticides and PCBs are defined for carcinogenic and non- carcinogenic risks. The OEHHA (2008) applied the following methodology to calculate the two sets of FCGs:

> *For each chemical, the toxicological literature was reviewed to establish an acceptable non-cancer reference dose (RfD; an estimate of daily human exposure to a chemical that is likely to be without significant risk of adverse effects during a lifetime) and/or a cancer slope factor (an upper-bound estimate of the probability that an individual will develop cancer over a lifetime as a consequence of exposure to a given dose of a specific carcinogen).*

For all the OC Pesticides and PCBs of concern in these TMDLs, the FCG based on cancer risk is the lower of the two FCG sets and is selected as the target.

2.2.2.4 Chlordane Numeric Targets

Total chlordane consists of a family of related chemicals, including cis- and trans-chlordane, oxychlordane, trans-nonachlor, and cis-nonachlor. As described above, water column targets for chlordane are based on beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0001 mg/L, or 0.1 µg/L (100 ng/L). The Basin Plan also requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). This objective is addressed through the CTR water quality criteria.

Acute and chronic criteria for chlordane in freshwater systems are defined by the California Toxics Rule as 2.4 µg/L (2,400 ng/L) and 0.0043 µg/L (4.3 ng/L), respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. The CTR also includes human health criteria for the consumption of water and organisms and for the consumption of organisms only as 0.00057 µg/L (0.57 ng/L) and 0.000059 µg/L (0.59 ng/L), respectively (USEPA, 2000a). California often implements these values on a 30 day average. Because the human health criterion for the consumption of water and organisms is the most restrictive criterion, a water column target of 0.00057 µg/L (0.57 ng/L) is the appropriate target for waterbodies with the MUN designated use (Puddingstone Reservoir). The human health criterion for the consumption of organisms only (0.000059 µg/L [0.59 ng/L]) is appropriate for waterbodies without an existing MUN designation (Echo Park Lake and Peck Road Park Lake).

Two target sediment concentrations for chlordane have been identified as potential targets (Section 2.2.2.2). There are no Basin Plan Objectives for toxicity levels in sediment; however sediment quality guidelines are reported by multiple agencies for the protection of sediment biota. MacDonald et al. (2000) compiled and evaluated the guidelines and derived consensus-based sediment quality guidelines that incorporate multiple recommendations. For chlordane, the consensus-based threshold effect concentration (TEC) is 3.24 µg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. An additional sediment target based on bioaccumulation in fish was also calculated for each impaired lake to ensure that the FCG is met using the BSAF approach described in Section 2.2.2.2.2. The lower of the two sediment target values is applied in each lake.

Fish tissue targets are described above in Section 2.2.2.3. The fish contaminant goal for chlordane defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 5.6 ppb based on cancer risk (the FCG based on non-cancer risk is 100 ppb). The resulting total chlordane targets for each lake are shown in Table 2-3.

Table 2-3. Total Chlordane Targets

Lake	Maximum Contaminant Level (ng/L)	Acute Criterion[1] (ng/L)	Chronic Criterion[2] (ng/L)	Criterion for Consumption of Water and Organisms (ng/L)	Human Health Criterion for Consumption of Organisms Only (ng/L)	Consensus-based TEC Sediment Target (µg/kg)[3]	BSAF-derived Sediment Target (µg/kg)	Fish Contaminant Goal (ppb)
Echo Park Lake	NA	2,400	4.3	0.57	0.59	3.24	2.10	5.6
Peck Road Park Lake	NA	2,400	4.3	0.57	0.59	3.24	1.73	5.6
Puddingstone Reservoir	100	2,400	4.3	0.57	0.59	3.24	0.75	5.6

Note: Shaded cells represent the selected targets for each waterbody.

[1] The acute criterion is a short term average not to be exceeded more than once every three years on the average.

[2] The chronic criterion is the highest four day average not to be exceeded more than once every three years on average.

[3] The consensus-based TEC sediment target value was used for setting alternative wasteload allocations when sufficient demonstration that the fish tissue targets are met has been made. Details on when each set of targets apply are included in the wasteload allocation sections of each relevant lake chapter.

2.2.3 Chlorophyll *a*, Total Nitrogen, and Total Phosphorus

To address the water quality standard for biostimulatory substances (nitrogen and phosphorus), the Regional Board and USEPA have determined that an average summer (May – September) and annual mean chlorophyll *a* concentration of 20 µg/L will protect each waterbody from nuisance aquatic growth. For lakes that are not meeting the chlorophyll *a* target, the NNE BATHTUB model was used to assess target concentrations of nitrogen and phosphorus in each waterbody that will not result in an average summer (May – September) and annual mean chlorophyll *a* concentration exceeding 20 µg/L. The unique conditions in each lake result in unique total nitrogen and total phosphorus targets for each lake that will result in the targeted chlorophyll *a* concentration. For lakes where currently available data indicate the chlorophyll *a* target is being met, the total nitrogen and total phosphorus targets are set at existing nutrient levels. More information on nutrient targets is included below.

2.2.3.1 Chlorophyll *a* Numeric Targets

A summer mean chlorophyll *a* concentration of 25 µg/L represents a general consensus for the boundary between eutrophic and degraded hypereutrophic conditions (Welch and Jacoby, 2004), and average concentrations should be maintained below this level to protect WARM uses. Impairment of recreational uses can occur at somewhat lower levels. Carlson (1977) shows that an average chlorophyll *a* concentration of around 20 µg/L corresponds to a Secchi disc depth of 3 m. The work of Walker (1987) suggests that a mean chlorophyll *a* concentration of 25 µg/L is associated with severe algal blooms (concentration greater than 30 µg/L) occurring about one quarter of the time, while a mean concentration of 20 µg/L should reduce the frequency of severe blooms to about 15-20 percent of the time. Lake aesthetics and recreation potential are generally found to be impaired above about 20 or 25 µg/L chlorophyll *a* (Bachmann and Jones, 1974; Heiskary and Walker, 1988). Based on these and other lines of evidence, Tetra Tech (2006) recommended to the State Water Quality Control Board that summer average chlorophyll *a* concentrations be not greater than 25 µg/L to support WARM uses and not greater than 20 µg/L to support REC-1 uses.

2.2.3.2 Total Nitrogen and Total Phosphorus Numeric Targets

As mentioned above the NNE BATHTUB Tool was used to calculate total nitrogen and total phosphorus targets for each lake. Appendix A (Nutrient TMDL Development) provides more details but a brief description is included here. The NNE BATHTUB tool finds combinations of N and P loading that result in predicted chlorophyll *a* being equal to the selected target. Similar to the chlorophyll *a* targets, the total nitrogen and total phosphorus targets are average summer (May – September) and annual mean values. Because algal growth can be limited by either N or P there is not a unique solution, and the Tool output supplies the user with a curve representing the loading combinations that will result in attainment of the selected chlorophyll *a* target. The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected. This ratio was chosen to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). A ratio of 10 typically limits the growth nuisance species, such as cyanobacteria (blue green algae) (Welch and Jacoby, 2004). For lakes with required reductions in loadings, maximum allowable alternative “Approved Lake Management Plan Wasteload Allocations” are also included. These alternative wasteload allocations are concentration-based and are based on USEPA’s technical guidance to States not to set phosphorus criteria for lakes and reservoirs any higher than 0.1 mg/L total phosphorus (USEPA, 2000d). A ratio of 10 was then applied to select the corresponding maximum allowable total nitrogen target.

For lakes where the currently available data indicate that the chlorophyll *a* target is being met, the total nitrogen target is based on the existing conditions and the total phosphorus target is based on the typical ratio of 10 between phosphorus and nitrogen in natural systems. The in-lake nitrogen and phosphorus targets as well as the chlorophyll *a* target are summer (May – September) and annual average values. However, compliance with these targets for the lakes that are receiving targets based on existing conditions will be based on a three year average to account for year to year variability. Table 2-4 presents the total phosphorous and total nitrogen targets associated with each lake.

Measuring compliance with the nitrogen and phosphorus targets will occur differently for three categories of lakes. The first category includes lakes where the currently available data indicate that the chlorophyll *a* target is being met. In these lakes compliance with the total phosphorus and total nitrogen allocations is based on a three year average rather than a one year value. Additionally, if applicable water quality criteria for ammonia, dissolved oxygen, and pH and the chlorophyll *a* target are met then the total phosphorus and total nitrogen allocations are considered attained. The second category includes lakes that require reductions to achieve the chlorophyll *a* target and are heavily managed lakes that receive the majority of their water from supplemental water additions to the lake. Responsible jurisdictions that discharge to these lakes may opt to request that alternative wasteload and load allocations apply to them if they develop a lake management plan. In this scenario if applicable water quality criteria for ammonia, dissolved oxygen, and pH and the chlorophyll *a* target are met then the total phosphorus and total nitrogen allocations are considered attained. Finally, the third category of lake is for lakes that require reductions to achieve the chlorophyll *a* target but are not heavily managed lakes and do not receive the majority of their water from supplemental water additions. The only lake in this category is Puddingstone Reservoir. Responsible jurisdictions that discharge to this lake must meet the total phosphorus and total nitrogen allocations as well as the applicable water quality criteria for ammonia, dissolved oxygen, and pH and the chlorophyll *a* target in order to demonstrate compliance. Details are included in the individual lake chapters.

Table 2-4. Total Phosphorus and Total Nitrogen Targets

Lake/Reservoir	Total Phosphorus Target (mg-P/L)	Total Nitrogen Target (mg-N/L)	Maximum Allowable Alternative target for Total Phosphorus (mg-P/L)	Maximum Allowable Alternative target Total Nitrogen (mg-N/L)
Peck Road Park Lake[1]	0.071	0.71	NA	NA
Lincoln Park Lake	0.088	0.88	0.1[2]	1.0[2]
Echo Park Lake[1]	0.12	1.20	NA	NA
Lake Calabasas	0.066	0.66	0.1[2]	1.0[2]
El Dorado Park Lakes Northern System	0.069	0.69	0.1[2]	1.0[2]
El Dorado Park Lakes Southern System[1]	0.125	1.25	NA	NA
Legg Lakes	0.065	0.65	0.1[2]	1.0[2]
Puddingstone Reservoir	0.071	0.71	0.1	1.0
Santa Fe Dam Park Lake[1]	0.063	0.63	NA	NA

[1] Limited data indicate these lakes are meeting the chlorophyll *a* target so the total nitrogen and total phosphorus targets are based on existing conditions. In these lakes compliance with the total phosphorus and total nitrogen allocations is based on a three year average rather than a one year value. Additionally, if applicable water quality criteria for ammonia, dissolved oxygen, and pH and the chlorophyll *a* target are met then the total phosphorus and total nitrogen allocations are considered attained.

[2] In these lakes responsible jurisdictions can request that these alternative allocations are applied to them based on factors set out in the individual lake chapters' wasteload and load allocation sections. Additionally, if applicable water quality criteria for ammonia, dissolved oxygen, and pH and the chlorophyll *a* target are met then the total phosphorus and total nitrogen allocations under the alternative allocations scenario are considered attained.

2.2.4 Copper

The Basin Plan requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Acute and chronic criterion for copper and lead in freshwater systems are included in the California Toxics Rule (CTR) 40 CFR 131.38. (USEPA, 2000a). The CTR establishes short-term (acute) and long-term (chronic) aquatic life criteria for metals in both freshwater and saltwater. The acute criterion, defined in the CTR as the Criteria Maximum Concentration, equals the highest concentration of a pollutant to which aquatic life can be exposed for a short period of time without deleterious effects. The chronic criterion, defined in the CTR as the Criteria Continuous Concentration, equals the highest concentration of a pollutant to which aquatic life can be exposed for an extended period of time (4 days) without deleterious effects.

CTR freshwater aquatic life criteria for certain metals are expressed as a function of hardness because hardness and/or water quality characteristics that are usually correlated with hardness can reduce or increase the toxicity of some metals. In order to assess compliance with the standards, copper and hardness should be determined at the same time. Hardness is used as a surrogate for a number of water quality characteristics, which affect the toxicity of metals in a variety of ways. Increasing hardness generally has the effect of decreasing the toxicity of metals. Water quality criteria to protect aquatic life may be calculated at different concentrations of hardness measured in milligrams per liter (mg/L) as calcium carbonate ($CaCO_3$). The CTR lists freshwater aquatic life criteria based on a hardness value of 100 mg/L and provides hardness dependent equations to calculate the freshwater aquatic life metals criteria using site-specific hardness data.

In the CTR, freshwater criteria for metals are expressed in terms of the dissolved fraction of the metal in the water column. These criteria were calculated based on methods in USEPA guidance (USEPA, 1985) developed under Section 304(a) of the CWA. This methodology is used to calculate the total recoverable fraction of metals in the water column and then appropriate conversion factors, included in the CTR, are applied to calculate the dissolved criteria.

The CTR allows for the adjustment of criteria through the use of a water-effect ratio (WER) to assure that the metals criteria are appropriate for the site-specific chemical conditions under which they are applied. A WER represents the ratio between metals that are measured and metals that are biologically available and toxic. The WER is used to account for site specific conditions that may alter the bioavailability of a toxicant with respect to laboratory water. For impaired waterbodies where no site specific data are available, a default WER of 1 can be assumed. The coefficients needed for hardness-based calculations are provided in the CTR and listed below in Table 2-5.

The equations for calculating the freshwater criteria for metals are:

$$\text{Acute Criterion} = \text{WER x ACF x EXP}[(m_a)(\ln(\text{hardness}))+b_a] \qquad \textbf{Equation 2-5}$$

$$\text{Chronic Criterion} = \text{WER x CCF x EXP}[(m_c)(\ln(\text{hardness}))+b_c] \qquad \textbf{Equation 2-6}$$

Where: WER = Water-Effect Ratio (assumed to be 1)

ACF = Acute conversion factor (to convert from the total to the dissolved fraction)

CCF = Chronic conversion factor (to convert from the total to the dissolved fraction)

m_a = slope factor for acute criteria

m_c = slope factor for chronic criteria

b_a = y intercept for acute criteria

b_c = y intercept for chronic criteria

Table 2-5. Coefficients used in Formulas for Calculating CTR Freshwater Criteria for Copper

Metal	ACF	m_a	b_a	CCF	m_C	b_C
Copper	0.960	0.9422	-1.700	0.960	0.8545	-1.702

Chronic copper freshwater targets for each lake are calculated based on the 50^{th} percentile of hardness values measured during copper sampling events, while the acute targets are calculated using the 90^{th} percentile hardness (Appendix G, Monitoring Data). These are presented as example calculations since the actual target varies with the hardness value measured during sample collection. Table 2-6 summarizes the acute and chronic criteria, as well as the human health criterion for the consumption of water and organisms from a waterbody, for each lake impaired by copper.

Table 2-6. Hardness-Dependent Acute and Chronic Copper Targets

Lake	WER	90th Percentile Hardness (mg/L as $CaCO_3$)	Acute Criterion[1] (µg/L dissolved fraction)	50th Percentile Hardness (mg/L as $CaCO_3$)	Chronic Criterion[2] (µg/L dissolved fraction)	Human Health Criterion[3] (µg/L total fraction)
Echo Park Lake	1	231	29.58	208	16.75	1,300
El Dorado Park Lakes	1	124	16.46	95	8.57	1,300
Legg Lakes	1	246	31.38	182	14.94	1,300
Santa Fe Dam Park Lake	1	131	17.33	100	8.96	1,300

Note: The median and 90th percentile hardness values were calculated from the observed data and used in the calculation of the chronic and acute targets, respectively. These are presented as example calculations since the actual target varies with the hardness value determined during sample collection.

[1] The acute criterion is a short term average not to be exceeded more than once every three years on the average.

[2] The chronic criterion is the highest four day average not to be exceeded more than once every three years on average.

[3] The human health criterion was specified for consumption of water and organisms. A human health criterion was not specified for consumption of organisms only.

2.2.5 Dieldrin

Selection of applicable OC Pesticides and PCBs targets are described above in Section 2.2.2.1 through Section 2.2.2.3. Water column targets for dieldrin are based on beneficial use (Section 2.2.2.1). Only one of the three dieldrin-impaired waters has an MUN designated use. The Basin Plan requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). This objective is addressed through the CTR water quality criteria.

Acute and chronic criteria for the protection of aquatic life in freshwater systems are included in the CTR for dieldrin as 0.24 µg/L (240 ng/L) and 0.056 µg/L (56 ng/L), respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. The CTR also includes human health criterion for the consumption of organisms only and for the consumption of organisms and water as 0.00014 µg/L (0.14 ng/L) (USEPA, 2000a). California often implements these values on a 30 day average. Because the human health criterion for the consumption of organisms only is the most restrictive criterion, a water column target of 0.00014 µg/L (0.14 ng/L) is the appropriate target for waterbodies without an existing MUN designated use (Echo Park Lake and Peck Road Park Lake). For the MUN use specified in Puddingstone Reservoir the CTR criterion is based on consumption of organisms and water, but is also equal to 0.00014 µg/L (0.14 ng/L).

Two target sediment concentrations for dieldrin have been identified (Section 2.2.2.2). There are no Basin Plan Objectives for toxicity levels in sediment; however sediment quality guidelines are reported by multiple agencies for the protection of sediment biota. MacDonald et al. (2000) compiled and evaluated the guidelines and derived consensus-based sediment quality guidelines that incorporate multiple recommendations. For dieldrin, the consensus-based threshold effect concentration (TEC) is 1.9 µg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. An additional sediment target based on bioaccumulation in fish was also calculated for each impaired lake to ensure that the FCG is met using the BSAF approach described in Section 2.2.2.2.2. The lower of the two sediment target values is applied in each lake. Additionally, these TMDLs include alternative wasteload allocations to be applied when a sufficient demonstration has been made that the fish tissue targets are met. These targets are based on the consensus-based TEC values. Details on when each set of targets apply are included in the wasteload allocation section of each relevant lake chapter.

Fish tissue targets are described above in Section 2.2.2.3. The fish contaminant goal for dieldrin defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 0.46 ppb based on cancer risk (the FCG based on non-cancer risk is 160 ppb). Similar to the sediment targets, the lowest fish tissue target value is applied in each lake. Table 2-7 summarizes the applicable targets for the two waterbodies listed for dieldrin addressed by this document.

Table 2-7. Dieldrin Targets

Lake	Acute Criterion[1] (ng/L)	Chronic Criterion[2] (ng/L)	Human Health Criterion for Consumption of Organisms Only (ng/L)	Consensus-based TEC Sediment Target (µg/kg)[3]	BSAF-derived Sediment Target (µg/kg)	Fish Contaminant Goal (ppb)
Echo Park Lake	240	56	0.14	1.90	0.80	0.46
Peck Road Park Lake	240	56	0.14	1.90	0.43	0.46
Puddingstone Reservoir	240	56	0.14	1.90	0.22	0.46

Note: Shaded cells represent the selected targets for each waterbody.

[1] The acute criterion is a short term average not to be exceeded more than once every three years on the average.

[2] The chronic criterion is the highest four day average not to be exceeded more than once every three years on average.

[3] The consensus-based TEC sediment target value was used for setting alternative wasteload allocations when sufficient demonstration that the fish tissue targets are met has been made. Details on when each set of targets apply are included in the wasteload allocation sections of each relevant lake chapter.

2.2.6 Dissolved Oxygen

Targets for dissolved oxygen (DO) depend on whether or not the waterbody is designated COLD in addition to the minimum designation of WARM, as is the case with Puddingstone Reservoir. Waterbodies designated COLD have more stringent dissolved oxygen targets. Table 2-8 summarizes the DO targets for each lake listed as impaired by low DO. Targets are specified as minimum values not to be depressed due to waste discharges. Target depths for each lake were set by the Regional Board and USEPA based on site specific conditions. Shallow, well mixed lakes must meet the target in the water column from the surface to 0.3 meters above the bottom of the lake. Deeper lakes that thermally stratify during the summer months, such as Peck Road Park Lake and Puddingstone Reservoir, must meet the DO target throughout the epilimnion of the water column.

The epilimnion is the upper stratum of more or less uniformly warm, circulating, and fairly turbulent water during summer stratification. The epilimnion floats above a cold relatively undisturbed region called the hypolimnion. The stratum between the two is the metalimnion and is characterized by a thermocline, which refers to the plane of maximum rate of decrease of temperature with respect to depth. For the purposes of these TMDLs, the presence of stratification will be defined by whether there is a change in lake temperature greater than 1 degree Celsius per meter. Deep lakes must meet the DO target in the water column from the surface to 0.3 meters above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the DO target must be met in the epilimnion, the portion of the water column above the thermocline.

Table 2-8. Dissolved Oxygen Targets

Lake/Reservoir	Minimum Mean Annual DO (mg/L)[1]	Minimum Instantaneous DO (mg/L)[2]	Target Depth (m)
Peck Road Park Lake	7.0	5.0	Throughout the epilimnion
Lincoln Park Lake	7.0	5.0	Surface to 0.3 meters above the bottom
Echo Park Lake	7.0	5.0	Surface to 0.3 meters above the bottom
Lake Calabasas	7.0	5.0	Surface to 0.3 meters above the bottom
El Dorado Park Lakes	7.0	5.0	Surface to 0.3 meters above the bottom
Legg Lakes	7.0	6.0	Surface to 0.3 meters above the bottom
Puddingstone Reservoir	7.0	6.0	Throughout the epilimnion
Santa Fe Dam Park Lake	7.0	5.0	Surface to 0.3 meters above the bottom

[1] The mean annual dissolved oxygen concentration shall be greater than 7 mg/L except when natural conditions cause lesser concentrations.

[2] The dissolved oxygen content shall not be depressed below this level as a result of waste discharges.

2.2.7 DDT

Dichlorodiphenyltrichloroethane (DDT) is a synthetic organochlorine insecticide once used throughout the world to control insects. Technical DDT consists of two isomers, 4,4’-DDT and 2,4’-DDT, of which the former is most toxic. In the environment, DDT breaks down to form two related compounds: DDD (tetrachlorodiphenylethane) and DDE (dichlorodiphenyl-dichloroethylene). DDD and DDE often predominate in the environment and USEPA (2000c) recommends that fish consumption guidelines be based on the sum of DDT, DDD, and DDE – collectively referred to as total DDTs.

Selection of applicable OC Pesticides and PCBs targets are described above in Section 2.2.2.1 through Section 2.2.2.3. Water column targets for DDT are based on beneficial use (Section 2.2.2.1). The Basin Plan requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). This objective is addressed through the CTR water quality criteria. Acute and chronic criteria for 4,4’-DDT in freshwater systems are included in the CTR as 1.1 µg/L (1,100 ng/L) and 0.001 µg/L (1 ng/L), respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. Acute and chronic values for other DDT compounds were not specified.

The CTR also includes human health criteria for the consumption of water and organisms or organisms only in several DDT compounds, but does not specify a target for total DDTs (USEPA, 2000a). California often implements these values on a 30 day average. These values include a water column target of 0.00059 µg/L (0.59 ng/L) for 4,4’-DDT for consumption of water and organisms as well as organisms only. The CTR also specifies a criterion of 0.00059 µg/L (0.59 ng/L) for 4,4’-DDE (for both consumption of water and organisms or organisms only), while for 4,4’-DDD the criteria are 0.00083 µg/L (0.83 ng/L) for consumption of water and organisms and 0.00084 µg/L (0.84 ng/L) for consumption of organisms only. The lowest applicable DDT target is selected for the purposes of representing Total DDTs. If analytical results that resolve individual DDT compounds are available, all of the CTR criteria should be applied individually. Because the human health criterion for the consumption of water and organisms is the most restrictive criterion, a water column target of 0.00059 µg/L (0.59 ng/L) is the appropriate target for waterbodies with the MUN designated use (Puddingstone Reservoir). The human health criterion for the consumption of organisms only (0.00059 µg/L [0.59 ng/L]) is appropriate for waterbodies without an existing MUN designated use (Peck Road Park Lake).

Two target sediment concentrations for total DDT have been identified (Section 2.2.2.2). There are no Basin Plan Objectives for toxicity levels in sediment; however sediment quality guidelines are reported by multiple agencies for the protection of sediment biota. MacDonald et al. (2000) compiled and evaluated the guidelines and derived consensus-based sediment quality guidelines that incorporate multiple recommendations. The consensus-based TEC for total DDTs is 5.28 μg/kg dry weight (MacDonald el al., 2000). Most data are provided for the total compound; therefore, the total DDTs TEC value is applicable for TMDL analyses. If data for individual compounds are available, separate TECs are also provided: for 4,4'- plus 2,4'-DDT the TEC is 4.16 μg/kg dry weight, for total DDE the TEC is 3.16 μg/kg dry weight, and the TEC for total DDD is 4.88 μg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. An additional sediment target based on bioaccumulation in fish was also calculated for each impaired lake to ensure that the FCG is met using the BSAF approach described in Section 2.2.2.2.2. The lower of the two sediment target values is applied in each lake. Additionally, the Puddingstone Reservoir DDT TMDL includes alternative wasteload allocations to be applied when a sufficient demonstration has been made that the fish tissue targets are met. This target is based on the consensus-based TEC values. Details on when each set of targets apply are included in the wasteload allocation section of the Puddingstone Reservoir DDT impairment chapter.

Fish tissue targets are described above in Section 2.2.2.3. The fish contaminant goal for total DDT defined by the OEHHA is 21 ppb (OEHHA, 2008) based on cancer risk (the FCG based on non-cancer risk is 1,600 ppb). The advisory tissue levels are based on various levels of fish consumption. Table 2-9 summarizes the applicable targets for the two waterbodies listed for DDT addressed by this document.

Table 2-9. DDT Target

Lake	Acute Criterion[1] for 4-4'-DDT (ng/L)	Chronic Criterion[2] for 4-4'-DDT (ng/L)	Human Health Criterion for Consumption of Water and Organisms (ng/L)	Human Health Criterion for Consumption of Organisms Only (ng/L)	Consensus-based TEC Sediment Target (µg/kg)	BSAF-derived Sediment Target (µg/kg)	Fish Contaminant Goal (ppb)
Peck Road Park Lake	1,100	1	0.59	0.59[3]	5.28	6.90	21
Puddingstone Reservoir	1,100	1	0.59[3]	0.59	5.28[4]	3.94	21

Note: Shaded cells represent the selected targets for each waterbody.

[1] The acute criterion is a short term average not to be exceeded more than once every three years on the average.

[2] The chronic criterion is the highest four day average not to be exceeded more than once every three years on the average.

[3] The target water column concentration of 0.59 ng/L specified in the CTR is for 4,4'-DDT. The CTR also specifies targets for DDE and DDD, but does not specify a target for total DDTs. The lowest DDT target is selected for the purposes of representing Total DDTs in this table. If analytical results that resolve individual DDT compounds are available, all of the CTR criteria should be applied individually.

[4] For Puddingstone Reservoir, the consensus-based TEC sediment target value was used for setting alternative wasteload allocations when sufficient demonstration that the fish tissue targets are met has been made. Details on when each set of targets apply are included in the wasteload allocation sections of the Puddingstone Reservoir DDT impairment chapter.

2.2.8 Lead

The Basin Plan requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). CTR 40 CFR 131.38 establishes short-term (acute) and long-term (chronic) aquatic life criteria for metals in both freshwater and saltwater (USEPA, 2000a). Refer to Section 2.2.4 for a detailed explanation of the procedure used to calculate metal targets. Coefficients for calculating lead criteria are listed in Table 2-10.

In addition to the CTR discussion in Section 2.2.4, the chronic and acute conversion factors for lead in freshwater are dependent on hardness and, therefore, should be calculated for each waterbody evaluated. In order to assess compliance with the standards, lead and hardness should be determined at the same time. The following equations can be used to calculate the acute and chronic lead conversion factors based on site-specific hardness data:

$$\text{Lead ACF} = 1.46203 - [(\ln\{\text{hardness}\})(0.145712)] \quad \textbf{Equation 2-7}$$

$$\text{Lead CCF} = 1.46203 - [(\ln\{\text{hardness}\})(0.145712)] \quad \textbf{Equation 2-8}$$

Table 2-10. Coefficients Used in Formulas for Calculating CTR Freshwater Criteria for Lead

Metal	ACF	m_a	b_a	CCF	m_C	b_C
Lead	*	1.273	-1.460	*	1.273	-4.705

*The ACF and CCF for lead are hardness-dependent, and are therefore calculated for each lake specifically (see Table 2-11).

Chronic lead freshwater targets for each lake are calculated based on the 50th percentile of hardness values measured during lead sampling events, while the acute targets are calculated using the 90th percentile hardness (Appendix G, Monitoring Data). These are presented as example calculations since the actual target varies with the hardness value measured during sample collection. Table 2-11 summarizes the acute and chronic criterion for each lake impaired by lead (note that CTR does not include a human health criterion for lead).

Table 2-11. Hardness-Dependent Acute and Chronic Lead Targets

Lake	WER	90th Percentile Hardness (mg/L as $CaCO_3$)	ACF[4]	Acute Criterion[1] (µg/L dissolved fraction)	50th Percentile Hardness (mg/L as $CaCO_3$)	CCF[4]	Chronic Criterion[2] (µg/L dissolved fraction)
Peck Road Park Lake	1	121	0.763	79.43	84	0.816	2.08
Lincoln Park Lake	1	332	0.616	231.75	315	0.624	8.55
Echo Park Lake	1	231	0.669	158.58	208	0.684	5.53
El Dorado Park Lakes	1	124	0.760	81.56	95	0.798	2.38
Legg Lakes	1	246	0.660	169.44	182	0.704	4.80
Santa Fe Dam Park Lake	1	131	0.752	86.54	100	0.791	2.52
Westlake Lake	1	468[3]	0.589	280.85	336	0.614	9.14

Note: The median and 90th percentile hardness values were calculated from the observed data and used in the calculation of the chronic and acute targets, respectively. These are presented as example calculations since the actual target varies with the hardness value measured during sample collection.

[1] The acute criterion is a short-term average not to be exceeded more than once every three years on the average.

[2] The chronic criterion is the highest four-day average not to be exceeded more than once every three years on average.

[3] The 90th percentile hardness was greater than 400 mg/L. According to CTR, if hardness is over 400 mg/L, a hardness of 400 mg/L should be used with a default WER of 1.0. Therefore, hardness of 400 mg/L was used in the acute target calculations for Westlake Lake.

[4] Conversion factors are hardness dependent. Refer to Equation 2-7 and Equation 2-8 to calculate the ACF and CCF, respectively.

2.2.9 Mercury

Mercury targets are provided to ensure protection of both human health and wildlife, consistent with the beneficial uses associated with the mercury-impaired waterbodies. As discussed below, the human health targets are considered protective of wildlife; therefore, the values presented in Table 2-13 are used for TMDL calculations and confirmation of impairments.

Table 2-12. Mercury Targets

Lake/Reservoir	Total Mercury Maximum Contaminant Level (µg/L)	Total Mercury Human Health Criterion for Consumption of Water and Organisms (µg/L total fraction)	Total Mercury Human Health Criterion for Consumption of Organisms Only (µg/L total fraction)	Dissolved Methyl-mercury Water Quality Targets (ng/L)	Methylmercury Fish Tissue Concentration in 350 mm (average length) Largemouth Bass (ppm)
El Dorado Park Lakes	2.0	0.050	0.051	0.081	0.22
Puddingstone Reservoir	2.0	0.050	0.051	0.081	0.22
Lake Sherwood	2.0	0.050	0.051	0.081	0.22

Note: Shaded cells represent the selected targets for each waterbody.

2.2.9.1 Protection of Human Health

Fish tissue and water column targets for methylmercury and mercury are chosen based on applicable beneficial uses. For waters designated MUN, the Basin Plan lists a water column maximum contaminant level of 0.002 mg/L, or 2 μg/L. The California Toxics Rule (CTR) includes human health criteria for the consumption of water and organisms or organisms only as 0.050 μg/L and 0.051 μg/L, respectively (USEPA, 2000a). California often implements these values on a 30 day average. Because the human health criterion for the consumption of water and organisms is the most restrictive criterion, a water column target of 0.050 μg/L is the appropriate target for waterbodies with the MUN designated use (Puddingstone Reservoir). The human health criterion for the consumption of organisms only (0.051 μg/L) is appropriate for waterbodies without the MUN designated use (El Dorado Park lakes and Lake Sherwood).

The fish contaminant goal for methylmercury defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 220 ppb or 0.22 ppm. This concentration is a chronic target designed to protect human health from the cumulative effects of long-term exposure to contaminated fish. It is based on a consumption rate of 8 ounces of fish per week, prior to cooking and is more restrictive than the federal Clean Water Act (CWA) 304(a) guidance criterion for the protection of human health of 0.3 ppm (USEPA, 2001a). The assessment data available for the three mercury impaired lakes report concentrations of total mercury in fish tissue, of which most is in the form of methylmercury. Comparison of the assessment data to the methylmercury fish contaminant goal results in slightly conservative TMDL calculations and is considered part of the implicit margin of safety.

In addition, a water column target for dissolved methylmercury of 0.081 ng/L is applicable for all three mercury-impaired lakes. This value is calculated by dividing the fish contaminant goal (0.22 ppm) with a national bioaccumulation factor (for dissolved methylmercury) of 2,700,000 applicable for trophic level 4 fish (and multiplying by a factor of 10^6 to convert from milligrams to nanograms) (USEPA, 2001a, Appendix A). A bioaccumulation factor or BAF is the ratio of the concentration of a chemical in the water column to the concentration of the chemical in fish tissue and are in units of liters per kilograms (L/kg).

The applicable numeric targets for these TMDLs are the California ambient water quality criterion of 50 ng/L or 51 ng/L total mercury in the water column, the calculated dissolved methylmercury water column concentration of 0.081 ng/L, and the OEHHA fish contaminant goal of 0.22 ppm methylmercury in fish tissue. As it is primarily methylmercury that accumulates in fish, the 0.22 ppm target may be applied to the total mercury concentration in the edible portion of fish. Total mercury concentrations in edible fish from each lake exceed the contaminant goal. Fish in each lake accumulate unacceptable tissue concentrations of mercury even though the ambient water column criterion appears to be met. The most restrictive target is the fish contaminant goal of 0.22 ppm methylmercury, and is selected as the primary numeric target for calculating these TMDLs.

Mercury bioaccumulates in the food chain, which means larger fish that consume smaller fish have higher concentrations. Within a lake fish community, top predators usually have higher mercury concentrations than forage fish, and size and tissue concentrations generally increase with age. Top predator fish (such as bass) are often target species for sport fishermen. Risks to human health from the consumption of mercury-contaminated fish are based on long-term, cumulative effects, rather than concentrations in individual fish. Therefore, the target is not applied to the extreme case of the most-contaminated fish within a target species; instead, the target is applied to average concentrations in a top predator species of a size likely to be caught and consumed.

Within each of the mercury-impaired lakes, the top predator sport fish, and also the fish with the highest reported tissue methylmercury body burden, is largemouth bass (*Micropterus salmoides*). Largemouth bass continue to bioaccumulate mercury with increasing size and age. The California Department of Fish and Game requires that anglers release largemouth bass less than 12 inches (305 mm) in length and that each angler keep no more than five fish per day. The largemouth bass caught for determination of fish tissue contaminant concentrations in these three lakes ranged in size from 200 to 598 mm in length, and exceedances of the fish contaminant goal occurred in largemouth bass ranging in length from 286 to 598 mm (Appendix G, Monitoring Data).

The range of length for assessing compliance with this fish tissue target is 325-375 mm for largemouth bass. However, an average of 350 mm largemouth bass is used for TMDL calculations. This length has been identified by two separate studies as the average length of largemouth bass caught with fishing lines from California lakes (personal communication, Aroon Melwani, San Francisco Estuary Institute (SFEI), to Valentina Cabrera-Stagno, US EPA Region IX, October 22, 2009). Setting the fish tissue target to this length protects human health over the average range of fish caught. Setting the fish tissue target to the minimum length where exceedances have been detected will be less protective of human health because all fish greater than that length may exceed the criterion. Setting the fish tissue target to the maximum length may be overly protective since most fish that are caught will be less than the maximum length.

Error! Reference source not found. above summarizes the applicable targets for the three waterbodies listed for mercury addressed by this document. The shaded cells in this table represent the selected targets for each waterbody. The fish tissue concentration targets are consistent; however, the water column targets differ. Specifically, Puddingstone Reservoir has an MUN designated use; therefore, the human health criterion for the consumption of water and organisms is appropriate (0.50 µg/L), while the target for El Dorado Park lakes and Lake Sherwood is 0.051 µg/L, associated with consumption of organisms only because these lakes do not have an existing MUN designated use so the criterion consistent with the REC-1 beneficial use is selected. The dissolved methylmercury water column target of 0.081 ng/L is applicable for all three lakes.

2.2.9.2 Protection of Wildlife

Wildlife species that eat fish or other aquatic organisms containing mercury are potentially at risk from the toxic effects of mercury. This risk is a function of ecosystem dynamics and understanding the risk requires evaluation of the potential for contaminants to move through an ecosystem via trophic levels. Trophic levels describe the position an organism occupies in the food chain (i.e., what the organism eats and what eats the organism). In a simple example of an aquatic ecosystem, plants (or primary producers) are at the base of the food chain (trophic level 1), followed by primary consumers in trophic level 2 (i.e., herbivorous organisms (fish, snails, macroinvertebrates, etc.)), secondary consumers in trophic level 3 (i.e., invertebrate feeding fish, predatory macroinvertebrates, etc.), and tertiary consumers in trophic level 4 (i.e., fish-eating fish, water snakes, etc.). The top-level consumers are followed by top-level predators, such as eagles, raccoons, and other carnivorous animals. It is important to note that organisms above trophic level 1 (plants) often occupy a number of trophic levels. For example, turtles are considered trophic level 2 when they feed on vegetation, trophic level 3 when they eat herbivorous invertebrates and fish, and trophic level 4 when they feed on predatory fish. Generally, the trophic level for a carnivore is one level higher than the trophic level of the animal it eats.

To evaluate risk associated with the toxic effects of mercury, the fish tissue concentration target of 0.22 ppm methylmercury in largemouth bass (a trophic level 4 fish) of 350 mm in length was analyzed to see whether it is protective of wildlife species (Note: this is the average size largemouth bass caught by humans with fishing lines in California lakes based on a minimum catch size of 305 mm; therefore, 350 mm is considered a large fish because many smaller fish [less than 305 mm] are also part of trophic level 4). The analysis draws on previous studies conducted by US Fish and Wildlife Service (USFWS) to determine safe levels of mercury in fish tissue for wildlife in California and looks at both generic wildlife receptor categories and specific threatened and endangered species found at the mercury-impaired lakes. USFWS recommended that the analysis include the following six receptor categories: fish, small piscivorous birds, large piscivorous birds, insectivorous passerine birds, carnivorous waterfowl, and piscivorous mammals (personal communication, Katie Zeeman, USFWS Carlsbad Office, to Valentina Cabrera-Stagno, USEPA Region IX, October 1, 2009). The target was found to be protective of wildlife, as described below.

In deriving the national CWA 304(a) guidance criterion to protect human health, USEPA developed draft national bioaccumulation factors (BAFs) that describe the bioaccumulation and biomagnifications between trophic levels (USEPA, 2001a). The national BAFs are ratios (in L/kg) which relate the concentration of dissolved methylmercury in the water column to its expected concentration in commonly consumed aquatic organisms in a specified trophic level. In addition, food chain multipliers can be calculated from the national BAFs. Food chain multipliers are the ratio of the BAF for one trophic level to the BAF for the trophic level directly below (for example, the food chain multiplier from trophic level 3 to 4 is the BAF for trophic level 4 divided by the BAF for trophic level 3 (2,700,000/680,000 = 4)). The BAFs and calculated food chain multipliers are shown Table 2-13. Using the food chain multipliers, one can calculate trophic level 3 and 2 concentrations from a trophic level 4 target. The methylmercury concentrations calculated for trophic levels 2 and 3 based on the trophic level 4 target in these TMDLs (0.22 ppm methylmercury) are shown in Table 2-13 (i.e., trophic level 3 concentration is the trophic level 4 target divided by the food chain multiplier from trophic level 3 to 4 (0.22 ppm/4 = 0.055 ppm)). The target in trophic level 4 is set for a large sized fish and is lower for the trophic level as a whole. Using this number to estimate trophic level 3 and 2 concentrations is highly conservative and leads to overestimates of the trophic level 3 and 2 concentrations.

Table 2-13. National Bioaccumulation Factors (BAFs) and Food Chain Multipliers

Bioaccumulation Factors and Food Chain Multipliers	Value
Draft National BAF for Trophic Level 4	2,700,000 L/kg
Draft National BAF for Trophic Level 3	680,000 L/kg
Draft National BAF for Trophic Level 2	120,000 L/kg
Food chain multiplier from trophic level 3 to 4 biota	4
Food chain multiplier from trophic level 2 to 3 biota	5.7

Table 2-14. Trophic Level Concentrations

Trophic Level	Methylmercury Fish Tissue Concentration (ppm wet weight)
Trophic Level 4 target concentration*	0.22
Calculated corresponding trophic level 3 concentration	0.055
Calculated corresponding trophic level 2 concentration	0.0096

*Note: The TMDL target is actually set for a large sized fish (350 mm) not for the trophic level as a whole. The trophic level concentration as a whole is lower and consequently the trophic level 3 and 2 levels will be lower than the values presented above.

2.2.9.2.1 Generic Wildlife Receptor Category Analysis

2.2.9.2.1.1 Fish

When USFWS evaluated the USEPA national CWA 304(a) human health 0.3 ppm methylmercury criterion, it found that threatened and endangered fish species in California were not likely to be adversely affected (USFWS, 2003). Since the USEPA criterion is higher than the selected target (0.22 ppm methylmercury fish tissue guideline (OEHHA, 2008)), these TMDLs are protective of threatened and endangered freshwater fish species, and thus, in general protective of any freshwater fish species, that may be living in the mercury-impaired lakes.

2.2.9.2.1.2 Small Piscivorous Birds

The Belted Kingfisher is a small piscivorous bird that has been previously evaluated by USFWS for a safe level of mercury. In the analysis of the numeric wildlife targets for the Guadalupe River Watershed TMDL, USFWS found that concentrations of 0.05 ppm methylmercury in 50-150 mm trophic level 3 fish would be protective of the Belted Kingfisher (USFWS, 2005). The fish tissue target in these TMDLs is expected to be as protective as those found necessary in the Guadalupe River Watershed TMDL analysis, for fish in the same size range and trophic level.

2.2.9.2.1.3 Large Piscivorous Birds

The Bald Eagle is a large piscivorous bird that has been sighted (albeit rarely) at these mercury-impaired lakes. When USFWS evaluated the USEPA national CWA 304(a) human health 0.3 ppm methylmercury criterion, it found that a target of 0.3 ppm methylmercury in trophic level 4 fish would be protective of bald eagles (USFWS, 2003). The target for these TMDLs (0.22 ppm methylmercury fish contaminant goal (OEHHA, 2008)) is lower than the CWA 304(a) human health criterion and is therefore considered protective of large piscivorous birds.

2.2.9.2.1.4 Insectivorous Passerine Birds

No studies on fish tissue mercury concentration impacts to insectivorous passerine bird species were readily available, so this endpoint was not assessed. The level of mercury anticipated to be in trophic level two species is very low (0.0096 ppm wet weight; Table 2-13.) and it is not expected to be a concern for insect-eating birds.

2.2.9.2.1.5 Carnivorous Waterfowl

The Common Merganser is a carnivorous waterfowl that has been evaluated in previous USFWS studies for a safe level of mercury. In the evaluation of numeric wildlife targets for the Guadalupe River Watershed TMDL, USFWS found that concentrations of 0.1 ppm methylmercury in 150-350 mm trophic level 3 fish would be protective of the Common Merganser (USFWS, 2005). The level anticipated in these TMDLs for trophic level 3 fish (0.055 ppm; Table 2-13.) is about half of that number and is therefore protective of the Common Merganser and other carnivorous waterfowl.

2.2.9.2.1.6 Piscivorous Mammals

Mink is a piscivorous mammal species that has been evaluated previously. USFWS previously evaluated mink. In its analysis of numeric wildlife targets for the Cache Creek and Sacramento-San Joaquin Delta Watersheds TMDL, USFWS found that concentrations of 0.077 ppm methylmercury in trophic level 3 fish smaller than 150 mm would be protective of mink (USFWS, 2004). The methylmercury level anticipated in these TMDLs for trophic level 3 fish (0.055 ppm; Table 2-13.) is well below that number and is therefore protective of piscivorous mammals.

2.2.9.2.2 Specific Threatened and Endangered Species Analysis

Threatened and endangered species are considered separately for Lake Sherwood, Puddingstone Reservoir, and El Dorado Park lakes. Species lists were requested from USFWS for each of the mercury-impaired lakes. Audubon Society bird lists and the California Department of Fish and Game's California Natural Diversity Database were also consulted.

2.2.9.2.2.1 Lake Sherwood

The USFWS Ventura Office indicated that the only federally listed or candidate species that may occur in proximity to Lake Sherwood is the endangered plant *Pentachaeta lyonii* (Lyon's pentachaeta) (Dellith, 2009). Additionally, a bird list provided by lake resident Mary Hansen did not include any federally listed or candidate species (personal communication, Mary Hansen to Valentina Cabrera-Stagno, USEPA Region IX, September 7, 2010). Plants will not be impacted by this fish tissue target.

2.2.9.2.2.2 Puddingstone Reservoir

The USFWS Carlsbad Office indicated that the federally threatened fish species Santa Ana sucker (*Catostomus santaanae*) may exist in San Dimas Creek and feed in Puddingstone Reservoir. As explained in the generic wildlife receptor category analysis above (Section 2.2.9.2.1.1), fish species are not anticipated to be adversely affected by the proposed mercury target. In addition, the federally threatened coastal California Gnatcatcher (*Polioptila californica californica*) occupies habitat surrounding the reservoir and feeds on insects that could be affected by water quality (personal communication, Christine Medak, USFWS Carlsbad Office, to Valentina Cabrera-Stagno, USEPA Region IX, November 24, 2009). The coastal California Gnatcatcher has not been specifically analyzed. Of the species that USFWS has analyzed previously, its life history is most similar to California Clapper Rail another invertivore. When USFWS evaluated the USEPA CWA 304(a) human health 0.3 ppm methylmercury criterion, it found that a target of 0.3 ppm methylmercury in trophic level 4 fish would be protective of California Clapper Rail (USFWS, 2003). The target for these TMDLs (0.22 ppm methylmercury fish tissue guideline (OEHHA, 2008)) is lower than the CWA 304(a) criterion and is therefore considered to be protective of California Clapper Rail and likely of the coastal California Gnatcatcher.

2.2.9.2.2.3 El Dorado Park Lakes

The USFWS Carlsbad Office did not respond to a request for species of concern at El Dorado Park lakes. The California Department of Fish and Game's California Natural Diversity Database (accessed on August 21, 2009) indicated the California Least Tern (*Sterna antillarum Browni*) may be the only rare or endangered avian species living in the area of the lakes. The Least Tern is also identified on the El Dorado Audubon Society's bird list as occasionally present in the summer (El Dorado Audubon Society, 2003). Fortunately, the California Least Tern was evaluated by USFWS in their 2003 evaluation of the USEPA CWA 304(a) human health 0.3 ppm methylmercury criterion. USFWS found that safe dietary levels for California Least Tern would be 0.005 ppm methylmercury wet weight for trophic level 2 fish, 0.03 ppm for trophic level 3 fish, and 0.12 ppm for trophic level 4 fish (USFWS, 2003). At first glance the trophic level 4 dietary value for California Least Tern looks lower than the chosen target of 0.22 ppm; however, terns are small birds that feed on small fish. The NatureServe Explorer online encyclopedia (accessed on November 24, 2009) indicates that this bird is both insectivorous and piscivorous and feeds on small fish generally less than 9 cm in length such as anchovy, topsmelt, surf-perch, killifish, and mosquitofish (NatureServe, 2009). No data exist for current concentrations of mercury in trophic level 4 fish in such a small size range (less than 90 mm) because the minimum fish size for the 2007 lakes survey was 200 mm. However, analyses have shown that fish size and mercury concentration generally have a linear relationship (Appendix C, Mercury TMDL Development), so smaller size fish will have lower mercury concentrations. Table 2-15 lists the concentration of mercury in all fish tissue samples less 250 mm in length at El Dorado Park lakes. Only total mercury was analyzed so the corresponding methylmercury concentrations will be slightly lower.

Table 2-15. El Dorado Park Lakes Fish Tissue Concentrations for Fish <250 mm in Length

Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
206	0.15
219	0.13

As indicated in this table, existing concentrations for fish more than twice the size of the 90 mm California Least Tern's maximum prey size are close to the 0.12 ppm methylmercury safe level indentified by USFWS. Fish that are 90 mm in length or shorter are likely already meeting this target at El Dorado Park lakes. Additionally, the target for 350 mm trophic level 4 fish in these TMDLs will reduce mercury levels in all size classes. This will lead to even lower concentrations in these small size class fish. USFWS found that safe dietary levels for California Least Tern would be 0.005 ppm methylmercury wet weight for trophic level 2 fish and 0.03 ppm for trophic level 3 fish (USFWS, 2003). As described above, given that the trophic level 4 fish target is likely already being met at El Dorado Park lakes, it is likely that trophic levels 2 and 3 fish targets for tern are also being met in the small size class that California Least Tern prey upon.

2.2.10 PCBs

Polychlorinated biphenyls (PCBs) consist of a family of many related congeners. The individual congeners are often referred to by their "BZ" number. Environmental analyses may address individual congeners, homologs (groups of congeners with the same number of chlorine atoms), equivalent concentrations of the commercial mixtures of PCBs known as Aroclors, or total PCBs. The environmental measurements and targets described in this document are in terms of total PCBs, defined as the "sum of all congener or isomer or homolog or aroclor analyses" (CTR, 40 CFR 131.38(b)(1) footnote v).

Selections of applicable OC Pesticides and PCBs targets are described above in Section 2.2.2.1 through Section 2.2.2.3. Water column targets for PCBs are based on beneficial use (Section 2.2.2.1). For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0005 mg/L, or 500 ng/L. The Plan also requires that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). This objective is addressed through the CTR water quality criteria.

A chronic criterion for the sum of PCB compounds in freshwater systems is included in the CTR as 0.014 µg/L (14 ng/L; USEPA, 2000a). The CTR also provides a human health criterion for the consumption of both water and organisms and organisms only of 0.00017 µg/L (0.17 ng/L). California often implements these values on a 30 day average. The human health criterion is the most restrictive of the criterion specified for water column concentrations and was selected as the target concentration for Echo Park Lake, Peck Road Park Lake, and Puddingstone Reservoir. CTR criteria are considered protective of aquatic life.

Two target sediment concentrations for total PCBs have been identified (Section 2.2.2.2). There are no Basin Plan Objectives for toxicity levels in sediment; however sediment quality guidelines are reported by multiple agencies for the protection of sediment biota. MacDonald et al. (2000) compiled and evaluated the guidelines and derived consensus-based sediment quality guidelines that incorporate multiple recommendations. The consensus-based TEC for total PCBs is 59.8 µg/kg dry weight, defined by CBSQG (MacDonald el al., 2000). The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. An additional sediment target based on bioaccumulation in fish was also calculated for each impaired lake to ensure that the FCG is met using the BSAF approach described in Section 2.2.2.2.2. The lower of the two sediment target values is applied in each lake. Additionally, these TMDLs include alternative wasteload allocations to be applied when a sufficient demonstration has been made that the fish tissue targets are met. These targets are based on the consensus-based TEC values. Details on when each set of targets apply are included in the wasteload allocation section of each relevant lake chapter.

Fish tissue targets are described above in Section 2.2.2.3. The fish contaminant goal for PCBs defined by the OEHHA (2008) is 3.6 ppb based on cancer risk (the FCG based on non-cancer risk is 63 ppb). Table 2-16 summarizes the applicable targets for the three waterbodies listed for total PCBs addressed by this document.

Table 2-16. Total PCB Targets

Lake	Maximum Contaminant Level (ng/L)	Chronic Criterion[1] (ng/L)	Human Health Criterion[2] (ng/L)	Consensus-based TEC Sediment Target (µg/kg)[3]	BSAF-derived Sediment Target (µg/kg)	Fish Contaminant Goal (ppb)
Echo Park Lake	500	14	0.17	59.8	1.77	3.6
Peck Road Park Lake	500	14	0.17	59.8	1.29	3.6
Puddingstone Reservoir	500	14	0.17	59.8	0.59	3.6

Note: Shaded cells represent the selected targets for each waterbody.

[1]The chronic criterion is the highest four day average not to be exceeded more than once every three years on the average.

[2]The human health criterion applies to both consumption of water and organisms and organisms only.

[3]The consensus-based TEC sediment target value was used for setting alternative wasteload allocations when sufficient demonstration that the fish tissue targets are met has been made. Details on when each set of targets apply are included in the wasteload allocation sections of each relevant lake chapter.

2.2.11 pH

As specified in the Basin Plan, lake waters must not be depressed below 6.5 or raised above 8.5 as a result of waste discharges or be changed by more than 0.5 units from the natural conditions as a result of waste discharges. These serve as the numeric targets for pH in these TMDLs.

Lakes listed as impaired by pH include Echo Park Lake, Lake Calabasas, El Dorado Park lakes, Legg Lake, and Santa Fe Dam Park Lake. Target depths for each lake were set by the Regional Board and USEPA based on site specific conditions. Shallow, well mixed lakes must meet the target in the water column from the surface to 0.3 meters above the bottom of the lake. Deeper lakes that thermally stratify during the summer months, such as Peck Road Park Lake and Puddingstone Reservoir, must meet the pH target throughout the epilimnion of the water column.

The epilimnion is the upper stratum of more or less uniformly warm, circulating, and fairly turbulent water during summer stratification. The epilimnion floats above a cold relatively undisturbed region called the hypolimnion. The stratum between the two is the metalimnion and is characterized by a thermocline, which refers to the plane of maximum rate of decrease of temperature with respect to depth.

For the purposes of these TMDLs, the presence of stratification will be defined by whether there is a change in lake temperature greater than 1 degree Celsius per meter. Deep lakes must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the pH target must be met in the epilimnion, the portion of the water column above the thermocline.

2.2.12 Trash

The target for trash is "zero trash." Lakes listed as impaired by trash include Echo Park Lake, Peck Road Park Lake, Lincoln Park Lake, and Legg Lake. Legg Lake has an existing TMDL for trash, the remaining three lakes are addressed in this document.

2.3 Basis for Listing

The Los Angeles Regional Board provided the basis for listing each of the 10 lakes addressed in this document on the State's 303(d) list in its *Water Quality Assessment & Documentation Report* (LARWQCB, 1996). Waterbody-pollutant combinations found to be either not supporting or partially supporting a beneficial use were identified as impairments on the 303(d) list. Impairments in the *Water Quality Assessment & Documentation Report* (LARWQCB, 1996) are described relative to the USEPA 305(b) beneficial uses, which are broad federal beneficial use categories described under the federal guidance for 305(b) reporting. For consistency with the state of California beneficial use categories, the California beneficial uses for the waterbodies addressed in this document are related to federal beneficial uses as shown in Table 2-17. The California use "NAV" was not assessed in the report (LARWQCB, 1996). It should be noted that the water quality standards or assessment methodology used in the 1996 assessment report are often not the same as current standards used to confirm impairments and calculate TMDLs in this report. Current standards and targets selected in these TMDLs are summarized in Section 2.2 and included in specific lake chapters. Regional Board currently follows California's Impaired Waters Guidance (SWRCB, 2005) in making 303(d) listing and delisting decisions (SWRCB, 2005). One of the major differences between the assessment methodology employed in developing the 1996 *Water Quality Assessment & Documentation Report* and current practice is that the partially supporting category no longer exists.

Table 2-17. Linkage Between California and Federal Beneficial Uses

Federal Beneficial Use	California Beneficial Use Code
Aquatic Life	WARM, WILD, WET, COLD, RARE
Primary Contact Recreation	REC1
Secondary Contact Recreation	REC2
Drinking Water Supply	MUN, GWR (where appropriate)
Agriculture	AGR, GWR (where appropriate)
Fish Consumption	REC1

This section summarizes the listing information by impairment. In some cases, more recent data may have resulted in additional impairments included on the 2008-2010 303(d) list (SWRCB, 2010) or identification of new impairments not currently on the 303(d) list. Data collected after the original listing are not included in this section, but are discussed in lake-specific sections of the report and are included in the summary in Table 2-31.

2.3.1 Algae

According to the *Water Quality Assessment & Documentation Report*, a waterbody was listed as impaired by algae if field observations indicated excessive growth impacting the primary or secondary contact recreation use (LARWQCB, 1996). Visual observations of algae were classified either as “none” or “significant amount observed.” Waterbodies were considered “not supporting” these uses if field observations indicated impairment in more than 25 percent of observations. Waterbodies were considered “partially supporting” if field observations indicated impairment in 11 to 25 percent of observations. “Fully supporting” waterbodies had indications of impairment in less than 11 percent of observations. Lake assessments were completed during the University of California, Riverside urban lakes study (UC Riverside, 1994).

Two of the lakes addressed by this document were listed for impairment due to algae (Table 2-18). Both are listed as “not supporting” the primary and secondary contact recreation uses.

Table 2-18. Listing Information for Lakes Impaired by Algae

Lake	Use: Support Status
Echo Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting
El Dorado Park Lakes	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting

2.3.2 Ammonia

Ammonia impairments in these lakes were based on the support status for aquatic life use, primary recreation, and secondary recreation (LARWQCB, 1996). Lakes classified as “not supporting” the aquatic life use were found to exceed the temperature/pH-based ammonia criteria in more than 10 percent of samples. Those classified as “partially supporting” exceeded criteria more than twice within a 6-year period, but in fewer than 10 percent of samples. A status of “fully supporting” resulted from no more than two violations of chronic criteria (acute criteria if no chronic criteria were available) within a 6-year period based on at least 20 grab or 1-day composite samples; if fewer than 20 samples were available, then best professional judgment was used considering the number of pollutants having violations and the magnitudes of the exceedance(s).

Lakes classified as not supporting the primary or secondary contact recreation use due to ammonia exceeded the taste and odor criterion of 0.037 mg/L in more than 25 percent of measurements. Partially supporting lakes exceeded the criterion in 11 to 25 percent of samples, and fully supporting lakes exceeded the criterion in less than 11 percent of samples.

Table 2-19 summarizes the federal beneficial uses and support status of the lakes impaired by ammonia. Summary statistics reported in the assessment report (LARWQCB, 1996) are also included. A value of “ND” indicates the sample concentration was non detect. The symbol “#” denotes that no standard deviation has been calculated because there was not a normal distribution or because there were less than three samples.

Table 2-19. Listing Information for Lakes Impaired by Ammonia

Lake	Use: Support Status	Number of Samples, Range (mg/L), Average ± Standard Deviation (mg/L)
Lincoln Park Lake	Aquatic Life: Not Supporting Primary Contact Recreation: Not Supporting	28, ND - 1.14, 0.34 ± 0.32
Echo Park Lake	Aquatic Life: Not Supporting Primary Contact Recreation: Not Supporting	31, ND - 0.71, 0.11#
Lake Calabasas	Aquatic Life: Not Supporting Primary Contact Recreation: Not Supporting	28, ND - 0.45, 0.06#
El Dorado Park Lakes	Aquatic Life: Not Supporting Primary Contact Recreation: Not Supporting	45, ND - 1.92, 0.30#
Legg Lakes	Aquatic Life: Partially Supporting	43, ND - 0.35, 0.05#

2.3.3 Chlordane

Chlordane impairments were assessed for both the aquatic life use and the fish consumption use against the Maximum Tissue Residue Level (MTRL) of 1.1 ppb (LARWQCB, 1996). MTRLs were established for fish filet samples by multiplying the human health water quality criteria in the CTR and the bioconcentration factor (BCF) for each substance. Waters with a support status of “not supporting” the fish consumption use were supposedly under a “no consumption” ban for fish and shellfish. Each water was also listed as “not supporting” the aquatic life use, indicating impairment of at least one assemblage of the biological community.

Fish tissue monitoring was conducted as part of the Toxic Substances Monitoring Program (TSMP). Summary data in the assessment report included the sample type, the year of sample collection, and the criterion exceeded by the sample (Table 2-20). Chlordane fish tissue samples were comprised of seven-fish composites for Peck Road Park Lake and six-fish composites for Puddingstone Reservoir. Samples from Peck Road Park Lake exceeded the MTRL in 1991 (14.1 ppb); samples from Puddingstone exceeded the MTRL in both 1991 (16.1 ppb) and 1992 (31.7 ppb).

Table 2-20. Listing Information for Lakes Impaired by Chlordane

Lake/Reservoir	Use: Support Status	Sample Type (Year): Impairment (Criterion)
Peck Road Park Lake	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): chlordane (MTRLs) Tissue ('92): No organic chemicals at elevated levels
Puddingstone Reservoir	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): chlordane (MTRLs) Tissue ('92): chlordane (MTRLs)

2.3.4 Copper

Copper impairments were assessed in relation to the aquatic life use. The criterion was based on a four-day average total recoverable copper concentration calculated from the following equation, which was based on USEPA National Ambient Water Quality Criteria published in 1986:

$$TotalCopper(\mu g / L) = \exp^{\{0.8545[\ln(hardness)]-1.465\}} \qquad \textbf{Equation 2-9}$$

Four lakes addressed by this document were classified as “not supporting” the aquatic life use, indicating the criterion was exceeded in more than 10 percent of samples. The summary table provided in the *Water*

Quality Assessment & Documentation Report lists the maximum total recoverable copper concentration observed at each lake; corresponding hardness values were not provided (Table 2-21) (LARWQCB, 1996).

Table 2-21. Listing Information for Lakes Impaired by Copper

Lake	Use: Support Status	Maximum Concentration of Total Recoverable Copper (µg/L)
Echo Park Lake	Aquatic Life: Not Supporting	105
El Dorado Park Lakes	Aquatic Life: Not Supporting	99
Legg Lakes	Aquatic Life: Not Supporting	97
Santa Fe Dam Park Lake	Aquatic Life: Not Supporting	56

2.3.5 Dieldrin

Dieldrin impairments were not identified in the assessment report (LARWQCB, 1996), but were subsequently observed after sample collection and analyses. These impairments and analyses are discussed in greater detail in the Peck Road Park Lake, Echo Park Lake, and Puddingstone Reservoir sections.

2.3.6 Dissolved Oxygen

Dissolved oxygen impairments were assessed relative to the aquatic life use. A support status of “not supporting” was assigned to waterbodies where more than 25 percent of measurements exceeded the criteria; “partially supporting” waterbodies had exceedances observed in 11 to 25 percent of measurements.

Table 2-22 summarizes the beneficial uses and support status of the lakes impaired by dissolved oxygen. Summary statistics reported in the assessment report (LARWQCB, 1996) are also included.

Table 2-22. Listing Information for Lakes Impaired by Low Dissolved Oxygen

Lake/Reservoir	Use: Support Status	Number of Samples, Range (mg/L), Average ± Standard Deviation (mg/L)
Peck Road Park Lake	Aquatic Life: Not Supporting	195, 0.2 – 15.2, 6.0 ± 4.0
Lincoln Park Lake	Aquatic Life: Partially Supporting	78, 0.1 - 13.7, 6.9 ± 3.3
Lake Calabasas	Aquatic Life: Partially Supporting	92, 0.2-15.7, 8.7 ± 3.3
Puddingstone Reservoir	Aquatic Life: Not Supporting	187, 0.1-14.9, 4.3 ± 3.5

2.3.7 DDT

DDT impairments were assessed for both the aquatic life use and the fish consumption use against the MTRL for DDT (32 ppb) (LARWQCB, 1996). Waters with a support status of “not supporting” the fish consumption use were supposedly under a “no consumption” ban for fish and shellfish. Each water was also listed as "not supporting" the aquatic life use, indicating impairment of at least one biological community assemblage.

Fish tissue monitoring was conducted as part of the TSMP. Summary data in the assessment report included the sample type, the year of sample collection, and the criterion exceeded by the sample (Table 2-23). The DDT seven-fish composite tissue sample from Peck Road Park Lake exceeded the MTRL in 1991 with a concentration of 39 ppb; the six-fish composite sample from Puddingstone exceeded the MTRL in 1992 (36 ppb).

Table 2-23. Listing Information for Lakes Impaired by DDT

Lake/Reservoir	Use: Support Status	Sample Type (Year): Impairment (Criterion)
Peck Road Park Lake	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): DDT (MTRLs) Tissue ('92): No organic chemicals at elevated levels
Puddingstone Reservoir	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): DDT not at elevated levels Tissue ('92): DDT (MTRLs)

2.3.8 Eutrophication

The eutrophication impairment was based on an assessment of the aquatic life use. An assessment of "fully supporting" indicated functioning, sustainable biological communities (e.g., macroinvertebrates, fish, or algae) none of which had been modified significantly beyond the natural range of the reference condition. "Partially supporting" waterbodies had at least one assemblage that indicated less than full support with slight to moderate modification of the biological community noted. Waterbodies listed as "not supporting" had at least one assemblage indicating nonsupport with data clearly indicating severe modification of the biological community (LARWQCB, 1996).

Further information regarding the eutrophication impairment was not specified in the *Water Quality Assessment & Documentation Report*. Four lakes addressed by this document were considered impaired by eutrophication (Table 2-24).

Table 2-24. Listing Information for Lakes Impaired by Eutrophication

Lake	Use: Support Status
Lincoln Park Lake	Aquatic Life: Not Supporting
Echo Park Lake	Aquatic Life: Not Supporting
Lake Calabasas	Aquatic Life: Not Supporting
El Dorado Park Lakes	Aquatic Life: Not Supporting

2.3.9 Lead

Lead impairments were assessed in relation to the aquatic life use. The criterion was based on a four-day average total recoverable lead concentration calculated from the following equation, which was based on USEPA National Ambient Water Quality Criteria published in 1986:

$$TotalLead\,(\mu g/L) = \exp^{\{1.273[\ln(hardness)]-4.705\}} \qquad \textbf{Equation 2-10}$$

Seven lakes addressed by this document were classified as "not supporting" the aquatic life use, indicating the criterion was exceeded in more than 10 percent of samples. The summary table provided in the *Water Quality Assessment & Documentation Report*, lists the maximum total recoverable lead concentration observed at each lake; corresponding hardness values were not provided (Table 2-25) (LARWQCB, 1996).

Table 2-25. Listing Information for Lakes Impaired by Lead

Lake	Use: Support Status	Maximum Concentration of Total Recoverable Lead (µg/L)
Peck Road Park Lake	Aquatic Life: Not Supporting	73
Lincoln Park Lake	Aquatic Life: Not Supporting	94
Echo Park Lake	Aquatic Life: Not Supporting	105
El Dorado Park Lakes	Aquatic Life: Not Supporting	108
Legg Lakes	Aquatic Life: Not Supporting	70
Santa Fe Dam Park Lake	Aquatic Life: Not Supporting	51
Westlake Lake	Aquatic Life: Not Supporting	91

2.3.10 Mercury

Mercury impairments were assessed for the aquatic life use and fish consumption use. Three waterbodies were listed as "not supporting" the aquatic life use due to mercury impairment, indicating the criterion was exceeded in more than 10 percent of samples. Summary data for water column measurements were not provided in the assessment report.

Three criteria were used to assess the fish consumption use. The *Water Quality Assessment & Documentation Report* lists a Food and Drug Administration (FDA) action level for freshwater and marine fish of 1,000 ppb (1 ppm), a MTRL for inland surface waters of 1,000 ppb (1 ppm), and a range of Median International Standards (MIS) for freshwater fish and marine shellfish of 100 to 1,000 ppb (0.1 to 1 ppm) (LARWQCB, 1996). Three of the waterbodies addressed by this document were found "not supporting" the fish consumption use, indicating that a "no consumption" ban for fish or shellfish is in effect for the general population, or a subpopulation that could be at potentially greater risk, for one or more fish or shellfish species; or a commercial fishing or shellfishing ban is in effect.

Waterbodies designated MUN were also assessed for drinking water use against a criterion of 2 µg/L of total mercury. Each waterbody was found "fully supporting" this use, indicating that the median value of total mercury concentrations was less than the criterion.

Table 2-26 summarizes the listing information for the lakes addressed by this document that are impaired by mercury.

Table 2-26. Listing Information for Lakes Impaired by Mercury

Lake/Reservoir	Use: Support Status	Sample Type (Year): Impairment (Criterion)
El Dorado Park Lake	Aquatic Life: Not Supporting	NA
Puddingstone Reservoir	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): mercury (MIS)
Lake Sherwood	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): mercury (MIS) Tissue ('92): mercury (MTRLs,FDA)

NA: Information not included for this waterbody.

2.3.11 Odor

The *Water Quality Assessment & Documentation Report* (LARWQCB, 1996) says that the odor impairments were based on observations recorded during the University of California, Riverside urban lakes study (UC Riverside, 1994). Waterbodies listed as "not supporting" either recreational beneficial use noted the "presence" of odor in more than 25 percent of observations.

Table 2-27 summarizes the support status for the lakes addressed by this document that are listed as impaired by odor. The University of California, Riverside urban lakes study (UC Riverside, 1994) described odors at each of these lakes as either fishy or related to ducks.

Table 2-27. Listing Information for Lakes Impaired by Odor

Lake	Use: Support Status	Odor Description (UC Riverside, 1994)
Peck Road Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting	Fishy
Lincoln Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting	Ducks
Echo Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting	Duck feces
Lake Calabasas	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting	Ducks
Legg Lakes	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting	Ducks

2.3.12 PCBs

PCB impairments were assessed for both the aquatic life use and the fish consumption use against the MTRL of 2.2 ppb (LARWQCB, 1996). Waters with a support status of "not supporting" the fish consumption use were supposedly under a "no consumption" ban for fish and shellfish. Each water was also listed as "not supporting" the aquatic life use, indicating impairment of at least one biological community assemblage.

Fish tissue monitoring was conducted as part of the TSMP. Summary data in the assessment report included the sample type, the year of sample collection, and the criterion exceeded by the sample (Table 2-28). PCB fish tissue composite samples were comprised of three fish at each of the waterbodies impaired by PCBs addressed by this document. Samples collected at Puddingstone Reservoir exceeded the MTRL in both 1991 and 1992. Samples collected at Echo Park Lake exceeded the MTRLs in 1987 and 1992. The 1991 composite sample from Echo Park Lake did not have detectable levels of PCBs.

Table 2-28. Listing Information for Lakes Impaired by PCBs

Lake/Reservoir	Use: Support Status	Sample Type (Year): Impairment (Criterion)
Echo Park Lake	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): No PCBs detected Tissue ('92): PCBs (MTRLs)
Puddingstone Reservoir	Aquatic Life: Not Supporting Fish Consumption: Not Supporting	Tissue ('91): PCBs (MTRLs) Tissue ('92): PCBs (MTRLs)

2.3.13 pH

In the 1996 *Water Quality Assessment & Documentation Report*, the criterion for assessing the aquatic life use with respect to pH was a range of 6.5 to 9.0 (LARWQCB, 1996). Five waterbodies addressed by this document were listed as "partially supporting" the aquatic life use, indicating that pH measurements were out of the allowable range in 11 to 25 percent of measurements. This report also presented a criterion for assessing the primary contact recreation use based on secondary MCLs for drinking water (ranging from pH of 6.5 to 8.5). Three of the five waterbodies were listed as "not supporting" this use, indicating that more than 25 percent of measurements were outside the allowable range. Three waterbodies were also listed as "not supporting" the drinking water use based on secondary MCL criteria. Table 2-29 summarizes the listing information for the five lakes addressed by this document that were impaired by pH.

Table 2-29. Listing Information for Lakes Impaired by pH

Lake	Use: Support Status	Number of Samples, Range (mg/L), Average ± Standard Deviation (mg/L)
Echo Park Lake	Aquatic Life: Partially Supporting Primary Contact Recreation: Not Supporting	69, 7.0-9.4, 8.5 ± 0.5
Lake Calabasas	Aquatic Life: Partially Supporting Drinking Water: Not Supporting	85, 7.4-9.3, 8.6 ± 0.4
El Dorado Park Lakes	Aquatic Life: Partially Supporting Primary Contact Recreation: Not Supporting	116, 6.9-9.4, 8.5 ± 0.6
Legg Lakes	Aquatic Life: Partially Supporting Drinking Water: Not Supporting	84, 7.6-8.9, 8.3 ± 0.3
Santa Fe Dam Park Lake	Aquatic Life: Partially Supporting Primary Contact Recreation: Not Supporting Drinking Water: Not Supporting	95, 7.5-9.6, 8.7 ± 0.3

2.3.14 Trash

Trash impairments were assessed for the primary and secondary contact recreation uses. Four lakes addressed by this document were listed as "not supporting" both recreation uses (Table 2-30), indicating that the presence of trash was observed during at least 25 percent of field observations (LARWQCB, 1996). The Regional Board has adopted a TMDL for trash for Legg Lake (LARWQCB, 2007).

Table 2-30. Listing Information for Lakes Impaired by Trash

Lake	Use: Support Status
Peck Road Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting
Lincoln Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting
Echo Park Lake	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting
Legg Lakes	Primary Contact Recreation: Not Supporting Secondary Contact Recreation: Not Supporting

2.4 SUMMARY OF IMPAIRMENTS

This TMDL document addresses impairments for 10 lakes in the Los Angeles Region. Table 2-31 identifies the waterbody-pollutant combinations addressed by this document. Table 2-31 also identifies for each lake: the impairments governed by the consent decree entered in Heal the Bay Inc. v. Browner; impairments addressed by a previous TMDL; and impairments listed in a prior 303(d) list but not listed on the current 303(d) list. Table 2-31 also identifies five impairments (Peck Road Park Lake, for dieldrin and PCBs; Echo Park Lake, for chlordane and dieldrin; and Puddingstone Reservoir for dieldrin) which are not on the current 303(d) list but which, after consideration of more recent data, USEPA has determined to address by this TMDL document. Further, Table 2-31 identifies 15 listings on the current 303(d) list which, after consideration of more recent data, USEPA believes no longer meet the Federal requirements for listing; USEPA is recommending that those listings be omitted from the next 303(d) list.

Table 2-31. Waterbody-pollutant Combinations for Ten Los Angeles Region Lakes

Lake/ Reservoir	Algae	Ammonia	Chlordane	Copper	DDT	Dieldrin	Eutrophication	Lead	Organic Enrichment / Low Dissolved Oxygen	Mercury	Odor	PCBs	pH	Trash
Peck Road Park Lake			●		●	○	●/	●/	◑		●	○		●
Lincoln Park Lake		●					●	●/	◑		●			●
Echo Park Lake	●	●/	○	●/		○	●	●/			◑	●	●/	●
Lake Calabasas		●			●/		●		◑		●		●	
El Dorado Park Lakes	●	●		●\			●	●/		●			●	
Legg Lakes		●/		●/				●/			●		●	◑\
Puddingstone Reservoir			●		●	○			●	●		●		
Santa Fe Dam Park Lake				●/				●/					●/	
Lake Sherwood	●\	●\					●\		●\	●				
Westlake Lake	●\	●\		●/			●\	●/	●\					

● Impairment included in the consent decree.

◑ Impairment listed since the consent decree and included in the 2008-2010 303(d) list.

○ Impairment identified by new data analyses (after the 2008-2010 303(d) list data cutoff).

/ Impairment is no longer identified as impaired and not included on the 303(d) list.

\ Impairment is addressed by another TMDL.

/ (dashed) No longer showing impairment in recent data analyses (see lake-specific chapters); USEPA recommends these impairments not be included in California's next 303(d) list.

3 Summary of Approach

The United States Environmental Protection Agency (USEPA) Region IX is establishing Total Maximum Daily Loads (TMDLs) for impairments in nine lakes in the Los Angeles Region. USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). These lakes are currently on the State's 303(d) list for nutrient related impairments, mercury, OC Pesticides and PCBs, and trash and TMDLs have been developed to address these impairments.

This section of the TMDL report describes the general approach that was used to develop the TMDLs for each impairment. Lake specific information is contained in the individual sections devoted to each impaired lake.

3.1 General Source Assessment

This section identifies the potential sources of pollutants that discharge into the impaired lakes. In general, pollutants can enter surface waters from both point and nonpoint sources. Point sources include discharges from a discrete human-engineered outfall. These discharges are regulated through National Pollutant Discharge Elimination System (NPDES) permits. Nonpoint sources, by definition, include pollutants that reach surface waters from a number of diffuse land uses and activities that are not regulated through NPDES permits. Specific sources for each lake are described in the lake chapters, while pollutant-specific sources are discussed in the appendices; the discussion below presents general information for point and nonpoint sources.

3.1.1 Point Sources

The NPDES permits in the watersheds draining to impaired lakes include municipal separate storm sewer system (MS4) permits, a California Department of Transportation (Caltrans) stormwater permit, general construction stormwater permits, general industrial stormwater permits, and a general NPDES permit (Table 3-1). Point sources associated with each lake are presented in the lake-specific chapters.

Table 3-1. NPDES Permits in the Watersheds Draining to Impaired Lakes

Type of NPDES Permit	Number of Permits
Municipal Separate Storm Sewer System (MS4)	3
California Department of Transportation Stormwater	1
General Construction Stormwater	1
General Industrial Stormwater	66
General NPDES Permits (Groundwater Discharges)	1
Total	**72**

3.1.1.1 Stormwater Permits

Stormwater runoff is regulated through the City of Long Beach MS4 permit, the Los Angeles County MS4 permit, the Ventura County MS4 permit, the statewide stormwater permit issued to Caltrans, the statewide Construction Activities Stormwater General Permit, and the statewide Industrial Activities Stormwater General Permit. The permitting process defines these discharges as point sources because the stormwater is discharged from the end of a stormwater conveyance system. Since the industrial and construction stormwater discharges are governed under NPDES permits, these discharges are treated as point sources in these TMDLs.

3.1.1.1.1 MS4 Stormwater Permits

In 1990, USEPA developed rules establishing Phase I of the NPDES stormwater program, designed to prevent pollutants from being washed by stormwater runoff into MS4s (or from being discharged directly into the MS4s) and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges.

Approved stormwater management programs for medium and large MS4s are required to address a variety of water quality-related issues, including roadway runoff management, municipally owned operations, and hazardous waste treatment. Large and medium MS4 operators are required to develop and implement Stormwater Management Plans that address, at a minimum, the following elements:

- Structural control maintenance
- Areas of significant development or redevelopment
- Roadway runoff management
- Flood control related to water quality issues
- Municipally owned operations such as landfills and wastewater treatment plants
- Municipally owned hazardous waste treatment, storage, or disposal sites
- Application of pesticides, herbicides, and fertilizers
- Illicit discharge detection and elimination
- Regulation of sites classified as associated with industrial activity
- Construction site and post-construction site runoff control
- Public education and outreach

The Los Angeles County MS4 Permit was renewed in December 2001 (Regional Board Order No. 01-182; CAS004001) and is on a five-year renewal cycle. There are 85 co-permittees covered under this permit, including 84 incorporated cities and the County of Los Angeles. The City of Long Beach MS4 permit was renewed on June 30, 1999 (Order No. R4-99-060; CAS004003) and is on a five-year renewal cycle. It solely covers the City of Long Beach. The Ventura County MS4 Permit was renewed in July 2010 (Order R4 2010-0108; CAS004002) and is on a five-year renewal cycle. This permit covers 12 co-permittees, including 10 incorporated cities, the County of Ventura, and the Ventura County Flood Control District (Principal Permittee).

3.1.1.1.2 Caltrans Stormwater Permit

Caltrans is regulated by a statewide stormwater discharge permit that covers all municipal stormwater activities and construction activities (State Board Order No. 99-06-DWQ; CAS000003). The Caltrans stormwater permit authorizes stormwater discharges from Caltrans properties such as the state highway system, park and ride facilities, and maintenance yards. The stormwater discharges from most of these Caltrans properties and facilities eventually end up in either a city or county storm drain.

3.1.1.1.3 General Stormwater Permits

In 1990, USEPA issued regulations for controlling pollutants in stormwater discharges from industrial sites (40 Code of Federal Regulations [CFR] Parts 122, 123, and 124) equal to or greater than five acres. The regulations require dischargers of stormwater associated with industrial activity to obtain an NPDES permit and to implement Best Available Technology Economically Achievable (BAT) to reduce or prevent nonconventional and toxic pollutants, including metals, in stormwater discharges and authorized non-storm discharges. On December 8, 1999, USEPA expanded the NPDES program to include stormwater discharges from construction sites that resulted in land disturbances equal to or greater than one acre (40 CFR Parts 122, 123, and 124).

On April 17, 1997, the State Board issued a statewide general NPDES permit for Discharges of Stormwater Associated with Industrial Activities Excluding Construction Activities Permit (Order No. 97-03-DWQ; CAS000002). This Order regulates stormwater discharges and authorized non-stormwater discharges from 10 specific categories of industrial facilities, including but not limited to, manufacturing facilities, oil and gas mining facilities, landfills, and transportation facilities. Potential pollutants from an industrial site will depend on the type of facility and operations that take place at that facility.

During wet weather, runoff from industrial sites has the potential to contribute pollutant loadings. During dry weather, the potential contribution of pollutant loadings from industrial stormwater is low because non-stormwater discharges are prohibited or authorized by the permit only under the following circumstances: when they do not contain significant quantities of pollutants, where Best Management Practices (BMPs) are in place to minimize contact with significant materials and reduce flow, and when they are in compliance with Regional Board and local agency requirements.

On September 2, 2009, the State Board adopted the statewide general NPDES permit for Discharges of Stormwater Associated with Construction and Land Disturbance Activities (Order No. 2009-0009-DQW;CAS000002). This General Construction Permit became effective on July 1, 2010. During wet weather, runoff from construction sites has the potential to contribute pollutant loadings. During dry weather, the potential contribution of pollutant loadings is low because discharges of non-stormwater are authorized by the permit only where they do not cause or contribute to a violation of any water quality standard and are controlled through implementation of appropriate BMPs for elimination or reduction of pollutants.

3.1.1.2 Other NPDES Permits

There are two types of non-stormwater NPDES permits: individual and general permits. An individual NPDES permit is classified as either a major or a minor permit. Other than the MS4 and Caltrans stormwater permits, there are no major individual NPDES permits in the watersheds draining to the impaired lakes. The discharge flows associated with minor individual NPDES permits and general NPDES permits are typically less than 1 million gallons per day (MGD). General NPDES permits often regulate episodic discharges (e.g., dewatering operations) rather than continuous flows.

Pursuant to 40 CFR parts 122 and 123, the State Board and the regional boards have the authority to issue general NPDES permits to regulate a category of point sources if the sources involve the same or substantially similar types of operations, discharge the same type of waste, require the same type of effluent limitations, and require similar monitoring. The Regional Board has issued general NPDES permits for six categories of discharges: construction and project dewatering, petroleum fuel cleanup sites, volatile organic compounds (VOCs) cleanup sites, potable water, non-process wastewater, and hydrostatic test water.

There is one facility in the Peck Road Park Lake watershed associated with the potable water general NPDES permit. The general NPDES permit for Discharges of Groundwater from Potable Water Supply

Wells to Surface Waters (Order No. R4-2003-0108; CAG994005) covers discharges of groundwater from potable supply wells generated during well purging, well rehabilitation and redevelopment, and well drilling, construction and development. The applicable numeric effluent limitations for these facilities can be found in Order No. R4-2003-0108.

3.1.2 Nonpoint Sources

A nonpoint source is a source that discharges via sheet flow or natural discharges, as well as agricultural stormwater discharges and return flows from irrigated agriculture. Nonpoint sources include atmospheric deposition directly onto lakes, areas that do not drain to a storm drain system, irrigation of parkland, and agricultural flows. Specific sources are described in the lake-specific chapters.

3.2 POLLUTANT-SPECIFIC APPROACH

This section provides a brief description of the technical approach used to develop TMDLs for nutrient-related, mercury, OC Pesticides and PCBs, and trash impairments. More details on the nutrient, mercury, and OC Pesticides and PCBs analyses are provided in Appendix A (Nutrient TMDL Development), Appendix C (Mercury TMDL Development), and Appendix H (Organochlorine Compounds TMDL Development), respectively.

3.2.1 Nutrient-related Impairments

Excessive algae in the urban lakes of the Los Angeles Region has resulted in several waterbodies not supporting their designated beneficial uses associated with aquatic life and recreation (LARWQCB, 1996). Algal biomass can lead to impairment of swimming and wading activities. In addition, the proliferation of algae can result in loss of invertebrate taxa through habitat alteration (Biggs, 2000). Algal growth in some instances has produced algal mats in the lakes (UC Riverside, 1994); these mats may result in eutrophic conditions where fluctuations in dissolved oxygen concentration and pH negatively affect aquatic life in the waterbody. The decay of these mats may also cause problems with scum and odors that affect recreational uses of the affected waterbody. In addition, the concentration of ammonia, a nitrogen compound, has been present in concentrations exceeding objectives designed to protect aquatic life (LARWQCB, 1996).

3.2.1.1 Source Assessment

Sources of nutrient loading to a lake may include both point and nonpoint sources. For purposes of allocations among nutrient sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which wasteload allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process. Point sources include MS4 dischargers and other NPDES discharges as well as additional inputs such as groundwater wells or potable water sources. Nutrient loading from nonpoint sources originates from sources that do not discharge at a defined point, including direct atmospheric deposition and watershed loadings not associated with an MS4 system. Appendices D and F (Wet and Dry Weather Loading, respectively) describe how loading from these point and nonpoint sources was estimated.

3.2.1.2 Linkage Analysis

To simulate the impacts of nutrient loading on each impaired lake, the Nutrient Numeric Endpoints (NNE) BATHTUB model was set up and calibrated to lake specific conditions (Appendix A, Nutrient TMDL Development, provides additional details). The NNE BATHTUB model is a risk-based approach for estimating site-specific nutrient numeric endpoints (NNE) for California waters (Tetra Tech, 2006). In recognizing the limitation of using ambient nutrient concentrations alone in predicting the impairment of beneficial uses, this approach uses secondary indicators. Secondary indicators are defined as parameters that are related to nutrient concentrations, but are more directly linked to beneficial uses than nutrient levels alone. The tool has been tested for several waterbodies in California as a series of case studies. The secondary indicator chosen to support TMDL development for these eight waterbodies is algal density, represented by chlorophyll *a*.

The NNE BATHTUB Tool was set up individually for each impaired lake. Bathymetry data for each lake were acquired from various sources to represent the general characteristics of the waterbody, such as surface area, volume, and average depth.

Cumulative nitrogen and phosphorus loads were input to each lake model as a sum of all known, quantifiable sources. Sources of loading resulting from wet weather are discussed in Appendix D; Appendix F summarizes the loading originating during dry weather conditions. Atmospheric deposition to each lake surface is quantified in Appendix E. Internal nutrient loading is discussed in Appendix B, but is not quantified directly due to lack of data (the BATHTUB model accounts for internal loading indirectly by using a net sedimentation rate (sedimentation minus resuspension)).

Once the bathymetry and loading inputs were set up, each model was calibrated to fit observed summer (May – September) mean concentrations of phosphorus, nitrogen, and chlorophyll *a*. The calibrated models were then used to determine the allowable loads of nitrogen and phosphorus that result in attainment of the chlorophyll *a* target concentration. Allowable loads were allocated among the wasteload allocations, load allocations, and margins of safety.

For Santa Fe Dam Park Lake, which is impaired by pH, the NNE BATHTUB Tool indicated that it is not directly impaired by elevated nutrient loads or excessive algal growth. To investigate the likely source of the pH impairment, a steady-state, chemical equilibrium model was also set up. Specifically, the geochemical speciation model, Visual MINTEQ V2.61 (Gustafsson, 2009), was used to investigate the pH conditions in the lake. The model was selected to perform pH simulation based on the available data for Santa Fe Dam Park Lake. The model requires total analytical concentrations and physical inputs to evaluate various geochemical reactions. The results were used to evaluate whether elevated pH was due to natural conditions, algal impacts, or the addition of chlorine in the form of sodium hypochlorite (NaOCl), for disinfection of the swim beach area.

3.2.2 Mercury Impairment

Mercury, like other metals, has great persistence due to its inability to be broken down. However, because bacterial processes can methylate it to create methylmercury, it also has some properties of a bioaccumulative organic chemical. Methylmercury is easily taken up by organisms and tends to bioaccumulate; it is very effectively transferred through the food web, magnifying at each trophic level. This can result in high levels of mercury in organisms high on the food chain, despite nearly unmeasurable quantities of mercury in the water column. While mercury can be toxic to fish and other aquatic organisms at high levels, the primary concerns at the levels found in these lakes are neurological and developmental effects in higher animals and humans. The two primary endpoints of concern are wildlife species that eat fish and people that consume sport fish.

Methylmercury is highly toxic to mammals, including people, and causes a number of adverse effects. Health studies and information showing neurotoxicity, particularly in developing organisms, are most abundant. The brain is the most sensitive organ for which suitable data are available to quantify a dose-response relationship. A study by the National Academy of Science (NRC, 2000) concluded that the population at highest risk is the children of women who consume large amounts of fish and seafood during pregnancy, and that the risk to that population may result in an increase in the number of children struggling to keep up in school and requiring remedial classes or special education (USEPA, 2001a). Each of the three lakes impaired by mercury have mercury levels in largemouth bass, a trophic level four species (see Section 2.2.9), above the recommended fish consumption guideline (OEHHA, 2008). Methylmercury is also toxic to fish-eating wildlife, including both mammals and birds. In addition to neurotoxic effects, methylmercury is implicated in reduced reproductive success in wildlife such as eagles, osprey, otter, and mink (Wiener et al., 2002).

3.2.2.1 Source Assessment

Sources of mercury loading to a lake may include both point and nonpoint sources. For purposes of allocating among mercury sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which wasteload allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). The most significant source of mercury in point source discharges is wastewater associated with the placement or removal of mercury amalgam dental fillings. Significant sources in the watershed include junkyards housing automobiles where mercury-containing switches have not been removed prior to crushing, and landfills where fluorescent light bulbs have not been properly disposed. Significant releases to the atmosphere may occur from coal-power plants, cement manufacturing facilities, oil refineries, and chlor-alkali plants.

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process. Point sources include MS4 dischargers and other NPDES discharges as well as additional inputs such as groundwater wells or potable water sources. Mercury loading from nonpoint sources originates from sources that do not discharge at a defined point, including direct atmospheric deposition, watershed loadings not associated with an MS4 system, methylation, and direct and indirect geologic sources. Appendices D and F (Wet and Dry Weather Loading, respectively) describe how loading from these point and nonpoint sources was estimated.

3.2.2.2 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. Specifically, models of watershed loading of mercury are combined with an estimated rate of bioaccumulation in the lake. This enables a translation between the numeric target (expressed as a fish tissue concentration of mercury) and mercury loading rates. The loading capacity is then determined via the linkage analysis as the mercury loading rate that is consistent with meeting the target fish tissue concentration. This process is described in detail in Appendix C (Mercury TMDL Development) and summarized below.

For the three mercury-impaired lakes addressed by this document, models of lake response and fish bioaccumulation have not been created at this time. Rather, it is assumed that, in the long term, fish tissue concentrations will respond approximately linearly to reductions in mercury load (see Appendix C, Mercury TMDL Development). Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass, the predominant trophic level 4 fish in each waterbody. To do this, a linear regression analysis was performed on tissue concentrations versus length from data collected in each lake, which was then used to predict the existing concentration associated with the target size fish.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in each lake. The target is established for a 350 mm largemouth bass to be measured in fish 325-375 mm in length. The predicted mercury concentration based on a one-sided 95 percent upper confidence limit on mean predictions about the regression line (95 percent UCL) for this length is compared to the target fish concentration to determine the required reduction in mercury loading, which includes a margin of safety as described in Appendix C (Mercury TMDL Development).

3.2.3 Organochlorine Pesticides and PCBs Impairments

Organochlorine (OC) Pesticides and PCBs are chemical substances that persist in the environment, bioaccumulate through the food web, and pose a risk of causing adverse effects to human health and the environment. In particular, they include a number of chlorinated legacy pollutants known or suspected to be carcinogenic and/or toxic to humans and wildlife. OC Pesticides and PCBs include a number of now-banned chlorinated pesticides (e.g., chlordane, dieldrin, and DDT) and polychlorinated biphenyls (PCBs) that are causes of impairment in Los Angeles Region lakes. OC Pesticides and PCBs are problematic because they do not break down easily, concentrate in organisms, and can be transported great distances. The primary concerns for the listed lakes are the high levels found in popularly consumed fish. Their continuous cycling in the food chain and accumulation in sediments creates difficulties in their removal from lake systems. While concentration in sediment and organisms may be high, concentrations in the water column are often undetectable.

The US has banned the manufacture or use of all the pollutants considered OC Pesticides (chlordane, DDT, and dieldrin) and PCBs that are listed as causes of impairment in the lakes. However, the past use of these chemicals was so widespread and unrestricted that there are still loads of these chemicals coming from waste and storage facilities as well as old equipment that used or contained the contaminants. Chlordane, DDT, and dieldrin were also widely applied for agricultural and domestic pest control purposes. Continued research and findings repeatedly demonstrate that these pollutants are ubiquitous.

3.2.3.1 Source Assessment

Sources of OC Pesticides and PCBs loading to a lake may include both point and nonpoint sources. All OC Pesticides and PCBs listed for the impaired lakes were banned from domestic and industrial use by the 1980s. Areas of concern include waste facilities that may contain old transformers, industrial sites, agriculture lands, and some residences that were treated heavily for pests (for example: chlordane was a popular termiticide in the 1970s). Even areas that do not have a history of OC Pesticides and PCBs use or storage are vulnerable due to atmospheric deposition, often derived from transcontinental transport.

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process. Point sources include MS4 dischargers and other NPDES discharges, as well as additional inputs such as groundwater wells or potable water sources. Loading from nonpoint sources originates from sources that do not discharge at a defined point, including direct atmospheric deposition and watershed loadings not associated with an MS4 system. The only sources of OC Pesticides and PCBs in the local area are watershed loadings, which were divided into wasteload allocations or load allocations, depending on the presence of storm drain systems in the drainage areas (i.e., areas draining to a storm drain will receive wasteload allocations). Atmospheric deposition is incorporated into the indirect loading from watershed runoff. Direct deposition to the lake surface is considered negligible. Appendix D (Wet Weather Loading) describes how loading from these point and nonpoint sources was estimated, and the calculated loadings and allocations are described in detail in Appendix H (Organochlorine Compounds TMDL Development).

3.2.3.2 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. Specifically, equilibrium models of watershed loading of OC Pesticides and PCBs, lake processes, and pollutant bioaccumulation in the fish have been developed. This enables a translation between numeric targets (expressed as a fish tissue concentration for each listed contaminant) and loading rates. This process is described in detail in Appendix H (Organochlorine Compounds TMDL Development) and summarized below.

The OC Pesticides and PCBs of concern have low solubility and a high affinity for organic solids and lipids. Thus, concentrations present in the sediment can result in unacceptable concentrations in fish tissue, due to food chain accumulation pathways that lead back to the lake sediment, even when concentrations in the water column are below criteria or non-detectable. The sediment concentration target is estimated using the Biota-Sediment Accumulation Factor (BSAF) of each contaminant. Starting from the fish tissue concentration target, the BSAF allows calculation of the necessary sediment concentration to support uses, and the allowable load to achieve the target sediment concentration. This is explained in detail in Appendix H (Organochlorine Compounds TMDL Development).

The target for fish tissue is provided by the 2008 Office of Environmental Health Hazard Assessment (OEHHA) Fish Contaminant Goal (FCG). The target fish concentrations are discussed further in Section 2 and Appendix H (Organochlorine Compounds TMDL Development). Addressing the fish tissue concentrations as the assessment endpoint also achieves most other applicable targets for sediment and water concentrations. The loading capacity for sediment-associated OC Pesticides and PCBs is then determined from the lower of the sediment concentration target to meet the FCG and any other applicable targets for sediment, such as the consensus-based sediment quality guidelines (MacDonald et al., 2000) designed to protect benthic organisms. This loading capacity is expressed as a sediment concentration (ng of pollutant per gram of dry sediment), which is applicable to both sediments already stored in the lake and new sediment washed into the lake. Runoff from the watershed must achieve this sediment concentration to satisfy the TMDL. Both wasteload allocations and load allocations may be translated into pollutant mass units by multiplying the OC Pesticides and PCBs concentration on sediment times the sediment load.

3.2.4 Trash Impairment

Trash in waterways causes significant water quality problems. Small and large floatables can inhibit the growth of aquatic vegetation, leading to shrinking spawning areas and habitats for fish and other living organisms. Wildlife living in lakes and riparian areas can be harmed by ingesting or becoming entangled in floating trash. With the exception of large items, settleables are not always obvious to the eye. This includes glass, cigarette butts, rubber, and construction debris. Settleables can be a problem for bottom feeders and can contribute to sediment contamination. Some debris (e.g., diapers, medical and household waste, and chemicals) are sources of bacteria and toxic substances.

For aquatic life, buoyant (floatable) materials tend to be more harmful than settleable elements, due to their ability to be transported throughout the waterbody and ultimately to the marine environment. Persistent elements such as plastics, synthetic rubber and synthetic cloth tend to be more harmful than degradable elements such as paper or organic waste. Glass and metal are less persistent because wave action and rusting can cause them to break into smaller pieces that are less sharp and harmful. Natural rubber and cloth can degrade but not as quickly as paper (USEPA, 2002). Smaller elements such as plastic resin pellets (a byproduct of plastic manufacturing) and cigarette butts can be ingested by a large number of small organisms which can then suffer malnutrition or internal injuries. Larger plastic elements such as plastic grocery bags are also harmful to larger aquatic life, which can mistake the trash for floating prey and ingest it, leading to starvation or suffocation.

Trash impaired waterbodies can threaten the health of people who swim and recreate in them. Of particular concern are bacteria and viruses associated with diapers, medical waste (e.g., used hypodermic needles and pipettes), and human or pet waste. Additionally, broken glass or sharp metal fragments in streams can cause puncture or laceration injuries. Such injuries can expose a person's bloodstream to microbes in the stream's water causing serious illnesses. Some trash items such as containers or tires can cause a pooling of water and create opportunities for mosquito production and increase health risks, such as encephalitis and West Nile virus.

Leaf litter is considered trash when there is evidence of intentional dumping. Leaves and pine needles in streams provide a natural source of food for organisms, but excessive amounts due to human influence can cause nutrient imbalance and oxygen depletion in streams. Clumps of leaf litter and yard waste from trash bags should be treated as trash during water quality assessments, and should not be confused with natural inputs of leaves to streams. In some instances, leaf litter may be trash if it originated from dense ornamental stands of nearby human planted trees that are overloading the stream's assimilative capacity for leaf inputs. Other biodegradable trash, such as food waste, can also negatively impact natural dissolved oxygen levels in the waterbodies.

Wildlife impacts due to trash occur in Peck Road Park Lake, Lincoln Park Lake, and Echo Park Lake. The two primary problems that trash poses to wildlife are entanglement and ingestion, with entanglement being the more common documented effect (Laist and Liffmann, 2000). Marine mammals, turtles, birds, fish, and crustaceans all have been affected by entanglement or ingestion of floatable debris. The most vulnerable species to floatable debris are those endangered or threatened by extinction.

Entanglement results when an animal becomes encircled or ensnared by debris. It can occur accidentally, or when the animal is attracted to the debris. Entanglement is harmful to wildlife for several reasons. Not only can it cause wounds leading to infections or loss of limbs, it can also cause strangulation or suffocation. In addition, entanglement can impair an animal's ability to swim, which can result in drowning, difficulty in moving, finding food, or escaping predators (USEPA, 2001a).

Ingestion occurs when an animal swallows floatable debris. It sometimes occurs accidentally, but usually animals feed on debris because it looks like food (e.g., plastic bags look like jellyfish, a prey item of sea turtles). Ingestion can lead to starvation or malnutrition if the ingested items block the intestinal tract and prevent digestion, or accumulate in the digestive tract, making the animal feel "full" and lessening its desire to feed. Ingestion of sharp objects can damage the mouth, digestive tract and/or stomach lining and cause infection or pain. Ingested items can also block air passages and prevent breathing, thereby causing death (USEPA, 2001a).

Common settled debris includes glass, cigarettes, rubber, and construction debris. Settleables are a problem for bottom feeders and dwellers and can contribute to sediment contamination.

In conclusion, trash in waterbodies can adversely affect humans, fish, and wildlife. Not all water quality effects of trash are equal in severity or duration. The water quality effects of trash depend on individual items and their buoyancy, degradability, size, potential health hazard, and potential hazards to fish and wildlife.

The prevention and removal of trash in waterbodies will ultimately lead to improved water quality, protection of aquatic life and habitat, improved opportunities for public recreational access and restoration activities, enhancement of public interest in the lakes, propagation of the vision of the watershed as a whole, and enhancement of the quality of life of riparian residents.

3.2.4.1 Source Assessment

The major source of trash in these lakes is due to litter, which is intentionally or accidentally discarded to the lake and watershed. Potential sources can be categorized as point sources and nonpoint sources depending on the transport mechanisms. For example:

1. Storm drains: trash deposited throughout the watershed and carried to various sections of the lake during and after rainstorms via storm drains. This is a point source.
2. Wind action: trash blown into the lake directly. This is a nonpoint source.
3. Direct disposal: direct dumping or littering into the lake. This is a nonpoint source.

3.2.4.1.1 Point Sources

Litter is the primary source of trash for point sources. This includes trash deposited throughout the watershed and carried to the waterbodies during and after rain events via storm drains.

3.2.4.1.2 Nonpoint Sources

Litter is also intentionally or accidentally discarded to the lake and shoreline. Trash deposited near the lake has the potential to be blown or transported by wildlife or overland flow into the lake. Trash directly dumped into the lake is also a nonpoint source.

3.2.4.2 Linkage Analysis

These TMDLs are based on numeric targets derived from narrative water quality objectives in the Los Angeles Basin Plan (LARWQCB, 1994) for floating materials and solid, suspended, or settleable materials. The narrative objectives state that waters shall not contain these materials in concentrations that cause nuisance or adversely affect beneficial uses. Since any amount of trash impairs beneficial uses, the loading capacity of all waterbodies is set to zero allowable trash.

4 Peck Road Park Lake TMDLs

Peck Road Park Lake (#CAL4053100020000303195323) is listed as impaired for chlordane, DDT, eutrophication (originally on the consent decree, but not on current 303(d) list), lead, odor, organic enrichment/ low dissolved oxygen, and trash (SWRCB, 2010). In addition, dieldrin and PCB impairments have been identified by new data analyses since the 2008-2010 303(d) list data cut off. This section of the TMDL report describes the impairments and the TMDLs developed to address them: nutrients (see Section 4.2), organochlorine (OC) pesticides and PCBs (Sections 4.4 through 4.7), and trash (Section 4.8). Nutrient TMDLs are identified here based on existing conditions since nitrogen and phosphorus levels are achieving the chlorophyll *a* target level. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that lead is currently achieving numeric targets at Peck Road Park Lake; therefore, a TMDL is not included for this pollutant. Analyses for lead are presented below (Section 4.3).

4.1 Environmental Setting

Peck Road Park Lake is located in the Los Angeles River Basin (HUC 18070105) in the city of Arcadia (Figure 4-1). The lake was originally a gravel pit that was converted to a lake and park in 1975 by the Los Angeles County Parks and Recreation Department (Figure 4-2). Recreation is primarily limited to fishing; trout are periodically stocked by the California Department of Fish and Game (CDFG, 2009). Visitors are not allowed to boat or swim in the lake. Bird feeding is another recreational activity at Peck Road Park Lake. While no bird feeding has been observed during recent fieldwork, birds do feed from trash cans and food litter at the park. The Arcadia Golf Course is located on the northwest shoreline and a recreational path encircles the lake. Restrooms in the park are connected to the city sewer system.

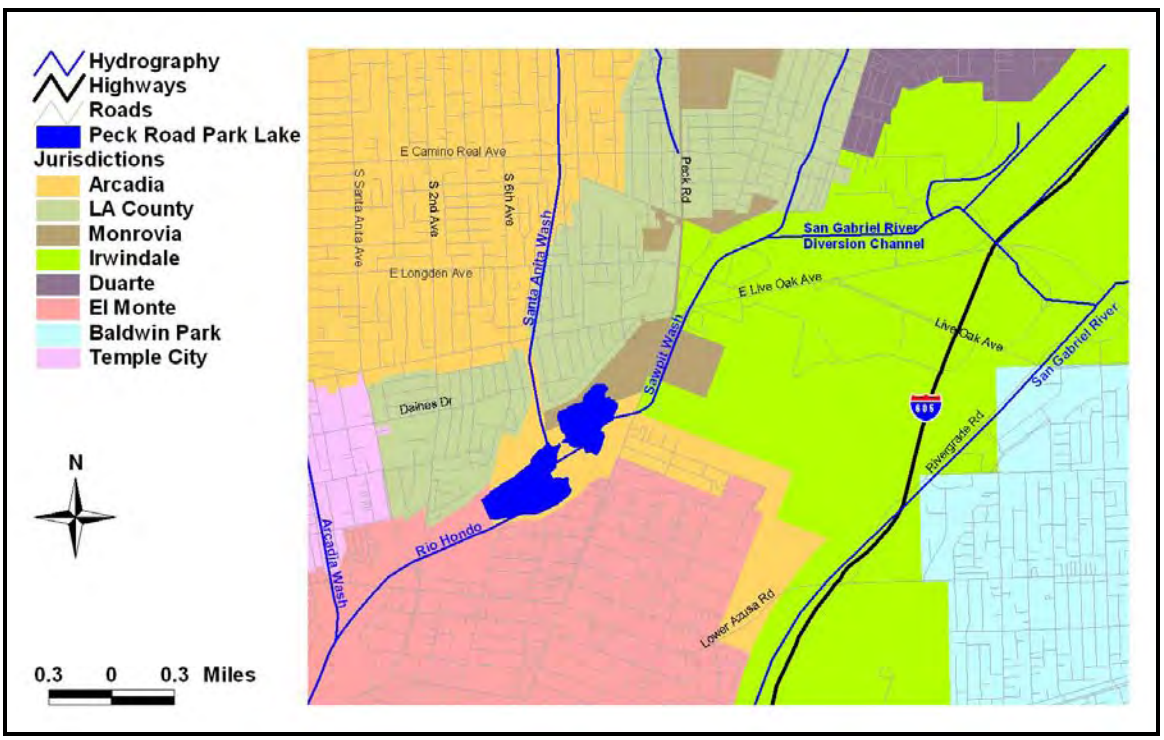

Figure 4-1. **Location of Peck Road Park Lake**

Two basins (north and south) connected by a narrow waterway have a surface area of 87.4 acres (based on Southern California Association of Governments [SCAG] 2005 land use), average depth of 30 feet (depth was calculated as an average of 2008 and 2009 sampling depths), and total volume of 2,622 acre-feet (calculated from the land use estimated surface area and average sampling depths). Inflows to the Lake include Sawpit Wash (Figure 4-3), Santa Anita Wash (Figure 4-4), and diversions from the Santa Fe Flood Control Basin. Water leaving Peck Road Park Lake discharges into Rio Hondo Wash. There is no known use of algaecide in this lake. Additional characteristics of the watershed are summarized below.

Figure 4-2. **Views of Peck Road Park Lake (Northern end on left; Southern lobe on right)**

Figure 4-3. **Sawpit Wash**

Figure 4-4. **Santa Anita Wash**

4.1.1 Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries

The Peck Road Park Lake watershed (23,564 total acres) ranges in elevation from 74 meters to 1,738 meters. The TMDL subwatershed boundaries selected for Peck Road Park Lake were based on more discrete boundaries obtained from the county of Los Angeles that were aggregated to three larger drainages. The subwatershed draining the western part of the watershed via Santa Anita Wash is 12,686 acres; the eastern subwatershed draining to Sawpit Wash is 10,557 acres. There is a mining operation in the southern part of the eastern watershed that has been removed from the loading analysis as it acts like a sink and does not drain towards the lake. The area surrounding the lake comprises 321 acres. Each subwatershed drains to a storm sewer system so all allocations except for trash will be wasteload allocations (Figure 4-5) (note: atmospheric deposition will be included as a load allocation). The spatial coverage for the storm drain network was obtained from the county of Los Angeles and is labeled on the figure accordingly. The trash TMDL includes load allocations due to direct dumping of trash along the shoreline and in the water by park visitors in the park area indicated in Figure 4-16 in the trash TMDL section.

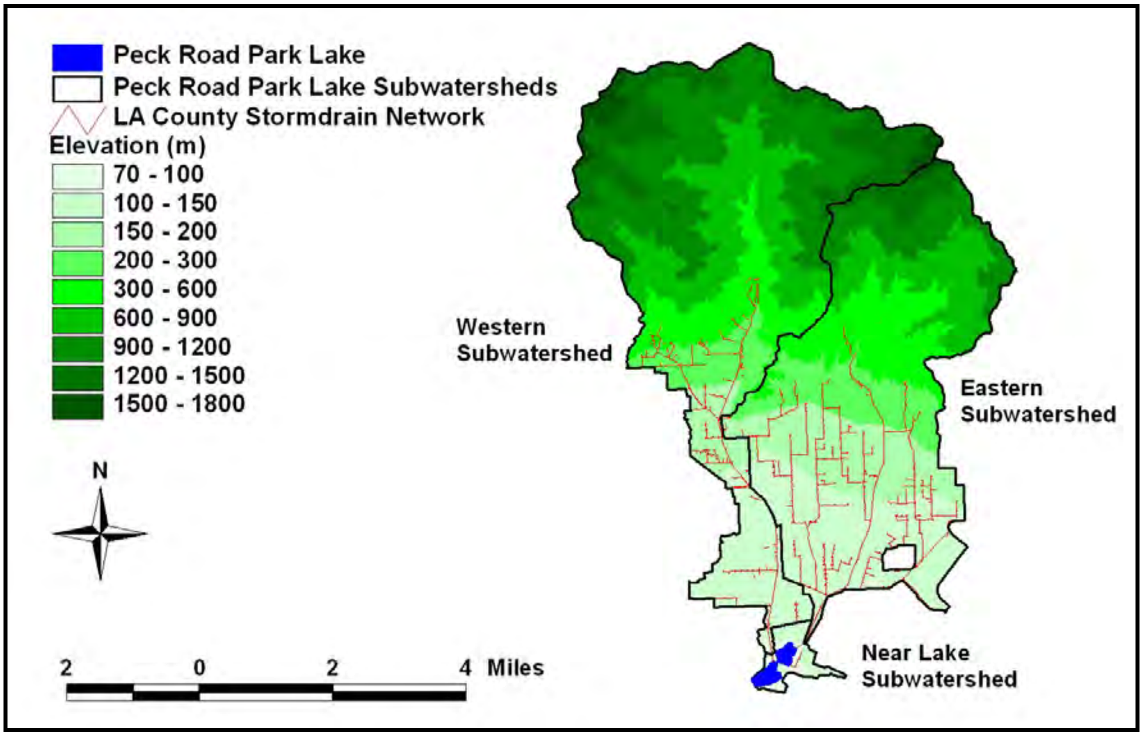

Figure 4-5. **Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries for Peck Road Park Lake**

4.1.2 MS4 Permittees

Figure 4-6 shows the MS4 stormwater permittees in the Peck Road Park Lake watershed. The western subwatershed is comprised of the county of Los Angeles, Sierra Madre, Arcadia, Monrovia, Angeles National Forest, and Caltrans areas. The eastern subwatershed is comprised of the county of Los Angeles, Monrovia, Duarte, Bradbury, Arcadia, Irwindale, Angeles National Forest, and Caltrans areas. The county of Los Angeles, Monrovia, Irwindale, Arcadia, and El Monte comprise the drainage around the lake. The park area is comprised of 152 acres adjacent to the lake.

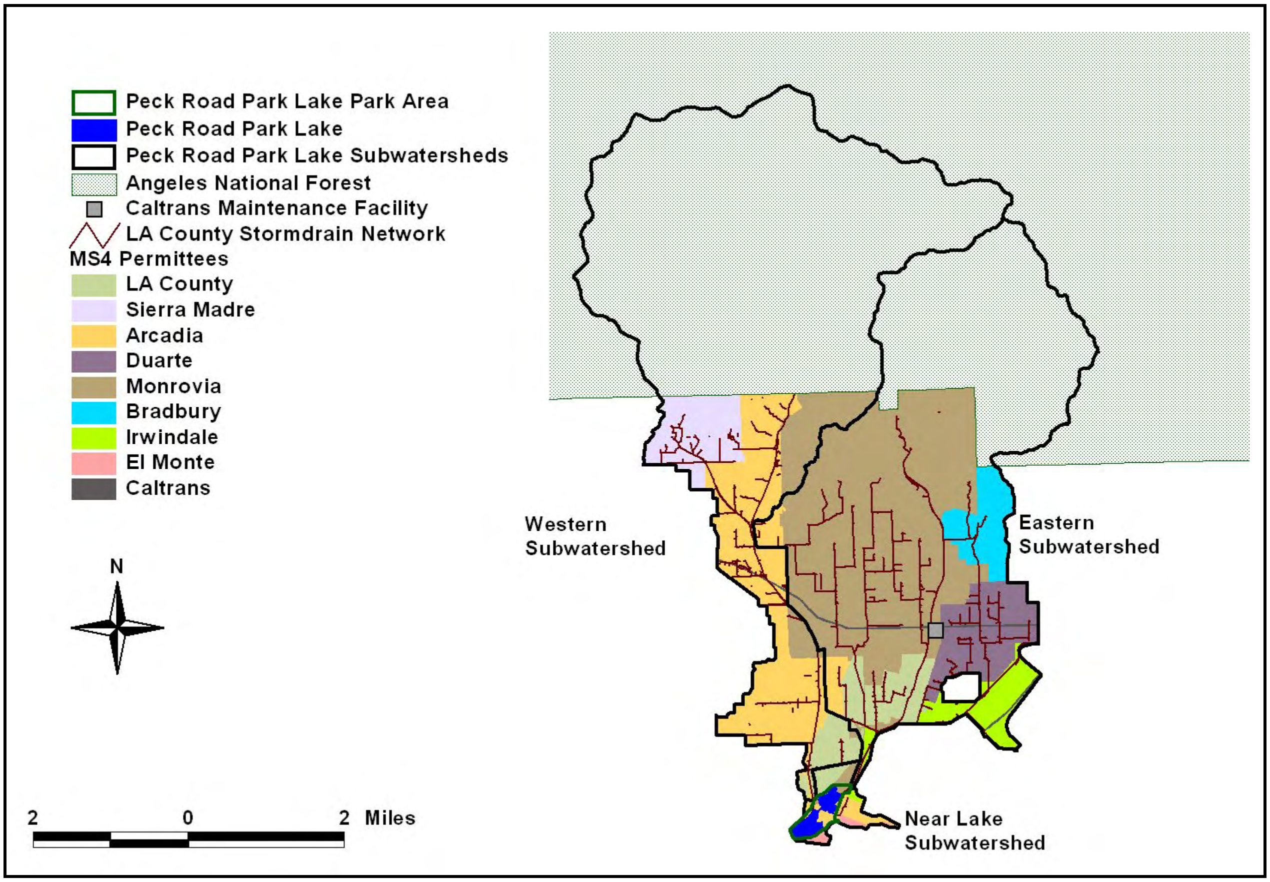

Figure 4-6. **MS4 Permittees and the Storm Drain Network in the Peck Road Park Lake Subwatersheds**

4.1.3 Non-MS4 NPDES Dischargers

There are several additional NPDES permits (non-MS4) in the Peck Road Park Lake watershed (Table 4-1). These include 53 dischargers covered under a general industrial stormwater permit (see Section 3.1 for a detailed discussion of these permit types) located throughout the watershed (Figure 4-7) that result in 510 disturbed acres. These permits were identified by querying excel files of permits from the Regional Board website (Excel files for each watershed are available from this link, www.waterboards.ca.gov/losangeles/water_issues/programs/regional_program/index.shtml#watershed, accessed on October 5, 2009). Specific information is not available regarding these dischargers; however, they are assigned existing loads and wasteload allocations based on their area (industrial stormwater) and their disturbed area (construction stormwater). There is one general NPDES permit for discharge of groundwater from potable water well maintenance activities, which will receive a concentration-based wasteload allocation.

Table 4-1. Non-MS4 Permits in the Peck Road Park Lake Watershed

Type of NPDES Permit	Number of Permits	Subwatershed	Jurisdiction	Disturbed Area
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	24	Eastern	Duarte	33.0

Type of NPDES Permit	Number of Permits	Subwatershed	Jurisdiction	Disturbed Area
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	10	Eastern	Irwindale	19.5
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	16	Eastern	Monrovia	133.5
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	1	Near Lake	Arcadia	14
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	1	Western	Arcadia	310
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	1	Western	Sierra Madre	0
General NPDES Permit for Potable Groundwater Well Discharges to Surface Water (Order No. R4-2003-0108, CAG994005)	1	Eastern	Arcadia	0

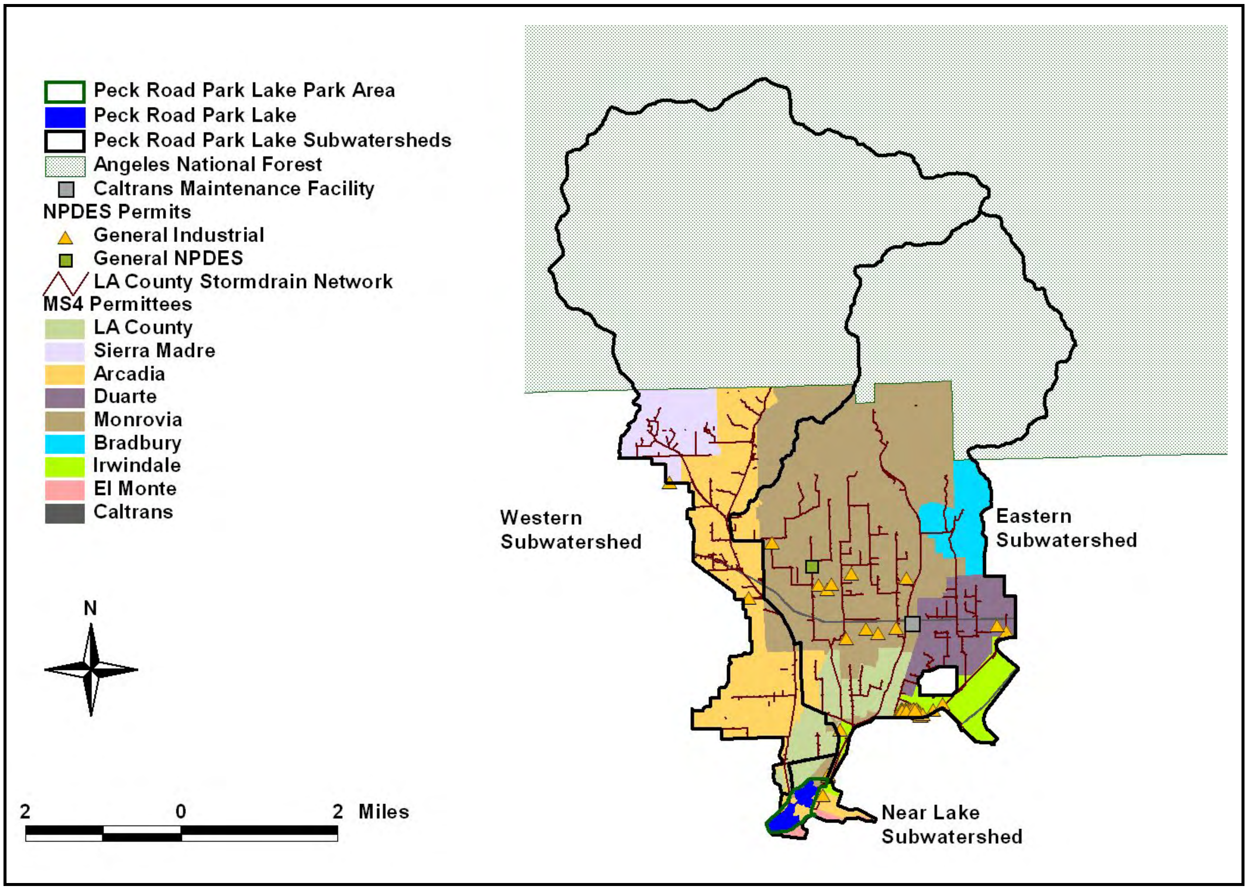

Figure 4-7. **Non-MS4 Permits in the Peck Road Park Lake Subwatersheds**

4.1.4 Land Uses and Soil Types

Several of the analyses for the Peck Road Park Lake watershed include source loading estimates obtained from the Los Angeles River Basin LSPC Model discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the Los Angeles River Basin LSPC model are shown in Figure 4-8. Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that a portion of the areas classified by the LSPC model as agriculture were inaccurate. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Approximately 82 acres classified by LSPC as agriculture corresponded to orchards, vineyards, and horse farms and were not altered. However, approximately 27 acres of agriculture were reclassified as open space and 28 acres were reclassified as residential. All areas within the Caltrans jurisdiction were simulated as industrial since the Los Angeles River Basin LSPC model grouped transportation uses into the industrial category. Table 4-2, Table 4-3, and Table 4-4 summarize the land use areas for each TMDL subwatershed and jurisdiction.

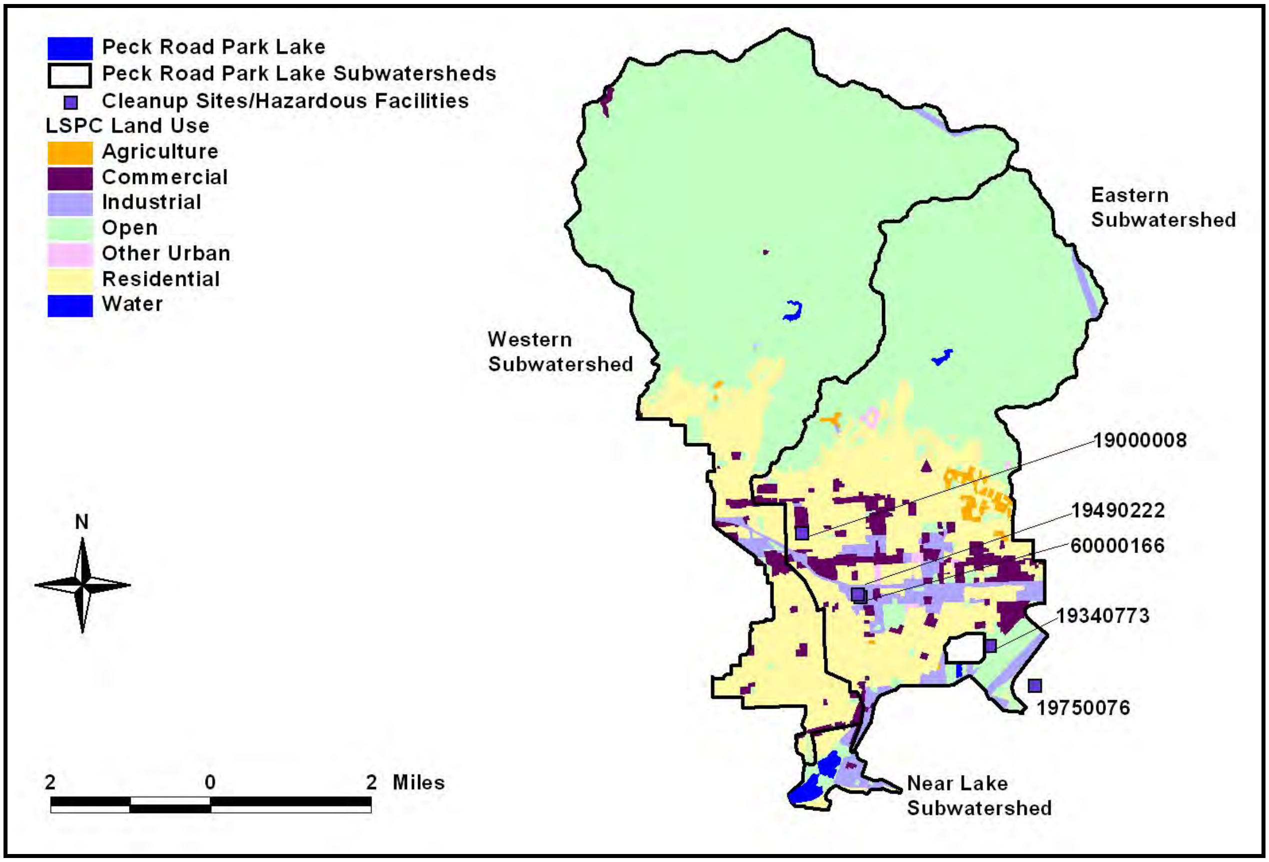

Figure 4-8. **LSPC Land Use Classes for the Peck Road Park Lake Subwatersheds**

Table 4-2. Land Use Areas (ac) Draining from Western Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Sierra Madre	Arcadia	Monrovia	Caltrans	Angeles National Forest	Total
Agriculture	0	4.19	0	0	0	**0**	**4.19**
Commercial	34.8	2.62	124	13.0	0	**0**	**175**
Industrial	0	0	70.4	0.319	16.9	**0**	**87.6**
Open	3.50	377	319	483	0	**9,104**	**10,286**
Other Urban	0	0	0.053	0	0	**0**	**0.053**
Residential	207	296	1,516	114	0	**0**	**2,133**
Total	**245**	**679**	**2,030**	**611**	**16.9**	**9,104**	**12,686**

Table 4-3. Land Use Areas (ac) Draining from Eastern Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Duarte	Bradbury	Arcadia	Irwindale	Caltrans	Angeles National Forest	Total
Agriculture	0	0	0	78.1	0	0	0	0	78.1
Commercial	24.8	430	232	0	33.9	12.7	0	0	733
Industrial	1.27	407	107	0	0	180	78.4	0	774
Open	5.29	1,419	53.5	229	16.0	274	0	3,511	5,508
Other Urban	0	51.0	1.74	2.90	1.71	0	0	0	57.3
Residential	467	2,149	424	193	158	15.5	0	0	3,406
Total	**499**	**4,456**	**818**	**503**	**209**	**483**	**78.4**	**3,511**	**10,557**

Table 4-4. Land Use Areas (ac) Draining from Near Lake Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Irwindale	Arcadia	El Monte	Total
Agriculture	0	0	0	0	0	**0**
Commercial	7.10	7.90	0	3.86	0	**18.9**
Industrial	0.0003	14.4	13.9	69.7	10.2	**108**
Open	0.233	24.6	0.187	61.6	0.984	**87.5**
Other Urban	0	0	0	0	0	**0**
Residential	60.4	1.30	0	4.18	40.9	**107**
Total	**67.7**	**48.1**	**14.1**	**139**	**52.1**	**321**

There are four Resource Conservation and Recovery Act (RCRA) cleanup sites within the Peck Road Park Lake watershed, and an additional RCRA cleanup site is located within 0.3 miles of the watershed. None of the active sites are expected to contribute to the existing nutrient, OC pesticides and PCBs, or trash impairments; however, some of the previously remediated locations may have historically contributed PCB loadings. In addition, as identified in Table 4-5, several facilities have the potential to discharge lead, but lead is currently meeting numeric targets in Peck Road Park Lake (Section 4.3). Table 4-5 summarizes the available information regarding these sites, which are illustrated in Figure 4-8.

Table 4-5. RCRA Cleanup Sites Located within or near the Peck Road Park Lake Watershed

Envirostor #	Facility Name	Cleanup Status	Potential Contaminants of Concern
19750076	Alpha II/Irwindale	No further action	Lead, polychlorinated biphenyls (PCBs), cadmium
60000166	Metric Machining	Active	Arsenic, motor oil, polycyclic aromatic hydrocarbons (PAHs)
19490222	So Cal Gas/Monrovia Mgp	Active	Lead, arsenic, polycyclic aromatic hydrocarbons (PAHs), cyanide
19340773	Southwest Products/Irwindale	No further action	Benzene
19000008	Trotter Apartments	Certified	Lead

Figure 4-9 shows the predominant soils identified by STATSGO in the Peck Road Park Lake subwatersheds. The most predominant soil type is Sobrante-Exchequer-Cieneba (MUKEY 660501), which is a hydrologic group C soil characterized as moderately-fine to fine-textured soils having low infiltration rates when wet and consisting chiefly of soils having a layer that impedes downward movement of water. In the headwaters of the watershed there is a small area of Tollhouse-Rock outcrop-Etsel family-Bakeoven soil, a hydrologic group D soil (MUKEY 660505), which has high runoff potential, very low infiltration rates, and consists chiefly of clay soils. The middle section of the watershed is comprised of Zamora-Urban land-Ramona soil (MUKEY 660480) for which the STATSGO database does not list the hydrologic soil group. Soil Urban land-Sorrento-Hanford (MUKEY 660473) makes up the southern part of the watershed. This soil is a hydrologic group B soil, which has moderate infiltration rates and moderately coarse textures.

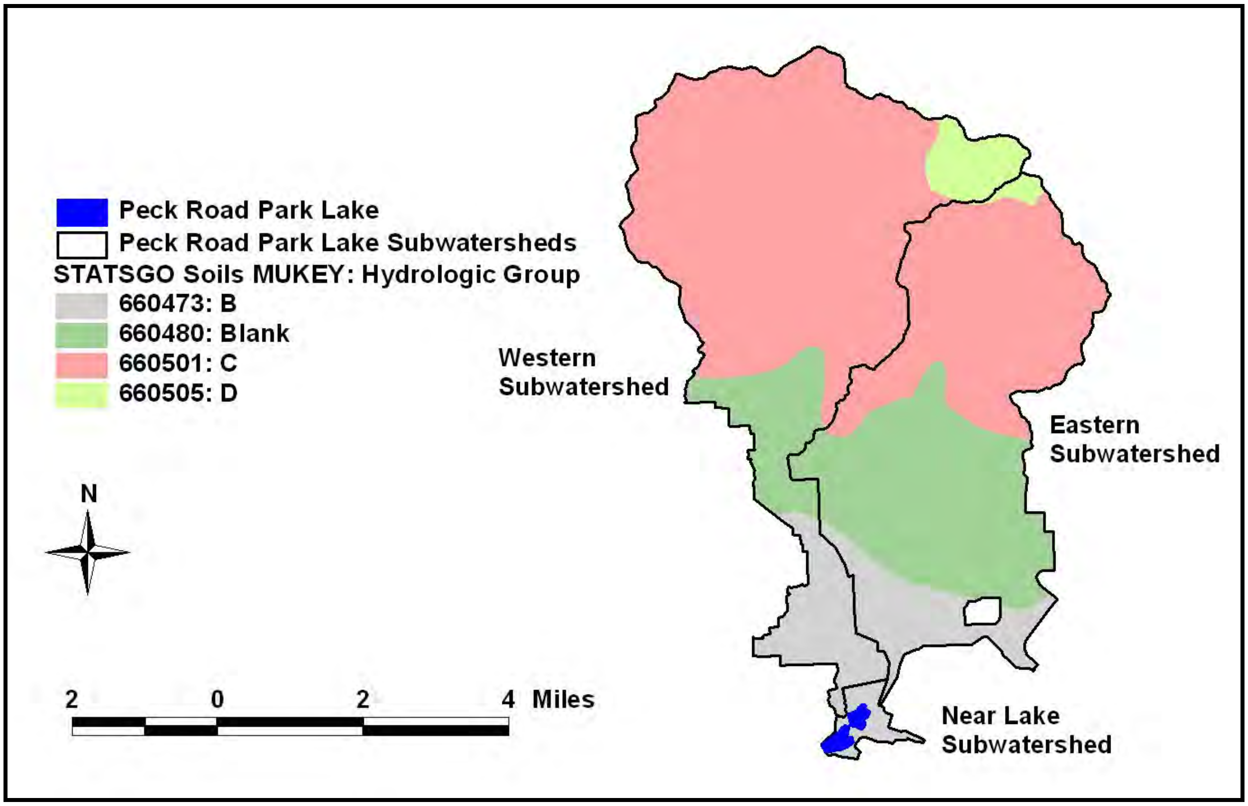

Figure 4-9. **STATSGO Soil Types Present in the Peck Road Park Lake Subwatersheds**

4.1.5 Additional Inputs

The 1994 Urban Lakes Study identified diversions of flow from the San Gabriel River as the primary source of water to Peck Road Park Lake. Based on data provided by the Los Angeles County Public Works Department, diversions provide an average of 8,737 ac-ft of water to Peck Road Park Lake annually. A small area of parkland is irrigated; however, it is greater than 600 ft from the lake and all of the water is expected to percolate into the ground and not reach the lake. It is therefore not included in the analysis.

4.2 Nutrient-Related Impairments

A number of the assessed impairments for Peck Road Park Lake may be associated with nutrients and eutrophication. Nutrient-related impairments for Peck Road Park Lake include odor and organic enrichment/low dissolved oxygen (DO) (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algae produce oxygen during photosynthesis but remove oxygen during respiration processes that occur in the absence of sunlight. Death and decay of large amounts of algae may cause odor problems by creating an anoxic environment that results in the release of sulfuric compounds.

4.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality

Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, and WARM uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels alter biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD and MUN uses could become impaired.

4.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Peck Road Park Lake. The following targets apply to the odor and organic enrichment/low DO (see Section 2 for additional details and Table 4-6 for a summary):

- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient (e.g., nitrogen and phosphorous) concentrations in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Peck Road Park Lake; however, as described in Tetra Tech (2006), summer (May – September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and is also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target must be met at half of the Secchi depth during the summer (May – September) and annual averaging periods.

- The Basin Plan states that "waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses."

- The Basin Plan states "at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges." Deep lakes that thermally stratify during the summer months, such as Peck Road Park Lake, must meet the DO target in the epilimnion of the water column.

  The epilimnion is the upper stratum of more or less uniformly warm, circulating, and fairly turbulent water during summer stratification. The epilimnion floats above a cold relatively undisturbed region called the hypolimnion. The stratum between the two is the metalimnion and is characterized by a thermocline, which refers to the plane of maximum rate of decrease of temperature with respect to depth. For the purposes of these TMDLs, the presence of stratification will be defined by whether there is a change in lake temperature greater than 1 degree Celsius per meter. Deep lakes, such as Peck Road Park Lake, must meet the DO and pH targets in the water column from the surface to 0.3 meters above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the DO and pH targets must be met in the epilimnion, the portion of the water column above the thermocline.

- The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." Deep lakes that thermally stratify during the summer months, such as Peck Road Park Lake, must meet the pH target in the epilimnion of the water column.

Nitrogen and phosphorus target concentrations within the lake are based on existing conditions as explained in Sections 4.2.5 and 4.2.6:

- 0.76 mg-N/L summer season average (May – September) and annual average
- 0.076 mg-P/L summer season average (May – September) and annual average

Table 4-6. Nutrient-Related Numeric Targets for Peck Road Park Lake

Parameter	Numeric Target	Notes
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.76 mg-N/L summer average (May – September) and annual average	Conservatively based on existing conditions, which are maintaining chlorophyll *a* levels below the target of 20 µg/L
Total Phosphorous	0.076 mg-P/L summer average (May – September) and annual average	Based on an in-lake TN to TP ratio of 10, typical of natural systems

4.2.3 Summary of Monitoring Data

Water quality in Peck Road Park Lake has been monitored since the early 1990s. This section summarizes the monitoring data relevant to the nutrient impairments. Additional details regarding monitoring are discussed in Appendix G (Monitoring Data).

The southern basin was sampled during the 1992-93 monitoring period in support of the Urban Lakes Study. Nutrient levels were analyzed at relatively high detection limits. Of the 90 orthophosphate samples collected, only one exceeds the detection limit of 0.1 mg-P/L. This measurement was collected at a depth of 8 meters and had a value of 0.4 mg-P/L. Only 1 of 90 total phosphorus samples exceeded the detection limit of 0.1 mg-P/L: at a depth of 5 meters the TP measurement was 0.9 mg-P/L. Three nitrite samples exceeded the detection limit for this dataset of 0.1 mg-N/L. All three had values of 0.2 mg-N/L and were located at depths ranging from 7 to 14 meters. For nitrate, 23 samples were less than the detection limit (0.1 mg-N/L) and the maximum nitrate concentration measured was 1.1 mg-N/L. Twelve measurements of Total Kjeldahl Nitrogen (TKN), which includes the organic and ammonia species of nitrogen, were less than the detection limit (0.1 mg-N/L) and the maximum TKN concentration observed was 2.0 mg-N/L. For ammonium, 55 out of 90 measurements were less than the detection limit (0.1 mg-N/L) and 35 samples ranged from 0.1 mg-N/L to 1.2 mg-N/L. pH ranged from 7.3 to 8.8. The summary table lists chlorophyll *a* concentrations ranging from <1 µg/L to 19 µg/L with an average of

8 μg/L. The graphs displaying the depth profile data for Peck Road Park Lake show that dissolved oxygen typically declines to 0 mg/L during the summer months at depths greater than 5 meters. At depths less than 5 meters, dissolved oxygen concentrations were typically around 7 mg/L during the summer months. The study reported a "fishy" smell around the lake.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Peck Road Park Lake states that dissolved oxygen (DO) was not supporting the aquatic life use: 195 measurements of DO were collected in the lake with concentrations ranging from 0.2 mg/L to 15.2 mg/L. The accompanying database does not contain the raw data associated with these measurements, so depth, temperature, date, and time cannot be established. The summary table also lists the odor impairment as not supporting both contact and non-contact recreation uses.

On June 17, 2008, the Regional Board sampled water quality from the middle of each lobe of Peck Road Park Lake (shoreline sampling is not discussed in this section but is described in Appendix G, Monitoring Data). Ammonia concentrations ranged from less than the detection limit (0.1 mg-N/L) to 0.437 mg-N/L. TKN ranged from 1.2 mg-N/L to 2.08 mg-N/L. Nitrite concentrations were less than the detection limit (0.1 mg-N/L) in both basins; nitrate was less than the detection limit (0.1 mg-N/L) in the southern basin and 0.24 mg-N/L in the northern basin. Orthophosphate and total phosphate measurements in both basins were less than the detection limits (0.4 mg-P/L and 0.5 mg-P/L, respectively). Field data were collected in both basins at depths ranging from the water surface to 2.5 meters. Temperature varied by approximately 1 ºC in the south basin and approximately 4 ºC in the north basin over the sampling depth. Dissolved oxygen in the lake was greater than 17 mg/L at all depths except in the northern basin at a depth of 2.5 meters where the concentration was 3 mg/L. pH measurements in the lake ranged from 8.0 to 9.4, although the meter was not calibrated due to equipment malfunction. Chlorophyll *a* measurements in the lake ranged from 4.0 μg/L to 11.4 μg/L. The field notes for this event did not mention odor.

Four sites were sampled by the Regional Board on December 11, 2008; samples were collected from the surface at each site. Measurements of TKN, nitrite, orthophosphate, and total phosphate were less than the detection limits at each site (1.0 mg-N/L, 0.1 mg-N/L, 0.4 mg-P/L, and 0.5 mg-P/L, respectively). Ammonia concentrations ranged from 0.209 mg-N/L to 0.273 mg-N/L; nitrate ranged from 0.162 mg-N/L to 0.287 mg-N/L. Chlorophyll *a* ranged from 1.8 μg/L to 4.0 μg/L. Field data were collected from the surface to 2.0 meters. DO ranged from 2.21 mg/L to 6.20 mg/L (field notes indicate that the meter was not calibrated prior to sampling and field team questioned accuracy of these readings). pH ranged from 7.47 to 7.81.

Water quality monitoring was also conducted by the USEPA and Regional Board on August 5, 2009 in both basins. Ammonia, TKN, nitrate, and nitrite were less than the detection limits (0.03 mg-N/L, 0.456 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively). Orthophosphate ranged from 0.0112 mg-P/L to 0.0135 mg-P/L, and total phosphorus ranged from 0.022 mg-P/L to 0.116 mg-P/L. Chlorophyll *a* ranged from 5.3 μg/L to 8.0 μg/L. DO in the epilimnion was greater than 8 mg/L in both basins. pH ranged from 8.17 to 8.71 in the epilimnion. Field notes report "an unappealing smell that is hard to describe in both the channel connecting the northern and southern lobes and in the northern lobe of Peck Road Park Lake. This smell could possibly be coming from the water or from the industry buildings which are close to the shore of the northern lobe of the lake."

On September 30, 2010, additional sampling was conducted at the mid-lake sites. Ammonia concentrations were below the detection limit of 0.03 mg-N/L. Nitrite ranged from 0.041 to 0.043 mg-N/L, and nitrate was below the detection limit of 0.01 mg-N/L. TKN ranged from 0.562 to 0.634 mg-N/L. Orthophosphate and total phosphorus ranged from 0.02 mg-P/L to 0.04 mg-P/L. Chlorophyll *a* ranged from 6.7 μg/L to 13.4 μg/L. During this event, two continuous monitoring probes were deployed over a 24-hour period. At an average depth of 0.6 meters, DO concentrations during the 24-hour period ranged from 8.6 mg/L to 10.1 mg/L. pH ranged from about 8.5 to 8.8. On September 30, 2010, DO measurements collected from the surface of the lake ranged from 8.5 mg/L to 10.9 mg/L. At 2 meters above the bottom, DO ranged from 0.2 to 4.0 mg/L.

In summary, Peck Road Park Lake has been sampled several times over the past two decades. Slight exceedances of the pH target have been observed in the lake and may be due to natural conditions. DO levels in the epilimnion are typically greater than 7 mg/L and impairment due to low DO is not evident in either the historic or recent sampling events (DO levels do approach zero in the deeper waters but no exceedances have been observed relative to the target depths). Readings collected in December 2008 were collected with an uncalibrated meter. Chlorophyll *a* concentrations are relatively low and no measurements greater than 19 µg/L (historic data) have been reported. The maximum chlorophyll *a* concentration measured recently is 13.4 µg/L and the average concentration is 6.2 µg/L. It does not appear, based on these data, that excessive nutrient loading is causing an impairment. It is unlikely that the source of the odor reported at Peck Road Park Lake is due to elevated nutrient and algal biomass levels. They are likely associated with the trash impairment addressed in Section 4.8. The nutrient TMDLs for Peck Road Park Lake presented in Section 4.2.6 are based on existing conditions.

4.2.4 Source Assessment

The source assessment for Peck Road Park Lake includes load estimates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading) and atmospheric deposition (Appendix E, Atmospheric Deposition) (Table 4-7). Watershed loading accounts for 55.5 percent of the total nitrogen load and 80.2 percent of the total phosphorus load. Diversions from the San Gabriel River to Peck Road Park Lake (via the eastern subwatershed) contribute 41.1 percent of the total nitrogen load and 15.3 percent of the total phosphorus load. All existing loads to Peck Road Park Lake are summarized in Table 4-7.

Table 4-7. Summary of Average Annual Flows and Nutrient Loading to Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Eastern	Arcadia	MS4 Stormwater[1]	206	383 (2.0)	2,320 (1.2)
Eastern	Bradbury	MS4 Stormwater[1]	291	497 (2.6)	3,223 (1.7)
Eastern	Caltrans	State Highway Stormwater[1]	99.9	158 (0.8)	1,165 (0.6)
Eastern	Duarte	MS4 Stormwater[1]	850	1,540 (8.0)	9,616 (5.1)
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	34.9	55.1 (0.3)	432 (0.2)
Eastern	Irwindale	MS4 Stormwater[1]	325	496 (2.6)	3,487 (1.9)
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	20.6	32.5 (0.2)	255 (0.1)
Eastern	County of Los Angeles	MS4 Stormwater[1]	488	924 (4.8)	5,532 (2.9)
Eastern	Monrovia	MS4 Stormwater[1]	3,527	6,243 (32.3)	38,736 (20.7)

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	141	223 (1.2)	1,748 (0.9)
Eastern	Angeles National Forest	Stormwater[1]	309	92.5 (0.5)	2,692 (1.4)
Diversion	Los Angeles County Department of Public Works	Water Diversion	8,737	2,960 (15.3)	76,970 (41.1)
Near Lake	Arcadia	MS4 Stormwater[1]	102	158 (0.8)	1,115 (0.6)
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	14.8	23.4 (0.1)	183 (0.1)
Near Lake	El Monte	MS4 Stormwater[1]	52.8	96.2 (0.5)	602 (0.3)
Near Lake	Irwindale	MS4 Stormwater[1]	17.8	28.2 (0.1)	207 (0.1)
Near Lake	County of Los Angeles	MS4 Stormwater[1]	68.1	129 (0.7)	773 (0.4)
Near Lake	Monrovia	MS4 Stormwater[1]	38.0	60.4 (0.3)	415 (0.2)
Western	Arcadia	MS4 Stormwater[1]	1,493	2,840 (14.7)	16,334 (8.7)
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	328	517 (2.7)	4,058 (2.2)
Western	Caltrans	State Highway Stormwater[1]	21.6	34.2 (0.2)	251 (0.1)
Western	County of Los Angeles	MS4 Stormwate[1]r	248	467 (2.4)	2,818 (1.5)
Western	Monrovia	MS4 Stormwater[1]	275	425 (2.2)	2,678 (1.4)
Western	Sierra Madre	MS4 Stormwater[1]	406	695 (3.6)	4,254 (2.3)
Western	Angeles National Forest	Stormwater[1]	802	240 (1.2)	6,981 (3.7)
Lake Surface		Atmospheric Deposition[3]	139	NA	69 (0.04)
Total			**19,034**	**19,319**	**186,914**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits.

[3] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

4.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions.

To simulate the impacts of nutrient loading on Peck Road Park Lake, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Peck Road Park Lake, the following inputs apply: surface area of 87.4 acres, average depth of 30 ft, and volume of 2,622 ac-ft. Based on the phosphorus turnover ratio for this lake (Walker, 1987), the summer averaging period is appropriate (i.e., loads delivered from May through September are input to the model rather than annual loads). Without adjusting calibration factors in the model (calibration factors on net sedimentation rates set to 1), the average annual loads presented in Section 4.2.4 yield total nitrogen, total phosphorus, and chlorophyll *a* concentrations of 1.19 mg-N/L, 0.077 mg-P/L, and 12.8 µg/L, respectively.

Average conditions for Peck Road Park Lake with regard to algal stimulation are assessed based on measurements collected between the surface and twice the observed Secchi depth. Average annual observed total nitrogen, total phosphorus, and chlorophyll *a* concentrations over the assessment depth (4.2 meters) are 0.76 mg-N/L, 0.05 mg-P/L, and approximately 6 µg/L, respectively, assuming measurements less than detection are equal to half the detection limit. Even with simulated nitrogen and phosphorus concentrations 2 to 3 times higher than those observed in the lake (i.e., calibration factors left at 1), simulated chlorophyll *a* (12.8 µg/L) remains below the target concentration of 20 µg/L. Calibrating the NNE BATHTUB Tool would result in lower simulated concentrations of nitrogen, phosphorus, and chlorophyll *a*. Thus, the NNE BATHTUB Tool indicates that Peck Road Park Lake is not directly impaired by elevated nutrient loads or excessive algal growth. (Since the calibration factor on the net phosphorus sedimentation rate would have been adjusted even lower during calibration, the method described in Appendix A (Nutrient TMDL Development) was used to estimate internal loading. Based on the inflow concentrations, in-lake concentrations, and residence time of this system, the internal loading calculation resulted in a negative number which indicates that settling is more dominant than resuspension, and internal loading of phosphorus is insignificant relative to other sources.)

Based on historic and recent monitoring data, Peck Road Park Lake is not impaired by low DO or excessive nutrient loading (Section 4.2.3). Though odor has been noted as a problem at the lake, it is likely not due to eutrophication as no algal blooms have been observed in the lake and chlorophyll *a* concentrations are relatively low. To protect Peck Road Park Lake from degradation, nutrient loading should remain at or below existing levels as an antidegradation measure to ensure future loading does not increase the chlorophyll *a* concentration.

4.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Based on observed levels of chlorophyll *a* and DO in Peck Road Park Lake, existing levels of nitrogen and phosphorus loading are resulting in attainment of both the chlorophyll *a* and DO targets. Monitoring data indicate that the average in-lake total nitrogen concentration is 0.76 mg-N/L (Appendix G, Monitoring Data). Because the majority of in-lake phosphorous samples have been less than the detection limits for the analytical laboratory, the phosphorus target concentration is based on an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10. This ratio was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 0.76 mg-N/L summer average (May – September) and annual average
- 0.076 mg-P/L summer average (May – September) and annual average

To prevent degradation of this waterbody, nutrient TMDLs will be allocated based on existing loading. These TMDLs are broken down into wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation. Note that the MOS is zero because these TMDLs are equal to the existing load.

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load is equal to the existing load and is divided among WLAs and LAs. The resulting TMDL equation for total nitrogen is then:

186,914 lb-N/yr = 186,845 lb-N/yr + 69.3 lb-N/yr + 0 lb-N/yr

For total phosphorus, the allocatable load is equal to the existing load and allocated to WLAs only: LAs are zero as explained in Section 4.2.6.2. The resulting TMDL equation for total phosphorous is then:

19,319 lb-P/yr = 19,319 lb-P/yr + 0 lb-P/yr + 0 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined for the lake based on recent and historical monitoring data (see Section 4.2.5). These in-lake concentrations reflect internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

4.2.6.1 Wasteload Allocations

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm. The Peck Road Park Lake watershed drains to a series of storm drains prior to discharging to the lake. Therefore, all nutrient loads associated with the surrounding drainage area are assigned wasteload allocations (WLAs). The Caltrans areas and facilities that operate under a general industrial stormwater permit also receive WLAs.

Relevant permit numbers are

- County of Los Angeles (including the cities of Arcadia, Bradbury, Duarte, Irwindale, Monrovia, and Sierra Madre): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

WLAs are presented in Table 4-8. Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available WLAs. These TMDLs establish WLAs at their point of discharge. Note that WLAs are equal to existing loading rates because no reductions in loading are required. These loading values (in pounds per year) represent the TMDLs wasteload allocations (Table 4-8). All responsible jurisdictions must meet the WLAs as a mass load except for storm water permittees under the general industrial stormwater permit and the general NPDES permit for the Colorado Well Aquifer (Order No. R4-2003-0108, CAG994005), that are receiving concentration-based WLAs. In Table 4-8 below, permittees under these general permits must meet the concentration values to achieve compliance with the WLAs. The phosphorous and nitrogen WLA concentrations are based on the average targeted concentrations of nutrients (allowable load divided by inflow volume): 0.37 mg-P/L and 3.61 mg-N/L. Each wasteload allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 4-8. Wasteload Allocations of Phosphorus and Nitrogen Loading to Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation Total Phosphorus (lb-P/yr)[4]	Wasteload Allocation Total Nitrogen (lb-N/yr)[4]
Eastern	Arcadia	MS4 Stormwater[1]	383	2,320
Eastern	Bradbury	MS4 Stormwater[1]	497	3,223
Eastern	Caltrans	State Highway Stormwater[1]	158	1,165
Eastern	Duarte	MS4 Stormwater[1]	1,540	9,616
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	55.1 (0.37 mg/L P)[2]	432 (3.61 mg/L N)[2]
Eastern	General Groundwater Discharge Permittees[3]	Groundwater Discharge	0.37 mg/L P[3]	3.61 mg/L N[3]
Eastern	Irwindale	MS4 Stormwater[1]	496	3,487
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)[3]	General Industrial Stormwater[1]	32.5 (0.37 mg/L P)[2]	255 (3.61 mg/L N)[2]
Eastern	County of Los Angeles	MS4 Stormwater[1]	924	5,532
Eastern	Monrovia	MS4 Stormwater[1]	6.243	38,736
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)[3]	General Industrial Stormwater[1]	223	1,748
Eastern	Angeles National Forest	Stormwater[1]	92.5	2,692
Diversion	Los Angeles County Department of Public Works	Water Diversion	2,960	76,970
Near Lake	Arcadia	MS4 Stormwater[1]	158	1,115
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)[3]	General Industrial Stormwater[1]	23.4 (0.37 mg/L P)[2]	183 (3.61 mg/L N)[2]
Near Lake	El Monte	MS4 Stormwater[1]	96.2	602
Near Lake	Irwindale	MS4 Stormwater[1]	28.2	207
Near Lake	County of Los Angeles	MS4 Stormwater[1]	129	773
Near Lake	Monrovia	MS4 Stormwater[1]	60.4	415
Western	Arcadia	MS4 Stormwater[1]	2,840	16,334
Western	General Industrial Stormwater Permittees (in the city of Arcadia)[3]	General Industrial Stormwater[1]	517 (0.37 mg/L P)[2]	4,058 (3.61 mg/L N)[2]
Western	Caltrans	State Highway Stormwater[1]	34.2	251

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation Total Phosphorus (lb-P/yr)[4]	Wasteload Allocation Total Nitrogen (lb-N/yr)[4]
Western	County of Los Angeles	MS4 Stormwater[1]	467	2,818
Western	Monrovia	MS4 Stormwater[1]	425	2,678
Western	Sierra Madre	MS4 Stormwater[1]	695	4,254
Western	Angeles National Forest	Stormwater[1]	240	6,981
Total			**19,319**	**186,845**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations (see footnote #3).

[3] For these responsible jurisdictions, the concentration-based WLA will be used to evaluate compliance.

[4] Each wasteload allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

4.2.6.2 Load Allocations

Atmospheric deposition of nitrogen to the lake surface is a nonpoint source and is assigned a load allocation (LA). Table 4-9 presents the LAs for atmospheric deposition, which are equivalent to existing loading rates because no reductions in loading are required. Atmospheric deposition does not contribute significant loads of phosphorus (Appendix E, Atmospheric Deposition). These loading values (in pounds per year) represent the TMDL load allocations (Table 4-9). Each load allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 4-9. Load Allocations of Nitrogen Loading to Peck Road Park Lake

Input	Load Allocation Total Phosphorus (lb-P/yr)[1]	Load Allocation Total Nitrogen (lb-N/yr)[1]
Atmospheric Deposition (to the lake surface)[2]	NA	69
Total	**NA**	**69**

[1] Each load allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

4.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This lake is currently achieving the in-lake chlorophyll *a* target and TMDLs are being established at the existing loads. This conservative anti-degradation measure is the implicit margin of safety for these TMDLs.

4.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are based on existing conditions as an anti-degradation measure since nitrogen and phosphorus levels are currently achieving the chlorophyll *a* target level. These TMDLs therefore protect for critical conditions.

4.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of nutrient loading to Peck Road Park Lake occurs during wet weather events that deliver pollutant loads from both the surrounding watershed and diversions from the San Gabriel River, the daily maximum allowable loads of nitrogen and phosphorus are calculated from the maximum daily storm flow rate (estimated from the 99^{th} percentile flow) to the Lake multiplied by the average allowable concentrations consistent with achieving the long-term loading targets. These maximum loads must be met each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the Peck Road Park Lake watershed, but there is a gage downstream. USGS Station 11101250, Rio Hondo above Whittier Narrows Dam, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99^{th} percentile flow for Rio Hondo (952 cfs) (Wolock, 2003). To estimate the peak flow to Peck Road Park Lake from the surrounding watershed, the 99^{th} percentile flow for Rio Hondo was scaled down by the ratio of drainage areas (23,564 acres/58,368 acres; Peck Road Park Lake watershed area/Rio Hondo watershed area at the gage). The resulting peak flow estimate for Peck Road Park Lake is 384 cfs. The 99^{th} percentile diverted flow from the San Gabriel River to Peck Road Park Lake is 328 cfs. Therefore, the total peak daily flow rate is 712 cfs.

The average allowable concentrations of phosphorus and nitrogen were calculated from the allowable loads (19,319 lb-P/yr and 186,914 lb-N/yr, respectively) divided by the total volume reaching the lake from runoff and diversions (19,034 ac-ft) (Table 4-7). Multiplying the average allowable concentrations (0.37 mg-P/L for phosphorous and 3.61 mg-N/L for nitrogen) by the 99^{th} percentile peak daily flow (712 cfs) yields the daily maximum load associated with wet weather runoff. The wet weather runoff daily maximum allowable loads of phosphorus and nitrogen for Peck Road Park Lake are 1,433 lb-P/d and 13,868 lb-N/d, respectively. These loads are associated with the MS4 stormwater permittees and the water diversion. As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

4.2.6.6 Future Growth

Much of the Peck Road Park Lake watershed remains in forested and other undisturbed land uses. As development occurs in this watershed, best management practices (BMPs) will be required such that loading rates are consistent with the allocations established by these TMDLs. Therefore, no load allocation has been set aside for future growth. It is unlikely that any dischargers of significant nutrient loading will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.3 Lead Impairment

Peck Road Park Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), USEPA and local agencies collected 30 additional samples (12 wet weather) between December 2008 and September 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 30 samples (Appendix G, Monitoring Data). USEPA also collected two sediment samples during September 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Peck Road Park Lake meets lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Peck Road Park Lake not be identified as impaired by lead in California’s next 303(d) list.

4.4 PCB Impairment

Polychlorinated biphenyls (PCBs) consist of a family of many related congeners. The individual congeners are often referred to by their “BZ” number. Environmental analyses may address individual congeners, homologs (groups of congeners with the same number of chlorine atoms), equivalent concentrations of the commercial mixtures of PCBs known by the trade name Aroclors, or total PCBs. The environmental measurements and targets described in this section are in terms of total PCBs, defined as the “sum of all congener or isomer or homolog or Aroclor analyses” (CTR, 40 CFR 131.38(b)(1) footnote v).

The PCB impairment of Peck Road Park Lake affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. PCBs are no longer in production. While some loading of PCBs continues to occur in watershed runoff, the primary source of PCBs in the water column and aquatic life in Peck Road Park Lake is from historic loads stored in the lake sediments. Like other organochlorine compounds, PCBs accumulate in aquatic organisms and biomagnify in the food chain. As a result, low environmental exposure concentrations can result in unacceptable levels in higher trophic level fish in the lake.

4.4.1 Problem Statement

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of PCBs potentially impair the REC1, REC2, WARM, WILD, and MUN uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impairing sport fishing recreational uses.

4.4.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of PCBs in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets used for PCBs are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for PCBs in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0005 mg/L, or 0.5 μg/L, total PCBs in water. The Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this report is designated WARM, at a minimum, and must meet this requirement. A chronic criterion for the sum of PCB compounds in freshwater systems to protect aquatic life is included in the CTR as 0.014 μg/L (USEPA, 2000a). The CTR also provides a human health-based water quality criterion for the consumption of both water and organisms and organisms only of 0.00017 μg/L (0.17 ng/L). The human health criterion of 0.17 ng/L is the most restrictive applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) for total PCBs in sediment is 59.8 μg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The existing sediment PCB concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for PCBs defined by OEHHA (2008) is 3.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For total PCBs, the corresponding sediment concentration target determined using the BSAF is 1.29 µg/kg dry weight, as described in detail in Section 4.4.5. All applicable targets are shown below in Table 4-10. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 4-10. PCB Targets Applicable to Peck Road Park Lake

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	3.6
Sediment (µg/kg dry weight)	Consensus-based TEC	59.8
Sediment (µg/kg dry weight)	BSAF-derived target	1.29
Water (ng/L)	CTR	0.17

Note: Shaded cells represent the selected targets for this TMDL.

4.4.3 Summary of Monitoring Data

This section summarizes the monitoring data for Peck Road Park Lake related to the PCB impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

For PCBs, as well as other organochlorine compounds, sample analyses include both a detection limit and a method reporting limit. For example, a typical detection limit for total PCBs in sediment reported by UCLA is 0.53 µg/kg dry weight, while the reporting limit is 15 µg/kg dry weight.

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 at five locations (six samples) and again in the fall of 2008 at two locations (three samples) in Peck Road Park Lake and its tributaries. Three of the samples collected during the summer were below detectable levels (1.5 – 1.58 ng/L; which is greater than the ambient water quality criterion of 0.17 ng/L), while two samples collected in the summer of 2008 and both samples collected in the fall of 2008 had detections of PCB congeners, but at levels too low to be quantified (at reporting limits of 15 – 16.67 ng/L). As the detection limit is greater than the CTR target these samples are greater than the ambient water quality criterion of 0.17 ng/L.

Additional water column sampling was conducted by the Regional Board on December 11, 2008 at four in-lake locations in Peck Road Park Lake. All four sites sampled were below detectable concentrations of PCBs (1 ng/L; the detection limit is above the water quality criterion). A summary of the water column data is shown in Table 4-11.

Table 4-11. Summary of Water Column Samples for PCBs in Peck Road Park Lake

Station	Average Water Concentration (ng/L)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
Sawpit Wash	[8.64]	2	2	2
Santa Anita Wash	[4.31]	3	2	2
North Basin Outfall	(0.76)	2	0	0
North Basin	(0.60)	2	0	0
South Basin	[2.30]	2	1	1
South Basin East	(0.50)	1	0	0

Station	Average Water Concentration (ng/L)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
South Basin West Side	(0.50)	1	0	0
In-Lake Average[2]	[2.37]			
Water Column Target	0.17			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages (excludes the tributary samples).

Concentrations of PCBs on suspended sediment were also analyzed at two in-lake stations during the summer and fall of 2008 as part of the UCLA study; one location was analyzed during the summer and two during the fall. During the summer event, PCB congener BZ-110 was detected below reporting limits (51.35 µg/kg dry weight), and the fall sampling detected congeners, including BZ-138 and BZ-180, but each was below reporting limits (23.63 µg/kg to 144.23 µg/kg dry weight).

Porewater was sampled as part of the UCLA study in the summer and fall of 2008. During the summer event, two of the four PCB samples were less than the detection limit of 15 ng/L, while the other two samples had detected, but not reportable concentrations (<150 ng/L). The three sites sampled for porewater during the fall of 2008 were all below the detection limit of 15 ng/L for total PCBs. Three porewater suspended sediment samples collected in the summer of 2008 were below reportable levels for total PCBs (22.55 µg/kg to 66.03 µg/kg dry weight), and one sample was below the detection limit of 9.25 µg/kg dry weight.

Suspended solids (TSS) from Peck Road Park Lake were collected in the summer and fall of 2008. In summer of 2008, only one station had enough suspended matter to perform the analysis. None of the pesticides were detected in the sample (detection limit of 5.14 µg/kg dry weight). PCB-110 was detected, but not within reportable limits (reporting limit of 51.35 µg/kg dry weight). In fall 2008, samples were analyzed at two stations with detection limits ranging from 2.36 µg/kg to 20.41 µg/kg dry weight. In one sample, PCB congener BZ-138 was detected, but not within reportable limits (reporting limit of 23.63 µg/kg dry weight), while BZ-180 was detected in the other sample, but below reporting limits (reporting limit of 144.23 µg/kg dry weight).

UCLA also collected bed sediment samples at four locations in Peck Road Park Lake in summer and fall 2008. Samples related to tributaries were collected in the lake near the tributary outfall. Two of the nine lake sediment samples collected during 2008 had reportable levels of PCBs, with a maximum of 276 µg/kg dry weight (in excess of the consensus-based TEC value of 59.8 µg/kg dry weight). Four in-lake locations were sampled by USEPA and the county of Los Angeles on November 16, 2009; total PCB concentrations ranged from 1.0 µg/kg to 23.3 µg/kg dry weight. All lake stations were averaged to estimate an exposure concentration of 12.28 µg/kg dry weight total PCBs (with non-detects included at one-half the detection limit for each sample). Stations located near outfalls, are taken as an estimate of the concentrations on incoming sediment. A summary of the sediment data is shown in Table 4-12.

Fish tissue concentrations of total PCBs from Peck Road Park Lake have been analyzed in largemouth bass (SWAMP and TSMP) by composite samples consisting of filet tissue from five fish. Total PCB concentrations in the fish tissue resulted in concentrations of 22.7 and 55.3 ppb, in two largemouth bass composite samples taken during the summer of 2007, while an April 2010 composite resulted in a concentration of 25.3 ppb total PCBs, both in excess of the fish tissue target for total PCBs (FCG of 3.6 ppb). Earlier analyses for PCB Aroclor analyzed from 1986-1992 resulted in nondetectable concentrations (at an unreported detection limit) in all four largemouth bass samples. Considering only data collected in the past 10 years, the average concentration of PCBs in largemouth bass was 34.4 ppb. This average is based on the three largemouth bass composite samples collected in 2007 and 2010 with an average lipid fraction of 0.54 percent. Recent fish-tissue data for Peck Road Park Lake are summarized in Table 4-13. Bottom-feeding fish data (e.g., carp) are not available for Peck Road Park Lake.

Table 4-12. Summary of Sediment Samples for PCBs in Peck Road Park Lake, 2008-2009

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
Near Sawpit Wash	5.89	1	1	0
Near Santa Anita Wash	49.52	3	2	0
North Basin	7.12	4	3	1
South Basin	[5.07]	3	2	2
North Inlet	[1.00]	1	1	1
South Inlet	[5.10]	1	1	1
In-Lake Average[2]	12.28			
Influent Average	15.38			
Consensus-based TEC	59.8			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.
[2] Overall average is the average of individual station averages.

Table 4-13. Summary of Recent Fish Tissue Samples for PCBs in Peck Road Park Lake

Sample Date	Fish Species	Total PCBs (ppb wet weight)[1]
6 June 2007	Largemouth Bass	55.3
6 June 2007	Largemouth Bass	22.7
19 April 2010	Largemouth Bass	25.3
2007 – 2010 Average		34.4
FCG		3.6

[1] Composite samples of filet from five individuals.

In sum, recent fish tissue samples collected from Peck Road Park Lake are an order of magnitude greater than the OEHHA fish consumption guidelines for total PCBs. Measured concentrations in sediment are below the consensus-based TEC. Concentrations in water have not exceeded method reporting limits; however, several recent samples were above detection limits that themselves exceed the CTR criterion.

4.4.4 Source Assessment

PCBs in Peck Road Park Lake are primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed PCB concentrations on sediment near inflows to the lake.

Watershed loads of PCBs may arise from spills from industrial and commercial uses, improper disposal, and atmospheric deposition. Industrial and commercial spills will tend to be associated with specific land areas, such as older industrial districts, junk yards, and transformer substations. Improper disposal could have occurred at various locations (indeed, waste PCB oils were sometimes used for dust control on dirt roads in the 1950s). Atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated PCB load within the watershed at this time. Therefore, an average concentration of sediment is applied to all contributing areas. The average concentration of PCBs on incoming sediment was estimated to be 15.38 µg/kg dry weight and the estimated annual sediment load to Peck Road Park Lake is 990.3 tons/yr, including sediment delivered through the water diversion (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load of PCBs is approximately 13.8 g/yr. Table 4-14 shows the annual PCB load estimated from each jurisdiction.

Table 4-14. Total PCB Loads Estimated for Each Jurisdiction and Subwatershed in the Peck Road Park Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total PCB Load (g/yr)	Percent of Total Load
Eastern	Arcadia	MS4 Stormwater[1]	12.1	0.17	1.22%
Eastern	Bradbury	MS4 Stormwater[1]	44.4	0.62	4.48%
Eastern	Caltrans	State Highway Stormwater[1]	9.6	0.13	0.96%
Eastern	Duarte	MS4 Stormwater[1]	57.2	0.80	5.78%
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	0.8	0.01	0.08%
Eastern	Irwindale	MS4 Stormwater[1]	23.3	0.33	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.6	0.02	0.16%
Eastern	County of Los Angeles	MS4 Stormwater[1]	28.6	0.40	2.89%
Eastern	Monrovia	MS4 Stormwater[1]	200	2.80	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	16.3	0.23	1.65%
Eastern	Angeles National Forest	Stormwater[1]	12.1	0.17	1.22%
Diversion	Los Angeles County Department of Public Works	Water Diversion	379	5.29	38.31%
Near Lake	Arcadia	MS4 Stormwater[1]	7.6	0.11	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.7	0.02	0.17%

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total PCB Load (g/yr)	Percent of Total Load
Near Lake	El Monte	MS4 Stormwater[1]	3.5	0.05	0.36%
Near Lake	Irwindale	MS4 Stormwater[1]	1.7	0.02	0.17%
Near Lake	County of Los Angeles	MS4 Stormwater[1]	4.0	0.06	0.41%
Near Lake	Monrovia	MS4 Stormwater[1]	2.6	0.04	0.26%
Western	Arcadia	MS4 Stormwater[1]	68.1	0.95	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	37.8	0.53	3.82%
Western	Caltrans	State Highway Stormwater[1]	2.1	0.03	0.21%
Western	County of Los Angeles	MS4 Stormwater[1]	14.7	0.21	1.49%
Western	Monrovia	MS4 Stormwater[1]	9.3	0.13	0.94%
Western	Sierra Madre	MS4 Stormwater[1]	19.9	0.28	2.01%
Western	Angeles National Forest	Stormwater[1]	31.4	0.44	3.18%
Total Load from Watershed			**990.3**	**13.7**	**100%**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of PCBs directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

4.4.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of PCBs into Peck Road Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of PCBs in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. PCBs are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of PCBs will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data in Peck Road Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment PCB concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target to achieve FCGs is calculated based on biota-sediment bioaccumulation (a BSAF approach), using the ratio of the FCG to existing fish tissue concentrations of 3.6/34.4 = 0.105. This ratio is applied to the observed in-lake sediment concentration of 12.28 μg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 1.29 µg/kg dry weight. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of PCBs are likely to have declined steadily since the cessation of production and use of the chemical. The resulting fish-tissue based concentration of PCBs in the sediment of Peck Road Park Lake is shown in Table 4-15.

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 59.8 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.17 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

Table 4-15. Fish Tissue-Based Total PCB Concentration Targets for Sediment in Peck Road Park Lake

Total PCB Concentration	Sediment (µg/kg dry weight)
Existing	12.28
BSAF-derived target	1.29
Required Reduction	89.5%

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 1,005 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 13.8 g/yr, or 1.4 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of PCBs in Peck Road Park Lake is primarily due to the storage of historic loads of PCBs in the lake sediment.

4.4.6 TMDL Summary

Because PCB impairment in Peck Road Park Lake is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a direct calculation of loading capacity expressed as mass per unit time. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake.

The PCB TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 1.29 µg/kg dry weight total PCBs. The wasteload allocations and load allocations are also equal to 1.29 µg/kg dry weight total PCBs in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

4.4.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for total PCBs ("Alternative WLAs if the Fish Tissue Target is Met") described in Section 4.4.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 4.4.6.1.1 if the conditions described in Section 4.4.6.1.2 are met.

4.4.6.1.1 Wasteload Allocations

The entire watershed of Peck Road Park Lake is contained in MS4 jurisdictions, and watershed loads are therefore assigned WLAs. The Caltrans areas and facilities that operate under a general industrial stormwater permit also receive WLAs.

Relevant permit numbers are

- County of Los Angeles (including the cities of Arcadia, Bradbury, Duarte, Irwindale, Monrovia, and Sierra Madre): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

PCBs in water flowing into Peck Road Park Lake are below detection limits, and most PCB load is expected to move in association with sediment. Therefore, no separate wasteload allocation or reduction is explicitly assigned to the Colorado Well Aquifer (Order No. R4-2003-0108, CAG994005) as it is not expected to deliver sediment loads. The suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for PCBs in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved PCBs and PCBs associated with suspended sediment. The existing concentration of sediment entering the lake is 15.38 µg/kg dry weight. Therefore, a reduction of 91.6 percent [(15.38 – 1.29)/ 15.38*100] is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 4-16 and each wasteload allocation must be met at the point of discharge.

Table 4-16. Wasteload Allocations for Total PCBs in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	1.29	0.17
Eastern	Bradbury	MS4 Stormwater[1]	1.29	0.17
Eastern	Caltrans	State Highway Stormwater[1]	1.29	0.17
Eastern	Duarte	MS4 Stormwater[1]	1.29	0.17
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	1.29	0.17
Eastern	Irwindale	MS4 Stormwater[1]	1.29	0.17

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.29	0.17
Eastern	County of Los Angeles	MS4 Stormwater[1]	1.29	0.17
Eastern	Monrovia	MS4 Stormwater[1]	1.29	0.17
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	1.29	0.17
Eastern	Angeles National Forest	Stormwater[1]	1.29	0.17
Diversion	Los Angeles County Department of Public Works	Water Diversion	1.29	0.17
Near Lake	Arcadia	MS4 Stormwater[1]	1.29	0.17
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.29	0.17
Near Lake	El Monte	MS4 Stormwater[1]	1.29	0.17
Near Lake	Irwindale	MS4 Stormwater[1]	1.29	0.17
Near Lake	County of Los Angeles	MS4 Stormwater[1]	1.29	0.17
Near Lake	Monrovia	MS4 Stormwater[1]	1.29	0.17
Western	Arcadia	MS4 Stormwater[1]	1.29	0.17
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.29	0.17
Western	Caltrans	State Highway Stormwater[1]	1.29	0.17
Western	County of Los Angeles	MS4 Stormwater[1]	1.29	0.17
Western	Monrovia	MS4 Stormwater[1]	1.29	0.17
Western	Sierra Madre	MS4 Stormwater[1]	1.29	0.17
Western	Angeles National Forest	Stormwater[1]	1.29	0.17

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.4.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 4-16 will be superseded, and the wasteload allocations in Table 4-17 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A

demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,

2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 4-17, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 4-17. Alternative Wasteload Allocations for Total PCBs in Peck Road Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	59.8	0.17
Eastern	Bradbury	MS4 Stormwater[1]	59.8	0.17
Eastern	Caltrans	State Highway Stormwater[1]	59.8	0.17
Eastern	Duarte	MS4 Stormwater[1]	59.8	0.17
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	59.8	0.17
Eastern	Irwindale	MS4 Stormwater[1]	59.8	0.17
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	59.8	0.17
Eastern	County of Los Angeles	MS4 Stormwater[1]	59.8	0.17
Eastern	Monrovia	MS4 Stormwater[1]	59.8	0.17
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	59.8	0.17
Eastern	Angeles National Forest	Stormwater[1]	59.8	0.17
Diversion	Los Angeles County Department of Public Works	Water Diversion	59.8	0.17
Near Lake	Arcadia	MS4 Stormwater[1]	59.8	0.17
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	59.8	0.17
Near Lake	El Monte	MS4 Stormwater[1]	59.8	0.17
Near Lake	Irwindale	MS4 Stormwater[1]	59.8	0.17
Near Lake	County of Los Angeles	MS4 Stormwater[1]	59.8	0.17
Near Lake	Monrovia	MS4 Stormwater[1]	59.8	0.17
Western	Arcadia	MS4 Stormwater[1]	59.8	0.17
Western	General Industrial Stormwater Permittees (in	General Industrial Stormwater[1]	59.8	0.17

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
	the city of Arcadia)			
Western	Caltrans	State Highway Stormwater[1]	59.8	0.17
Western	County of Los Angeles	MS4 Stormwater[1]	59.8	0.17
Western	Monrovia	MS4 Stormwater[1]	59.8	0.17
Western	Sierra Madre	MS4 Stormwater[1]	59.8	0.17
Western	Angeles National Forest	Stormwater[1]	59.8	0.17

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.4.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for total PCBs ("Alternative LAs if the Fish Tissue Target is Met") described in Section 4.4.6.2.2. The alternative load allocations will supersede the load allocations in Section 4.4.6.2.1 if the conditions described in Section 4.4.6.2.2 are met.

4.4.6.2.1 Load Allocations

No part of the watershed of Peck Road Park Lake is outside MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to atmospheric deposition of PCBs.

The legacy PCB stored in lake sediment is the major cause of use impairment due to elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (County of Los Angeles) should achieve a PCB concentration of 1.29 µg/kg dry weight in lake bottom sediments (Table 4-18).

Table 4-18. Load Allocations for Total PCBs in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	1.29

4.4.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 4-18 will be superseded, and the load allocations in Table 4-19 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,

2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 4-19, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Table 4-19. Alternative Load Allocations for Total PCBs in Peck Road Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	59.8

4.4.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

4.4.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate PCBs, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

4.4.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the PCB WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Peck Road Park Lake watershed. USGS Station 11101250, on the Rio Hondo River above the Whittier Narrows Dam, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for the Rio Hondo (952 cfs) (Wolock, 2003). To estimate the peak flow to Peck Road Park Lake, the 99th percentile flow for the Rio Hondo was scaled down by the ratio of drainage areas (23,564 acres/58,368 acres; Peck Road Park Lake watershed area/Rio Hondo watershed area at the gage). The resulting peak flow estimate for Peck Road Park Lake is 384 cfs. The 99th percentile diverted flow from the San Gabriel River to Peck Road Park Lake is 328 cfs. Therefore, the total peak daily flow rate is 712 cfs.

The event mean concentration of sediment in stormwater (71.7 mg/L) was calculated from the estimated existing watershed sediment load of 990.3tons/yr (Table 4-14) divided by the stormwater flow volume entering the lake (10,158 ac-ft, Table 4-7). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (712 cfs) yields a daily maximum sediment load from stormwater of 137.7 tons/d. Applying the wasteload allocation concentration of 1.29 ng total PCBs per dry g of sediment yields the stormwater daily maximum allowable load of 0.161 g/d of total PCBs. This load is associated with the MS4 stormwater permittees and the water diversion. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

4.4.6.6 Future Growth

USEPA regulates PCBs under the Toxic Substances Control Act (TSCA), which generally bans the manufacture, use, and distribution in commerce of the chemicals in products at concentrations of 50 parts per million or more, although TSCA allows USEPA to authorize certain uses, such as to rebuild existing electrical transformers during the transformers' useful life. Therefore, no additional allowance is made for future growth in the PCB TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.5 CHLORDANE IMPAIRMENT

Total chlordane consists of a family of related chemicals, including cis- and trans-chlordane, oxychlordane, trans-nonachlor, and cis-nonachlor. Observations and targets discussed in this section all refer to total chlordane. Chlordane was used as a pesticide in field, commercial, and residential uses. Chlordane is no longer in production, but persists in the environment from legacy loads.

The chlordane impairment of Peck Road Park Lake affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. While some loading of chlordane continues to occur in watershed runoff, the primary source of chlordane in the water column and aquatic life in Peck Road Park Lake is from historic loads stored in the lake sediments. Chlordane, like other organochlorine compounds, accumulates in aquatic organisms and biomagnifies in the food chain. As a result, low environmental concentrations can result in unacceptable levels in higher trophic level fish in the lake. The approach for chlordane is similar to that for PCBs.

4.5.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of chlordane are currently impairing the REC1, REC2 and WARM uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impairing sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

4.5.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of chlordane listed in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), for chlordane defined by the Office of Environmental Health Hazard Assessment (OEHHA) for fish consumption. The numeric targets used for chlordane are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for chlordane in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0001 mg/L, or 0.1 μg/L. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Acute and chronic criterion for chlordane in freshwater systems are defined by the California Toxics Rule as 2.4 μg/L and 0.0043 μg/L, respectively (USEPA, 2000a). The CTR also includes human health criteria for the consumption of water and organisms and for the consumption of organisms only as 0.00057 μg/L and 0.00059 μg/L, respectively (USEPA, 2000a). For Peck Road Park Lake, the Regional Board has determined that the appropriate human health criterion is 0.00059 μg/L (0.59 ng/L) as the MUN use is not an existing use and may be removed.

For sediment, the consensus-based sediment quality guidelines provided in Macdonald et al. (2000) for the threshold effects concentration (TEC) for chlordane is 3.24 μg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider “the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans).” The existing sediment chlordane concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for chlordane defined by OEHHA (2008) is 5.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For chlordane, the corresponding sediment concentration determined using the BSAF is 1.73 μg/kg dry weight, as described in Section 4.5.5. All applicable targets are shown below in Table 4-20. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 4-20. Total Chlordane Targets Applicable to Peck Road Park Lake

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	5.6
Sediment (ng /dry g)	Consensus-based TEC	3.24
Sediment (μg/kg dry weight)	BSAF-derived target	1.73
Water (ng/L)	CTR	0.59

Note: Shaded cells represent the selected targets for this TMDL.

4.5.3 Summary of Monitoring Data

This section summarizes the monitoring data for Peck Road Park Lake related to the chlordane impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 at five locations (six samples) and again in the fall of 2008 at two locations (three samples) in Peck Road Park Lake. These samples measured cis- and trans-chlordane, but not oxychlordane or nonachlor. All of these samples were less than sample detection limits (1.5 – 1.67 ng/L; note that the detection limit for chlordane is higher than the water quality criterion of 0.59 ng/L). Additional water column sampling was conducted by the Regional Board on December 11, 2008 at four in-lake locations in Peck Road Park Lake, including the oxychlordane and nonachlor components. All four samples were below the detection limit (1 ng/L, which is also above the water quality criterion). A summary of the water column data is shown in Table 4-21. (Note that these results are identical to those shown for PCBs because all samples were non-detect and the detections limits were the same for chlordane and PCBs.)

Table 4-21. Summary of Water Column Samples for Total Chlordane in Peck Road Park Lake

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
Sawpit Wash	(0.81)[2]	2	0
Santa Anita Wash	(0.78)	3	0
North Basin Outfall	(0.76)	2	0
North Basin	(0.60)	2	0
South Basin	(0.60)	2	0
South Basin East	(0.50)	1	0
South Basin West Side	(0.50)	1	0
In-Lake Average[3]	(0.60)		
Water Column Target	0.59		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that the sample is based only on the detection limits of the samples, and that no chlordane were detected in any of the collected samples.

[3] Overall average is the average of individual station averages.

In 2008, concentrations of chlordane on suspended sediment were analyzed in the summer at one location and in the fall at two locations as part of the UCLA study. All three samples were below detectable limits (2.26 µg/kg to 20.41 µg/kg dry weight). Porewater was sampled by UCLA in both the summer and fall of 2008. Specifically, chlordane concentrations in the porewater sampled at four sites during the summer of 2008 and three sites during the fall were all less than the detection limit of 15 ng/L. All four porewater suspended sediment samples collected in the summer of 2008 were below detectable levels (2.26 µg/kg to 9.25 µg/kg dry weight).

UCLA also collected sediment samples at four locations in Peck Road Park Lake in summer and fall 2008. As with the water column analyses by UCLA, these report cis- and trans-chlordane, but not oxychlordane or nonachlor. Only one of nine lake sediment samples was above the detection limit (which ranged from 0.34 µg/kg to 0.72 µg/kg dry weight) with a maximum of 7.1 µg/kg dry weight (in excess of the consensus-based TEC for sediment of 3.24 µg/kg dry weight).

Four in-lake sediment locations were sampled by USEPA and the county of Los Angeles on November 16, 2009, resulting in concentrations from 1.0 µg/kg to 19.5 µg/kg dry weight, with three of the four samples exceeding the consensus-based TEC of 3.24 µg/kg dry weight. These analyses do include oxychlordane and nonachlor. All lake stations were averaged to estimate an exposure concentration for chlordane in Peck Road Park Lake sediments of 4.14 µg/kg dry weight (with non-detects included at one-half the detection limit for each sample). Stations located near outfalls, are taken as an estimate of the concentrations on incoming sediment. A summary of the sediment data is shown in Table 4-22.

Table 4-22. Summary of Sediment Samples for Total Chlordane in Peck Road Park Lake

Station	Average Sediment Concentration (ng dry/g)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection Limit and Reporting Limit
Near Sawpit Wash	(0.19)	1	0	0
Near Santa Anita Wash	(0.23)	3	0	0
North Basin	5.96	4	2	0
South Basin	6.30	3	1	0
North Inlet	[1.00]	1	1	1
South Inlet	11.20	1	1	0
In-Lake Average[2]	4.14			
Influent Average	3.15			
Consensus-based TEC	3.24			

[1] Total chlordane in a sample represents the sum of all reported measurements for alpha and gamma chlordane, oxychlordane, and cis- and trans-nonachlor, including results reported below the method reporting limit. If all components were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no chlordane quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations of total chlordane from Peck Road Park Lake have been analyzed in largemouth bass (SWAMP and TSMP). Four largemouth bass samples collected between 1986 and 1992 ranged from non-detect to 42 ppb with an average of 21 ppb, well in excess of the FCG for chlordane

(5.6 ppb). Because chlordane is no longer in use, fish tissue concentrations are likely to have declined since these samples were taken. Recent fish tissue concentrations of chlordane have been analyzed in largemouth bass in two composite samples of filet tissue from five fish collected in summer 2007 and another composite sample collected in April 2010 (Table 4-23). These had an average total chlordane concentration of 13.44 ppb, in excess of the FCG. The average lipid fraction was 0.54 percent. Data from bottom-feeding fish (e.g., carp) are not available for Peck Road Park Lake.

Table 4-23. Summary of Recent Fish Tissue Samples for Total Chlordane in Peck Road Park Lake

Sample Date	Fish Species	Total Chlordane (ppb wet weight)[1]
6 June 2007	Largemouth Bass	19.212
6 June 2007	Largemouth Bass	8.637
19 April 2010	Largemouth Bass	12.465
2007 - 2010 Average		13.44
FCG		5.6

[1]Composite sample of filets from five individuals.

In sum, recent fish tissue concentrations in Peck Road Park Lake are consistently above the FCG in the three available largemouth bass composite samples. The average concentration in sediment is below the consensus-based TEC, although individual samples exceed the TEC. Water column samples have all been below detection limits.

4.5.4 Source Assessment

Chlordane in Peck Road Park Lake is primarily due to historical loading and storing within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed chlordane concentrations on sediment near inflows to the lake. Watershed loads of chlordane may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources within the watershed at this time. Therefore, an average concentration of sediment is applied to all contributing areas. The average concentration of chlordane on incoming sediment was estimated to be 3.15 μg/kg dry weight (Table 4-22), and the annual sediment load to Peck Road Park Lake is 990.3 tons/yr, including sediment delivered through the water diversion (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load of chlordane is approximately 2.83 g/yr (Table 4-24).

Table 4-24. Total Chlordane Loads Estimated for Each Jurisdiction and Subwatershed in the Peck Road Park Lake Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total Chlordane Load (g/yr)	Percent of Total Load
Eastern	Arcadia	MS4 Stormwater[1]	12.1	0.034	1.22%
Eastern	Bradbury	MS4 Stormwater[1]	44.4	0.127	4.48%
Eastern	Caltrans	State Highway Stormwater[1]	9.6	0.027	0.96%
Eastern	Duarte	MS4 Stormwater[1]	57.2	0.163	5.78%
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	0.8	0.002	0.08%
Eastern	Irwindale	MS4 Stormwater[1]	23.3	0.067	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.6	0.005	0.16%
Eastern	County of Los Angeles	MS4 Stormwater[1]	28.6	0.082	2.89%
Eastern	Monrovia	MS4 Stormwater[1]	200	0.573	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	16.3	0.047	1.65%
Eastern	Angeles National Forest	Stormwater[1]	12.1	0.035	1.22%
Diversion	Los Angeles County Department of Public Works	Water Diversion	379	1.084	38.31%
Near Lake	Arcadia	MS4 Stormwater[1]	7.6	0.022	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.7	0.005	0.17%
Near Lake	El Monte	MS4 Stormwater[1]	3.5	0.010	0.36%
Near Lake	Irwindale	MS4 Stormwater[1]	1.7	0.005	0.17%
Near Lake	County of Los Angeles	MS4 Stormwater[1]	4.0	0.012	0.41%
Near Lake	Monrovia	MS4 Stormwater[1]	2.6	0.007	0.26%
Western	Arcadia	MS4 Stormwater[1]	68.1	0.195	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	37.8	0.108	3.82%
Western	Caltrans	State Highway Stormwater[1]	2.1	0.006	0.21%
Western	County of Los Angeles	MS4 Stormwater[1]	14.7	0.042	1.49%
Western	Monrovia	MS4 Stormwater[1]	9.3	0.026	0.94%

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total Chlordane Load (g/yr)	Percent of Total Load
Western	Sierra Madre	MS4 Stormwater[1]	19.9	0.057	2.01%
Western	Angeles National Forest	Stormwater[1]	31.4	0.090	3.18%
Total Load from Watershed			**990.3**	**2.83**	**100%**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of chlordane directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

4.5.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of total chlordane into Peck Road Park Lake. The loading capacity is used to estimate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and other nonpoint sources (load allocations).

Lake sediments are often the predominant source of total chlordane in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Chlordanes are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of total chlordane will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data in Peck Road Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment chlordane concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target to achieve FCGs is calculated based on biota-sediment bioaccumulation (a BSAF approach), using the ratio of the FCG to existing fish tissue concentrations of 5.6/13.44 = 0.417. This ratio is applied to the observed sediment concentration of 4.14 μg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 1.73 μg/kg dry weight. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of chlordane are likely to have declined steadily since the cessation of production and use of the chemical. The resulting target concentration of chlordane in the sediment in Peck Road Park Lake is shown in Table 4-25.

Table 4-25. Fish Tissue-Based Chlordane Concentration Targets for Sediment in Peck Road Park Lake

Total Chlordane Concentration	Sediment (µg/kg dry weight)
Existing	4.14
BSAF-derived Target	1.73
Required Reduction	58.2%

The BSAF-derived sediment target is less than the consensus-based TEC of 3.24 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.59 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 696 g/yr would be required to maintain observed sediment concentrations under steady state conditions. The estimated watershed loading rate is 2.83 g/yr, or 0.4 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of chlordane in Peck Road Park Lake is primarily due to the storage of historic loads of chlordane in the lake sediment.

4.5.6 TMDL Summary

Because chlordane impairment in Peck Road Park Lake is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue (The concentration targets apply to water and sediment entering the lake and within the lake.

The chlordane TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 1.73 µg/kg dry weight chlordane. The wasteload allocations and load allocations are also equal to 1.73 µg/kg dry weight chlordane in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

4.5.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for chlordane (“Alternative WLAs if the Fish Tissue Target is Met”) described in Section 4.5.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 4.5.6.1.1 if the conditions described in Section 4.5.6.1.2 are met.

4.5.6.1.1 Wasteload Allocations

The entire watershed of Peck Road Park Lake is contained in MS4 jurisdictions, and therefore receives WLAs. The Caltrans areas and facilities that operate under a general industrial stormwater permit also receive WLAs.

Relevant permit numbers are

- County of Los Angeles (including the cities of Arcadia, Bradbury, Duarte, Irwindale, Monrovia, and Sierra Madre): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Total chlordane concentrations in water flowing into Peck Road Park Lake are below detection limits, and most chlordane load is expected to move in association with sediment. Therefore no separate wasteload allocation or reduction is explicitly assigned to the Colorado Well Aquifer (Order No. R4-2003-0108, CAG994005) as it is not expected to deliver sediment loads. On the other hand, the suspended sediment in the water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for chlordane in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved chlordane and chlordane associated with suspended sediment. The existing concentration of sediment entering the lake is 3.15 μg/kg dry weight. Therefore, a reduction of (3.15 – 1.73)/3.15 = 45.1 percent is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 4-26 and each wasteload allocation must be met at the point of discharge.

Table 4-26. Wasteload Allocations for Total Chlordane in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (μg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	1.73	0.59
Eastern	Bradbury	MS4 Stormwater[1]	1.73	0.59
Eastern	Caltrans	State Highway Stormwater[1]	1.73	0.59
Eastern	Duarte	MS4 Stormwater[1]	1.73	0.59
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	1.73	0.59
Eastern	Irwindale	MS4 Stormwater[1]	1.73	0.59
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.73	0.59
Eastern	County of Los	MS4 Stormwater[1]	1.73	0.59

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
	Angeles			
Eastern	Monrovia	MS4 Stormwater[1]	1.73	0.59
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	1.73	0.59
Eastern	Angeles National Forest	Stormwater[1]	1.73	0.59
Diversion	Los Angeles County Department of Public Works	Water Diversion	1.73	0.59
Near Lake	Arcadia	MS4 Stormwater[1]	1.73	0.59
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.73	0.59
Near Lake	El Monte	MS4 Stormwater[1]	1.73	0.59
Near Lake	Irwindale	MS4 Stormwater[1]	1.73	0.59
Near Lake	County of Los Angeles	MS4 Stormwater[1]	1.73	0.59
Near Lake	Monrovia	MS4 Stormwater[1]	1.73	0.59
Western	Arcadia	MS4 Stormwater[1]	1.73	0.59
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.73	0.59
Western	Caltrans	State Highway Stormwater[1]	1.73	0.59
Western	County of Los Angeles	MS4 Stormwater[1]	1.73	0.59
Western	Monrovia	MS4 Stormwater[1]	1.73	0.59
Western	Sierra Madre	MS4 Stormwater[1]	1.73	0.59
Western	Angeles National Forest	Stormwater[1]	1.73	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.5.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 4-26 will be superseded, and the wasteload allocations in Table 4-27 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 4-27, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 4-27. Alternative Wasteload Allocations for Total Chlordane in Peck Road Park Lake if the Fish Tissue Target is are Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	3.24	0.59
Eastern	Bradbury	MS4 Stormwater[1]	3.24	0.59
Eastern	Caltrans	State Highway Stormwater[1]	3.24	0.59
Eastern	Duarte	MS4 Stormwater[1]	3.24	0.59
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	3.24	0.59
Eastern	Irwindale	MS4 Stormwater[1]	3.24	0.59
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	3.24	0.59
Eastern	County of Los Angeles	MS4 Stormwater[1]	3.24	0.59
Eastern	Monrovia	MS4 Stormwater[1]	3.24	0.59
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	3.24	0.59
Eastern	Angeles National Forest	Stormwater[1]	3.24	0.59
Diversion	Los Angeles County Department of Public Works	Water Diversion	3.24	0.59
Near Lake	Arcadia	MS4 Stormwater[1]	3.24	0.59
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	3.24	0.59
Near Lake	El Monte	MS4 Stormwater[1]	3.24	0.59
Near Lake	Irwindale	MS4 Stormwater[1]	3.24	0.59
Near Lake	County of Los Angeles	MS4 Stormwater[1]	3.24	0.59
Near Lake	Monrovia	MS4 Stormwater[1]	3.24	0.59
Western	Arcadia	MS4 Stormwater[1]	3.24	0.59

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	3.24	0.59
Western	Caltrans	State Highway Stormwater[1]	3.24	0.59
Western	County of Los Angeles	MS4 Stormwater[1]	3.24	0.59
Western	Monrovia	MS4 Stormwater[1]	3.24	0.59
Western	Sierra Madre	MS4 Stormwater[1]	3.24	0.59
Western	Angeles National Forest	Stormwater[1]	3.24	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.5.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for chlordane ("Alternative LAs if the Fish Tissue Target is Met") described in Section 4.5.6.2.2. The alternative load allocations will supersede the load allocations in Section 4.5.6.2.1 if the conditions described in Section 4.5.6.2.2 are met.

4.5.6.2.1 Load Allocations

No part of the Peck Road Park Lake watershed is located outside of an MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to net direct atmospheric deposition of chlordane. The legacy chlordane stored in lake sediment is the major cause of use impairment due to elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdictions (County of Los Angeles) should achieve a total chlordane concentration of 1.73 µg/kg dry weight of lake bottom sediments (Table 4-28).

Table 4-28. Load Allocations for Total Chlordane in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	1.73

4.5.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 4-28 will be superseded, and the load allocations in Table 4-29 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 4-29, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Table 4-29. Alternative Load Allocations for Total Chlordane in Peck Road Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	3.24

4.5.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

4.5.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate chlordane, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

4.5.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the PCB WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99th percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Peck Road Park Lake watershed. USGS Station 11101250, on the Rio Hondo River above the Whittier Narrows Dam, was selected as a surrogate for flow determination. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for the Rio Hondo (952 cfs) (Wolock, 2003). To estimate the peak flow to Peck Road Park Lake, the 99th percentile flow for the Rio Hondo was scaled down by the ratio of drainage areas (23,564 acres/58,368 acres; Peck Road Park Lake watershed area/Rio Hondo watershed area at the gage). The resulting peak flow estimate for Peck Road Park Lake is 384 cfs. The 99th percentile diverted flow from the San Gabriel River to Peck Road Park Lake is 328 cfs. Therefore, the total peak daily flow rate is 712 cfs.

The event mean concentration of sediment in stormwater (71.7 mg/L) was calculated from the estimated existing watershed sediment load of 990.3tons/yr (Table 4-14) divided by the stormwater flow volume reaching the lake (10,158 ac-ft, Table 4-7). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (712 cfs) yields a daily maximum sediment load from stormwater of 137.7 tons/d. Applying the wasteload allocation concentration of 1.73 ng total chlordane per dry g of sediment yields the stormwater daily maximum allowable load of 0.216 g/d of total chlordane. This load is associated with the MS4 stormwater permittees and the water diversion. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

4.5.6.6 Future Growth

The manufacture and use of chlordane is currently banned. Therefore, no additional allowance is made for future growth in the chlordane TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.6 DDT Impairment

Dichlorodiphenyltrichloroethane (DDT) is a synthetic organochlorine insecticide once used throughout the world to control insects. Technically DDT consists of two isomers, 4,4’-DDT and 2,4’-DDT, of which the former is the most toxic. In the environment, DDT breaks down to form two related compounds: DDD (tetrachlorodiphenylethane) and DDE (dichlorodiphenyl-dichloroethylene). DDD and DDE often predominate in the environment and USEPA (2000c) recommends that fish consumption guidelines be based on the sum of DDT, DDD, and DDE – collectively referred to as total DDTs.

The DDT impairment of Peck Road Park Lake affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. DDT, like PCBs and chlordane, is an organochlorine compound that is strongly sorbed to sediment and lipids, and is no longer in production. As such, the approach for the DDT impairment is similar to that for PCBs and chlordane.

4.6.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of DDT are currently impairing the REC1, REC2 and WARM uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

4.6.2 Numeric Targets

Targets for DDT are complex because of the many different ways in which the compound is measured. The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses for several DDTs. There are no numeric criteria specified for sediment or fish tissue concentrations of DDTs listed in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets used for DDTs are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for DDT in the Basin Plan are associated with a specific beneficial use. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this report is designated WARM, at a minimum, and must meet this requirement. Acute and chronic criteria for 4,4'-DDT in freshwater systems are included in the CTR as 1.1 μg/L and 0.001 μg/L, respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. Acute and chronic values for other DDT compounds were not specified. The CTR also includes human health criteria for 4,4'-DDT for the consumption of water and organisms or organisms only as 0.00059 μg/L for both uses (USEPA, 2000a). Because the human health criterion is the most restrictive applicable criterion, a water column target of 0.00059 μg/L (0.59 ng/L) for 4,4'-DDT is the appropriate target. The CTR also specifies a criterion of 0.59 ng/L for 4,4'-DDE (for both consumption of water and organisms or organisms only), while for 4,4'-DDD the criteria are 0.83 ng/L for consumption of water and organisms and 0.84 ng/L for consumption of organisms only. For Peck Road Park Lake, the Regional Board has determined that the appropriate human health criterion for 4,4'-DDD is 0.00084 μg/L (0.84 ng/L) as the MUN use is not an existing use. The CTR does not specify a criterion for total DDTs. For this TMDL the DDT, DDD, and DDE targets in CTR are selected as water column targets.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) for 4,4'- plus 2,4'-DDT is 4.16 μg/kg dry weight, and the TEC for total DDTs is 5.28 μg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. These targets are designed to protect benthic dwelling organisms and explicitly do not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for total DDTs defined by OEHHA (2008) is 21 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For DDTs, the corresponding sediment concentration target determined using the BSAF is 6.90 μg/kg dry weight, as described in further detail in Section 4.6.5. All applicable targets are shown below in Table 4-30. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 4-30. DDT Targets Applicable to Peck Road Park Lake

Medium	Source	4,4'-DDT	4,4'-DDT + 2,4'-DDT	DDE[1]	DDD[1]	Total DDTs
Fish (ppb wet weight)	OEHHA FCG					21
Sediment (μg/kg dry weight)	Consensus-based TEC		4.16	3.16[1]	4.88[1]	5.28
Sediment (μg/kg dry weight)	BSAF-derived target					6.90
Water (ng/L)	CTR	0.59		0.59[1]	0.84[1]	

[1] CBSQG specifies sediment targets for total DDE and total DDD. The CTR specifies water column targets specifically for 4,4'-DDE and 4,4'-DDD.

Note: Shaded cells represent the selected targets for this TMDL.

4.6.3 Summary of Monitoring Data

This section summarizes the monitoring data for Peck Road Park Lake related to the DDT impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 at five locations (six samples) and again in the fall of 2008 at two locations (three samples) in Peck Road Park Lake. These analyses quantified only the 4,4' isomers of DDT, DDD, and DDE. All samples collected as part of the UCLA study during the summer and fall, were less than the sample detection limits (3.0 – 3.3 ng/L, all higher than the water quality criteria of 0.59 – 0.84 ng/L). Additional water column sampling was conducted by the Regional Board on December 11, 2008 at four in-lake locations in Peck Road Park Lake, including both the 4,4' and 2,4' isomers. All four sites sampled were below detectable levels of DDT (1 ng/L, which is also higher than the water quality criterion). A summary of the water column data is shown in Table 4-31.

Table 4-31. Summary of Water Column Samples for Total DDTs in Peck Road Park Lake

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
Sawpit Wash	(1.62)[1]	2	0
Santa Anita Wash	(1.56)	3	0
North Basin Outfall	(1.52)	2	0
North Basin	(1.0)	2	0
South Basin	(1.0)	2	0

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
South Basin East	(0.50)	1	0
South Basin West Side	(0.50)	1	0
In-Lake Average[3]	(0.80)		
Water Column Target	0.59		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no DDTs were detected in any of the collected samples.

[3] Overall average is the average of individual station averages (excludes the tributary samples).

Concentrations of total DDTs on suspended sediment were also analyzed by UCLA in the summer and fall of 2008. One in-lake location was analyzed in the summer and two in the fall; all three samples were below detectable limits for DDT (4.73 μg/kg to 40.82 μg/kg dry weight). Porewater samples were collected during the summer and fall of 2008; DDT concentrations in all of the porewater samples were less than the detection limit of 30 ng/L. All four porewater suspended sediment samples collected in the summer of 2008 were below detectable levels (4.51 μg/kg to 18.50 μg/kg dry weight).

UCLA also collected bed sediment samples at four locations in Peck Road Park Lake in summer and fall 2008. As with the UCLA water column samples, these included only the 4,4' isomers. Only one of nine sediment samples collected in 2008 (average of 10.2 μg/kg dry weight) was above method reporting limits for DDTs; two samples were detected at less than the reporting limits (which ranged from 6.87 μg/kg to 13.06 μg/kg dry weight). Four in-lake locations were sampled by USEPA and the county of Los Angeles on November 16, 2009. Three of four samples were above the detection limit (1 μg/kg dry weight), with a maximum of 11.8 μg/kg dry weight (in excess of the consensus-based TEC for sediment of 4.16 μg/kg dry weight).

All lake stations were averaged to estimate an exposure concentration of 5.09 μg/kg dry weight total DDTs (with non-detects included at one-half the detection limit for each sample). Stations located near outfalls are taken as an estimate of the concentrations on incoming sediment. The lake-wide average of 5.09 μg/kg dry weight is slightly less than the consensus-based TEC of 5.28 μg/kg dry weight. A summary of the sediment data is shown in Table 4-32.

Table 4-32. Summary of Sediment Samples for Total DDTs in Peck Road Park Lake, 2008-2009

Station	Average Sediment Concentration (μg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
Near Sawpit Wash	10.22	1	1	0
Near Santa Anita Wash	[0.54]	3	1	1
North Basin	3.94	4	2	1
South Basin	4.32	3	1	0
North Inlet	(0.50)	1	0	0
South Inlet	11.0	1	1	0

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
In-Lake Average[2]	5.09			
Influent Average	5.57			
Consensus-based TEC	5.28			

[1] Total DDT in a sample represents the sum of all reported measurements for DDT, DDE, and DDD isomers, including results reported below the method reporting limit. If all components were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no chlordane quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations of DDT from Peck Road Park Lake have been analyzed in largemouth bass (by TSMP and SWAMP). Total DDT concentrations in fish tissue collected between 1986 and 1992 ranged up to 39 ppb, with an average of 26.5 ppb, in excess of the FCG of 21 ppb. Because DDT is no longer in use, fish tissue concentrations are likely to have declined since these samples were taken. Considering only data collected in the past 10 years, the average concentration of total DDTs in largemouth bass was 15.5 ppb, at an average lipid content of 0.54 percent. This average is based on two largemouth bass composite samples (each containing filets from five individual fish) collected by SWAMP in the summer of 2007 and an additional composite collected in April 2010. Based on the current data, average fish tissue levels of total DDTs are less than the FCG of 21 ppb (Table 4-33). Data from bottom-feeding fish (e.g., carp) are not available for Peck Road Park Lake.

Table 4-33. Summary of Recent Fish Tissue Samples for Total DDTs in Peck Road Park Lake

Sample Date	Fish Taxa	Total DDTs (ppb wet weight)[1]
6 June 2007	Largemouth Bass	24.4
6 June 2007	Largemouth Bass	9.0
19 April 2010	Largemouth Bass	13.109
2007 Average		15.5
FCG		21

[1] Composite sample of filets from five individuals.

In sum, the average of recent fish tissue samples collected from Peck Road Park Lake is approximately 25 percent lower than the FCG, although one of three composite samples exceeded the FCG. Measured concentrations in sediment are within 2 percent of the consensus-based TEC with several samples based on half of the detection limit. However, individual stations had concentrations well above the TEC, indicating that the lake continues to be impaired by DDT. Concentrations in water were less than the detection limits.

4.6.4 Source Assessment

Total DDTs present in Peck Road Park Lake are primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed DDT concentrations on sediment data near inflows to the lake. Watershed loads of DDT may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated DDT load within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas. The average concentration of total DDTs on incoming sediment was estimated to be 5.57 μg/kg dry weight (Table 4-32), and the annual sediment load to Peck Road Park Lake is 990.3 tons/yr, including sediment delivered through the water diversion (see Appendix D, Wet Weather Loading). The resulting estimated wet-weather load of total DDTs is approximately 5.0 g/yr (Table 4-34).

Table 4-34. Total DDTs Loads Estimated for Each Jurisdiction and Subwatershed in the Peck Road Park Lake Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total DDTs Load (g/yr)	Percent of Total Load
Eastern	Arcadia	MS4 Stormwater[1]	12.1	0.061	1.22%
Eastern	Bradbury	MS4 Stormwater[1]	44.4	0.224	4.48%
Eastern	Caltrans	State Highway Stormwater[1]	9.6	0.048	0.96%
Eastern	Duarte	MS4 Stormwater[1]	57.2	0.289	5.78%
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	0.8	0.004	0.08%
Eastern	Irwindale	MS4 Stormwater[1]	23.3	0.118	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.6	0.008	0.16%
Eastern	County of Los Angeles	MS4 Stormwater[1]	28.6	0.145	2.89%
Eastern	Monrovia	MS4 Stormwater[1]	200	1.013	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	16.3	0.061	1.22%
Eastern	Angeles National Forest	Stormwater[1]	12.1	1.917	38.31%
Diversion	Los Angeles County Department of Public Works	Water Diversion	379	0.038	0.77%
Near Lake	Arcadia	MS4 Stormwater[1]	7.6	0.009	0.17%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.7	0.018	0.36%
Near Lake	El Monte	MS4 Stormwater[1]	3.5	0.009	0.17%
Near Lake	Irwindale	MS4 Stormwater[1]	1.7	0.020	0.41%
Near Lake	County of Los Angeles	MS4 Stormwater[1]	4.0	0.013	0.26%

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total DDTs Load (g/yr)	Percent of Total Load
Near Lake	Monrovia	MS4 Stormwater[1]	2.6	0.344	6.88%
Western	Arcadia	MS4 Stormwater[1]	68.1	0.191	3.82%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	37.8	0.010	0.21%
Western	Caltrans	State Highway Stormwater[1]	2.1	0.074	1.49%
Western	County of Los Angeles	MS4 Stormwater[1]	14.7	0.047	0.94%
Western	Monrovia	MS4 Stormwater[1]	9.3	0.100	2.01%
Western	Sierra Madre	MS4 Stormwater[1]	19.9	0.159	3.18%
Western	Angeles National Forest	Stormwater[1]	31.4	0.061	1.22%
Total Load from Watershed			**990.3**	**5.00**	**100%**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of DDTs directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

4.6.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity for DDTs in Peck Road Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of DDT in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. DDT is strongly sorbed to sediment and has a long half-life in sediment and water. Incoming loads of DDT will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data in Peck Road Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. A sediment target to achieve FCGs is calculated based on biota-sediment bioaccumulation (a BSAF approach), using the ratio of the FCG to existing fish tissue concentrations of 21/15.5 = 1.355. This ratio is applied to the estimated lake sediment concentration of 5.09 μg/kg dry weight to obtain the site-specific sediment target concentration to maintain fish tissue goals of 6.90 μg/kg dry weight. The BSAF-derived sediment target is greater than the estimated existing sediment concentration because the average recent fish tissue concentration does not exceed the fish tissue based target concentration.

The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of total DDT are likely to have declined steadily since the cessation of production and use of the chemical. The resulting fish tissue-based target concentrations of DDT in sediment of Peck Road Park Lake are shown in Table 4-35.

Table 4-35. Fish Tissue-Based Total DDTs Concentration Targets for Sediment in Peck Road Park Lake

Total DDTs Concentration	Sediment (µg/kg dry weight)
Existing	5.09
BSAF-derived Target	6.90
Required Reduction	0%

The BSAF-derived sediment target is greater than the consensus-based TEC for total DDTs of 5.28 µg/kg dry weight. The consensus-based TEC of 5.28 µg/kg dry weight is therefore the most restrictive target and is used as the target in this TMDL. Selection of the consensus-based TEC target protects the benthic biota and ensures continued attainment of the fish tissue based target concentration. The estimated existing concentration in lake of 5.09 µg/kg is less than the TEC, which would imply that no reduction from existing in-lake sediment concentrations may be needed. However, the estimated influent concentration is greater than the TEC.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 84 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 5 g/yr, or 6 percent of this amount. Thus, concentrations of total DDTs in fish tissue in Peck Road Park Lake appear to be primarily due to the storage of historic loads of DDT in the lake sediment.

4.6.6 TMDL Summary

Because DDT impairment in Peck Road Park Lake is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of maintaining the existing concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake.

The DDT TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 5.28 µg/kg dry weight total DDTs. The wasteload allocations and load allocations are also equal to 5.28 µg/kg dry weight total DDTs in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

4.6.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). The entire watershed of Peck Road Park Lake is contained in MS4 jurisdictions, and watershed loads are therefore assigned WLAs. The Caltrans areas and facilities that operate under a general industrial stormwater permit also receive WLAs.

Relevant permit numbers are

- County of Los Angeles (including the cities of Arcadia, Bradbury, Duarte, Irwindale, Monrovia, and Sierra Madre): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

DDT in water flowing into Peck Road Park Lake is below detection limits, and most DDT load is expected to move in association with sediment. Therefore, no separate wasteload allocation or reduction is explicitly assigned to the Colorado Well Aquifer (Order No. R4-2003-0108, CAG994005)as it is not expected to deliver sediment loads. On the other hand, the suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for DDT in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved DDT and DDT associated with suspended sediment. Each wasteload allocation applies at the point of discharge. The existing concentration of sediment entering the lake is 5.57 µg/kg dry weight. Therefore, a reduction of 5.2 percent [(5.57 – 5.28)/5.57*100] is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 4-36 and each wasteload allocation must be met at the point of discharge.

Table 4-36. Wasteload Allocations for Total DDTs in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for DDT Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column (ng/L)[3,4]
Eastern	Arcadia	MS4 Stormwater[1]	5.28	0.59[3]
Eastern	Bradbury	MS4 Stormwater[1]	5.28	0.59
Eastern	Caltrans	State Highway Stormwater[1]	5.28	0.59
Eastern	Duarte	MS4 Stormwater[1]	5.28	0.59
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	5.28	0.59
Eastern	Irwindale	MS4 Stormwater[1]	5.28	0.59
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	5.28	0.59
Eastern	County of Los Angeles	MS4 Stormwater[1]	5.28	0.59
Eastern	Monrovia	MS4 Stormwater[1]	5.28	0.59

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for DDT Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column (ng/L)[3,4]
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	5.28	0.59
Eastern	Angeles National Forest	Stormwater[1]	5.28	0.59
Diversion	Los Angeles County Department of Public Works	Water Diversion	5.28	0.59
Near Lake	Arcadia	MS4 Stormwater[1]	5.28	0.59
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	5.28	0.59
Near Lake	El Monte	MS4 Stormwater[1]	5.28	0.59
Near Lake	Irwindale	MS4 Stormwater[1]	5.28	0.59
Near Lake	County of Los Angeles	MS4 Stormwater[1]	5.28	0.59
Near Lake	Monrovia	MS4 Stormwater[1]	5.28	0.59
Western	Arcadia	MS4 Stormwater[1]	5.28	0.59
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	5.28	0.59
Western	Caltrans	State Highway Stormwater[1]	5.28	0.59
Western	County of Los Angeles	MS4 Stormwater[1]	5.28	0.59
Western	Monrovia	MS4 Stormwater[1]	5.28	0.59
Western	Sierra Madre	MS4 Stormwater[1]	5.28	0.59
Western	Angeles National Forest	Stormwater[1]	5.28	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

[4] The target water column concentration of 0.59 ng/L specified in the CTR is for 4,4'-DDT. The CTR also specifies targets for DDE and DDD, but does not specify a target for total DDTs. The lowest DDT target is selected for the purposes of representing Total DDTs in this table. If analytical results that resolve individual DDT compounds are available, all of the CTR criteria should be applied individually.

4.6.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. No part of the Peck Road Park Lake watershed is outside MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to atmospheric deposition of DDTs. The legacy DDT stored in lake sediment is the major cause of exposure to aquatic organisms and sport fish, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdictions (County of Los Angeles) should achieve or maintain a total DDTs concentration of 5.28 µg/kg dry weight or less in lake bottom sediments (Table 4-37).

Table 4-37. Load Allocations for Total DDT in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	5.28

4.6.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected consensus-based TEC concentration in sediment is considerably lower than the BSAF-derived target.

4.6.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate DDT, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

4.6.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the DDT WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Peck Road Park Lake watershed. USGS Station 11101250, on the Rio Hondo River above the Whittier Narrows Dam, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99^{th} percentile flow for the Rio Hondo (952 cfs) (Wolock, 2003). To estimate the peak flow to Peck Road Park Lake, the 99^{th} percentile flow for the Rio Hondo was scaled down by the ratio of drainage areas (23,564 acres/58,368 acres; Peck Road Park Lake watershed area/Rio Hondo watershed area at the gage). The resulting peak flow estimate for

Peck Road Park Lake is 384 cfs. The 99th percentile diverted flow from the San Gabriel River to Peck Road Park Lake is 328 cfs. Therefore, the total peak daily flow rate is 712 cfs.

The event mean concentration of sediment in stormwater (71.7 mg/L) was calculated from the estimated existing watershed sediment load of 990.3 tons/yr (Table 4-14) divided by the stormwater volume reaching the lake (10,158 ac-ft, Table 4-7). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (712 cfs) yields a daily maximum sediment load from stormwater of 137.7 tons/d. Applying the wasteload allocation concentration of 5.28 ng total DDT per dry g of sediment yields the stormwater daily maximum allowable load of 0.659 g/d of total DDT. This load is associated with the MS4 stormwater permittees and the water diversion. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

4.6.6.6 Future Growth

The manufacture and use of DDT is currently banned. Therefore, no additional allowance is made for future growth in the DDT TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.7 Dieldrin Impairment

Dieldrin is a chlorinated insecticide originally developed as an alternative to DDT and was in wide use from the 1950s to the 1970s. Dieldrin in the environment also arises from use of the insecticide aldrin. Aldrin is not itself toxic to insects, but is metabolized to dieldrin in the insect body. The use of both dieldrin and aldrin was discontinued in the 1970s.

The dieldrin impairment of Peck Road Park Lake affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. Dieldrin, like PCBs, chlordane and DDT, is an organochlorine compound that is strongly sorbed to sediment and lipids and is no longer in production. As such, the approach for dieldrin impairment is similar to that for PCBs, chlordane, and DDT.

4.7.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo River below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of dieldrin are currently impairing the REC1, REC2 and WARM uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

4.7.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of dieldrin in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets for dieldrin are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for dieldrin in the Basin Plan are associated with a specific beneficial use. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Acute and chronic criterion for the protection of aquatic life and wildlife in freshwater systems are included in the CTR for dieldrin as 0.24 µg/L and 0.056 µg/L, respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. The CTR also provides a human health-based water quality criterion for the consumption of organisms only and the consumption of water and organisms as 0.00014 µg/L (0.14 ng/L). The human health criterion of 0.00014 µg/L (0.14 ng/L) is the most restrictive of the applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) of dieldrin in sediment is 0.46 µg/kg. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The estimated existing sediment dieldrin concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for dieldrin defined by the OEHHA (2008) is 0.46 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For dieldrin, the corresponding sediment concentration target is estimated using the BSAF approach is 0.43 µg/kg dry weight, as described in detail in Section 4.7.5. All applicable targets are shown below in Table 4-38. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 4-38. Dieldrin Targets Applicable to Peck Road Park Lake

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	0.46
Sediment (µg/kg dry weight)	Consensus-based TEC	1.9
Sediment (µg/kg dry weight)	BSAF-derived target	0.43
Water (ng/L)	CTR	0.14

Note: Shaded cells represent the selected targets for this TMDL.

4.7.3 Summary of Monitoring Data

This section summarizes the monitoring data for Peck Road Park Lake related to the dieldrin impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 at five locations (six samples) and again in the fall of 2008 at two locations (three samples) in Peck Road Park Lake. All samples collected as part of the UCLA study during the summer and fall, were less than the sample detection limit (3.0 ng/L to 3.3 ng/L; all greater than the water quality criterion of 0.14 ng/L). Additional water column sampling was conducted by the Regional Board on December 11, 2008 at four in-lake locations in Peck Road Park Lake. All four sites sampled had non-detectable concentrations of dieldrin (less than 1 ng/L, also greater than the water column criterion). A summary of the water column data is shown in Table 4-39.

Table 4-39. Summary of Water Column Samples for Dieldrin in Peck Road Park Lake

Station	Average Water Concentration (ng/L)[2]	Number of Samples	Number of Samples Above Detection Limits[1]
Sawpit Wash	(1.62)	2	0
Santa Anita Wash	(1.56)	3	0
North Basin Outfall	(1.52)	2	0
North Basin	(1.0)	2	0
South Basin	(1.0)	2	0
South Basin East	(0.50)	1	0
South Basin West Side	(0.50)	1	0
In-Lake Average[3]	(0.80)		
Water Column Target	0.17		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no dieldrin was detected in any of the collected samples.

[3] Overall average is the average of individual station averages (excludes the tributary samples).

Concentrations of dieldrin on suspended sediment were also analyzed by UCLA in the summer and fall of 2008. One in-lake location was analyzed in the summer and two were sampled in the fall, all three samples were below detectable limits for dieldrin (4.73 µg/kg to 40.83 µg/kg dry weight). Porewater was sampled by UCLA in both the summer and fall of 2008. Specifically, dieldrin concentrations in the porewater sampled at four sites during the summer of 2008 were all less than the detection limit of 30 ng/L; three sites sampled during the fall of 2008 were also below the detection limit of 30 ng/L. All four porewater suspended sediments collected in the summer of 2008 were below detectable levels (4.51 µg/kg to 18.50 µg/kg dry weight).

UCLA also collected bed sediment samples at four locations in Peck Road Park Lake in summer and fall 2008 (Table 4-40). All nine sediment samples collected during 2008 resulted in dieldrin concentrations below the detection limit (which ranged from 0.69 µg/kg to 1.44 µg/kg dry weight). Four in-lake sediment locations were sampled by USEPA and the county of Los Angeles on November 16, 2009; all were below detection limit (1 µg/kg dry weight). The average of all samples with non-detects set equal to one-half of the individual sample detection limit is 0.49 µg/kg dry weight. Because dieldrin does appear in fish at levels greater than the FCG, and because these body burdens of dieldrin are believed to arise from the sediment, EPA decided to represent statistical estimates for the sediment concentrations of dieldrin by setting the concentration of non-detected samples to the detection limit. For an upper bound analysis the average with all samples set equal to the detection limit is 0.98 µg/kg dry weight. Stations located near outfalls are taken as an estimate of the concentrations on incoming sediment. The lake-wide average of <0.98 µg/kg dry weight for dieldrin is still less than the consensus-based TEC of 5.28 µg/kg dry weight.

Table 4-40. Summary of Sediment Samples for Dieldrin in Peck Road Park Lake, 2008-2009

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples Above Detection Limits[1]
Near Sawpit Wash	(0.74)	1	0
Near Santa Anita Wash	(0.90)	3	0
North Basin	(1.13)	4	0
South Basin	(1.11)	3	0
North Inlet	(1.00)	1	0
South Inlet	(1.00)	1	0
In-Lake Average[2]	(0.98)		
Influent Average	(0.91)		
Consensus-based TEC	1.9		

[1] Non-detect samples are included in reported averages at the detection limit. Numbers in round parentheses indicate a result is based only on the detection limits of the samples, and that no dieldrin was detected in any of the samples collected at that station.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations for dieldrin from Peck Road Park Lake have been analyzed in largemouth bass (TSMP and SWAMP). Dieldrin concentrations in the fish tissue ranged from non-detect to 0.97 ppb. Two of the four samples of largemouth bass were taken in 1991 and 1992 and both were below detection limits (value not stated). Considering only data collected in the past 10 years, the average concentration of dieldrin in largemouth bass was 1.06 ppb, in excess of the FCG of 0.46 ppb. This average is based on the two largemouth bass composite samples (each containing filet tissue from five individual fish) collected by SWAMP in the summer of 2007 and an additional composite sample collected in April 2010, with an average lipid fraction of 0.54 percent. Recent fish-tissue data for Peck Road Park Lake are summarized in Table 4-41. Data from bottom-feeding fish (e.g., carp) are not available for Peck Road Park Lake.

Table 4-41. Summary of Recent Fish Tissue Samples for Dieldrin in Peck Road Park Lake

Sample Date	Fish Taxa	Dieldrin (ppb wet weight)[1]
6 June 2007	Largemouth Bass	0.965
6 June 2007	Largemouth Bass	0.542
19 April 2010	Largemouth Bass	1.66
2007 - 2010 Average		1.06
FCG		0.46

[1] Composite sample of filets from five individuals.

In sum, recent fish tissue concentrations in Peck Road Park Lake are consistently above the FCG in largemouth bass composite samples. Sediment and water column concentrations have all been below detection limits.

4.7.4 Source Assessment

Dieldrin in Peck Road Park Lake is primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed could not be directly estimated because all sediment and water samples were below detection limits. Watershed loads of dieldrin may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations.

There is no definitive information on specific sources within the watershed at this time. Therefore, an average concentration of sediment is applied to all contributing areas.

An upper-bound analysis for dieldrin is performed using the simulated sediment load and detection limit to determine the maximum potential loading rate of dieldrin from the watershed. The dieldrin sediment concentration is assigned as the upper bound estimate of concentration on influent sediment (0.91 µg/kg dry weight, calculated with non-detects set equal to the individual sample detection limits). The annual sediment load to Peck Road Park Lake, including sediment delivered through the water diversion (see Appendix D, Wet Weather Loading) is 990.3 tons/yr,. The resulting estimated upper bound on wet-weather load of dieldrin from the watershed is 0.82 g/yr or less (Table 4-42).

Table 4-42. Maximum Potential Dieldrin Loads for Each Jurisdiction and Subwatershed in the Peck Road Park Lake Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total Dieldrin Load (g/yr)	Percent of Total Load
Eastern	Arcadia	MS4 Stormwater[1]	12.1	<0.010	1.22%
Eastern	Bradbury	MS4 Stormwater[1]	44.4	<0.037	4.48%
Eastern	Caltrans	State Highway Stormwater[1]	9.6	<0.008	0.96%
Eastern	Duarte	MS4 Stormwater[1]	57.2	<0.047	5.78%
Eastern	General Industrial Stormwater Permittees[2]	General Industrial	0.8	<0.001	0.08%

Subwatershed	Responsible Jurisdiction	Input	Sediment (tons/yr)	Total Dieldrin Load (g/yr)	Percent of Total Load
	(in the city of Duarte)	Stormwater[1]			
Eastern	Irwindale	MS4 Stormwater[1]	23.3	<0.019	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.6	<0.001	0.16%
Eastern	County of Los Angeles	MS4 Stormwater[1]	28.6	<0.024	2.89%
Eastern	Monrovia	MS4 Stormwater[1]	200.5	<0.165	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	16.3	<0.013	1.65%
Eastern	Angeles National Forest	Stormwater[1]	12.1	<0.010	1.22%
Diversion	Los Angeles County Department of Public Works	Water Diversion	379	<0.313	38.31%
Near Lake	Arcadia	MS4 Stormwater[1]	7.6	<0.006	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.7	<0.001	0.17%
Near Lake	El Monte	MS4 Stormwater[1]	3.5	<0.003	0.36%
Near Lake	Irwindale	MS4 Stormwater[1]	1.7	<0.001	0.17%
Near Lake	County of Los Angeles	MS4 Stormwater[1]	4.0	<0.003	0.41%
Near Lake	Monrovia	MS4 Stormwater[1]	2.6	<0.002	0.26%
Western	Arcadia	MS4 Stormwater[1]	68.2	<0.056	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	37.8	<0.031	3.82%
Western	Caltrans	State Highway Stormwater[1]	2.1	<0.002	0.21%
Western	County of Los Angeles	MS4 Stormwater[1]	14.7	<0.012	1.49%
Western	Monrovia	MS4 Stormwater[1]	9.3	<0.008	0.94%
Western	Sierra Madre	MS4 Stormwater[1]	19.9	<0.016	2.01%
Eastern	Angeles National Forest	Stormwater[1]	31.4	<0.026	3.18%
Total Load from Watershed			**990.3**	**<0.818**	**100%**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. The disturbed area associated with general construction and general industrial stormwater permittees (510 acres) was subtracted out of the appropriate city areas and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of dieldrin directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

4.7.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of dieldrin into Peck Road Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of dieldrin in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Dieldrin is strongly sorbed to sediments and has a long half-life in sediment and water. Incoming loads of dieldrin will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data in Peck Road Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The estimated existing sediment dieldrin concentrations in Peck Road Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated using ratio of the FCG to existing fish tissue concentrations in largemouth bass of 0.46/1.06 = 0.434. Sediment concentrations of dieldrin in Peck Road Park Lake are reported as below detection limits ranging from 0.7 to 1.44 µg/kg dry weight. However, dieldrin is highly bioaccumulative, and low sediment concentrations can lead to unacceptable fish tissue concentrations (see Appendix H, Organochlorine Compounds TMDL Development). Using an estimated concentration of 0.98 µg/kg dry weight based on the average of the sample detection limits, the resulting target concentration would be 0.43 µg/kg dry weight to obtain FCGs. Calculation with a literature-based BSAF (Appendix G, Monitoring Data) suggests that even lower concentrations might be needed. However, the literature-based BSAF is highly uncertain and may not be directly applicable to conditions in Peck Road Park Lake. Therefore, the target based on the detection limits is used, with acknowledgment that the estimate may need to be refined if additional data are collected at lower detection limits. The resulting fish tissue-based target concentration of dieldrin in the sediment of Peck Road Park Lake is shown in Table 4-43.

Table 4-43. Fish Tissue-Based Dieldrin Concentration Targets for Sediment in Peck Road Park Lake

Total Dieldrin Concentration	Sediment (µg/kg dry weight)
Existing	< 0.98
BSAF-derived Target	0.43
Required Reduction	< 56.1%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline of 1.9 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.14 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

4.7.6 TMDL Summary

Dieldrin was below detection limits in both water and sediment samples of Peck Road Park Lake and its tributaries. The concentration observed in fish is most likely due to historic loads stored in the lake sediment, which is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue concentrations. The concentration targets apply to water and sediment entering the lake and within the lake.

The dieldrin TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 0.43 μg/kg dry weight dieldrin. The wasteload allocations and load allocations are also equal to 0.43 μg/kg dry weight dieldrin in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

4.7.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for dieldrin ("Alternative WLAs if the Fish Tissue Target is Met") described in Section 4.7.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 4.7.6.1.1 if the conditions described in Section 4.7.6.1.2 are met.

4.7.6.1.1 Wasteload Allocations

The entire watershed of Peck Road Park Lake is contained in MS4 jurisdictions, and watershed loads are therefore assigned WLAs. The Caltrans areas and facilities that operate under a general industrial stormwater permit also receive WLAs.

Relevant permit numbers are

- County of Los Angeles (including the cities of Arcadia, Bradbury, Duarte, Irwindale, Monrovia, and Sierra Madre): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Measurements of dieldrin in sediment and water flowing into Peck Road Park Lake are below detection limits, but most dieldrin load is expected to move in association with sediment. Therefore no separate wasteload allocation or reduction is assigned to the Colorado Well Aquifer (Order No. R4-2003-0108, CAG994005) as it is not expected to deliver sediment loads. On the other hand, the suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for dieldrin in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved dieldrin and dieldrin associated with suspended sediment. Comparing the sediment concentration target to the average detection limit for the influent samples of 0.91 μg/kg dry weight suggests that a reduction of approximately 53 percent in dieldrin loads is needed.

The wasteload allocations are shown in Table 4-44 and each wasteload allocation must be met at the point of discharge.

Table 4-44. Wasteload Allocations for Dieldrin in Peck Road Park Lake

Sub-watershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	0.43	0.14
Eastern	Bradbury	MS4 Stormwater[1]	0.43	0.14
Eastern	Caltrans	State Highway Stormwater[1]	0.43	0.14
Eastern	Duarte	MS4 Stormwater[1]	0.43	0.14
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	0.43	0.14
Eastern	Irwindale	MS4 Stormwater[1]	0.43	0.14
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	0.43	0.14
Eastern	County of Los Angeles	MS4 Stormwater[1]	0.43	0.14
Eastern	Monrovia	MS4 Stormwater[1]	0.43	0.14
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	0.43	0.14
Eastern	Angeles National Forest	Stormwater[1]	0.43	0.14
Diversion	Los Angeles County Department of Public Works	Water Diversion	0.43	0.14
Near Lake	Arcadia	MS4 Stormwater[1]	0.43	0.14
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	0.43	0.14
Near Lake	El Monte	MS4 Stormwater[1]	0.43	0.14
Near Lake	Irwindale	MS4 Stormwater[1]	0.43	0.14
Near Lake	County of Los Angeles	MS4 Stormwater[1]	0.43	0.14
Near Lake	Monrovia	MS4 Stormwater[1]	0.43	0.14
Western	Arcadia	MS4 Stormwater[1]	0.43	0.14
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	0.43	0.14
Western	Caltrans	State Highway Stormwater[1]	0.43	0.14
Western	County of Los Angeles	MS4 Stormwater[1]	0.43	0.14
Western	Monrovia	MS4 Stormwater[1]	0.43	0.14

Sub-watershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Western	Sierra Madre	MS4 Stormwater[1]	0.43	0.14
Western	Angeles National Forest	Stormwater[1]	0.43	0.14

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.7.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 4-44 will be superseded, and the wasteload allocations in Table 4-45 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 4-45, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 4-45. Alternative Wasteload Allocations for Dieldrin in Peck Road Park Lake if the Fish Tissue Target is Met

Sub-watershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Eastern	Arcadia	MS4 Stormwater[1]	1.90	0.14
Eastern	Bradbury	MS4 Stormwater[1]	1.90	0.14
Eastern	Caltrans	State Highway Stormwater[1]	1.90	0.14
Eastern	Duarte	MS4 Stormwater[1]	1.90	0.14
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	General Industrial Stormwater[1]	1.90	0.14
Eastern	Irwindale	MS4 Stormwater[1]	1.90	0.14
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	General Industrial Stormwater[1]	1.90	0.14

Sub-watershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Eastern	County of Los Angeles	MS4 Stormwater[1]	1.90	0.14
Eastern	Monrovia	MS4 Stormwater[1]	1.90	0.14
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	General Industrial Stormwater[1]	1.90	0.14
Eastern	Angeles National Forest	Stormwater[1]	1.90	0.14
Diversion	Los Angeles County Department of Public Works	Water Diversion	1.90	0.14
Near Lake	Arcadia	MS4 Stormwater[1]	1.90	0.14
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.90	0.14
Near Lake	El Monte	MS4 Stormwater[1]	1.90	0.14
Near Lake	Irwindale	MS4 Stormwater[1]	1.90	0.14
Near Lake	County of Los Angeles	MS4 Stormwater[1]	1.90	0.14
Near Lake	Monrovia	MS4 Stormwater[1]	1.90	0.14
Western	Arcadia	MS4 Stormwater[1]	1.90	0.14
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	General Industrial Stormwater[1]	1.90	0.14
Western	Caltrans	State Highway Stormwater[1]	1.90	0.14
Western	County of Los Angeles	MS4 Stormwater[1]	1.90	0.14
Western	Monrovia	MS4 Stormwater[1]	1.90	0.14
Western	Sierra Madre	MS4 Stormwater[1]	1.90	0.14
Western	Angeles National Forest	Stormwater[1]	1.90	0.14

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the Cities of Arcadia, Duarte, Irwindale and Monrovia. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

4.7.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for dieldrin ("Alternative LAs if the Fish Tissue Target is Met") described in Section 4.7.6.2.2. The alternative load allocations will supersede the load allocations in Section 4.7.6.2.1 if the conditions described in Section 4.7.6.2.2 are met.

4.7.6.2.1 Load Allocations

No part of the watershed of Peck Road Park Lake is outside MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to atmospheric deposition of dieldrin. The legacy dieldrin stored in lake sediment is the major cause of impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdictions (County of Los Angeles) should achieve a dieldrin concentration of 0.43 µg/kg dry weight in lake bottom sediments (Table 4-46).

Table 4-46. Load Allocations for Dieldrin in Peck Road Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	0.43

4.7.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 4-46 will be superseded, and the load allocations in Table 4-47 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 4-47, and
3. USEPA does not object to the Regional Board’s determination within 60 days of receiving notice of it.

Table 4-47. Alternative Load Allocations for Dieldrin in Peck Road Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	County of Los Angeles	Lake bottom sediments	1.90

4.7.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

4.7.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate dieldrin, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

4.7.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the dieldrin WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Peck Road Park Lake watershed. USGS Station 11101250, on the Rio Hondo River above the Whittier Narrows Dam, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99^{th} percentile flow for the Rio Hondo (952 cfs) (Wolock, 2003). To estimate the peak flow to Peck Road Park Lake, the 99^{th} percentile flow for the Rio Hondo was scaled down by the ratio of drainage areas (23,564 acres/58,368 acres; Peck Road Park Lake watershed area/Rio Hondo watershed area at the gage). The resulting peak flow estimate for Peck Road Park Lake is 384 cfs. The 99^{th} percentile diverted flow from the San Gabriel River to Peck Road Park Lake is 328 cfs. Therefore, the total peak daily flow rate is 712 cfs.

The event mean concentration of sediment in stormwater (71.7 mg/L) was calculated from the estimated existing watershed sediment load of 990.3 tons/yr (Table 4-14) divided by the total stormflow volume reaching the lake (10,158 ac-ft, Table 4-7). Multiplying the sediment event mean concentration by the 99^{th} percentile peak daily flow (712 cfs) yields a daily maximum sediment load from stormwater of 137.7 tons/d. Applying the wasteload allocation concentration of 0.43 ng dieldrin per dry g of sediment yields the stormwater daily maximum allowable load of 0.054 g/d of dieldrin. This load is associated with the MS4 stormwater permittees and the water diversion. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

4.7.6.6 Future Growth

The manufacture and use of dieldrin is currently banned. Therefore, no additional allowance is made for future growth in the dieldrin TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.8 TRASH IMPAIRMENT

4.8.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Peck Road Park Lake was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Rio Hondo below Spreading Grounds) apply: REC1, REC2, WARM, WILD, MUN, and GWR (personal communication, Regional Board, December 22, 2009). Descriptions of these uses are listed in Section 2 of this TMDL report. Trash can potentially impair the REC1, REC2, and WARM in a variety of ways, including causing toxicity to aquatic organisms, damaging habitat, impairing aesthetics, and impeding recreation.

4.8.2 Numeric Targets

The numeric target is derived from the narrative water quality objective in the Los Angeles Basin Plan (LARWQCB, 1994) for floating material:

> *"Waters shall not contain floating materials, including solids, liquids, foams, and scum, in concentrations that cause nuisance or adversely affect beneficial uses"*

and for solid, suspended, or settleable materials:

> *"Waters shall not contain suspended or settleable material in concentrations that cause nuisance or adversely affect beneficial uses."*

The numeric target for the Peck Road Park Lake Trash TMDL is 0 (zero) trash in or on the water and on the shoreline. Zero trash is defined as no allowable trash discharged into the waterbody of concern, shoreline, and channels. No information has been found to justify any value other than zero that would fully support the designated beneficial uses. Furthermore, court rulings have found that a numeric target of zero trash is legally valid (*City of Arcadia et al. v. Los Angeles Regional Water Quality Control Board et al. (2006) 135 Cal.App.4th 1392*). The numeric target was used to calculate the waste load allocations for point sources and load allocations for nonpoint sources, as described in the following sections of this report.

4.8.3 Summary of Monitoring Data

The existing beneficial uses are impaired by the accumulation of suspended and settled debris. Common items that were observed include plastic bags, plastic pieces, paper items, plastic and glass bottles, Styrofoam, bottle caps, and cigarette butts. Heavier debris has also been transported during storms or dumped on the shoreline or in the lake.

According to California's 2006 303(d) Impaired Waterbodies List, trash is causing water quality problems in Peck Road Park Lake. USEPA and Regional Water Quality Control Board staff confirmed the trash impairment during a site visit to Peck Road Park Lake on March 9, 2009. Staff conducted quantitative trash assessments and documented the trash impairment with photographs. Trash was observed in the lake, along shores and fences surrounding the lake, and at the outlet of storm drains discharging into the lake. Trash of major concern, found on March 9, 2009, included a chicken carcass with numerous egg shells (a biohazard) near the industrial facilities, furniture in the water, a large tattered blanket near the park, and a decomposing animal near Sawpit Wash.

Three quantitative trash assessments were conducted according to the Rapid Trash Assessment protocol which gives each shoreline a numeric score out of a possible 120 points (SWAMP, 2007). Higher scores correspond to cleaner areas, with 120 points representing a clean area. The severity of the trash problem was scored based upon the condition of the following parameters: level of trash, actual number of trash items found, threat to aquatic life, threat to human health, illegal dumping and littering, and accumulation of trash. Trash assessments were conducted within a 100 ft long by 10 ft wide area. If the shoreline was too steep, trash was observed from a distance. Any piece of trash visible from greater than 10 ft away was considered a large piece of trash. The site visit evaluated different land use types surrounding Peck Road Park Lake, including recreational use, industrial businesses, and urban runoff.

4.8.3.1 Peck Road Park

In the park area near the parking lot were roughly 20 picnic tables with barbeque grills and four trash cans. More trash cans were placed near the bathroom but none were observed near the trail. These uncovered trash cans can be a source of trash because animals or wind may transport trash from the cans to the shoreline or lake. People were observed to be fishing, walking around the lake, sitting at picnic tables, and recreating near the water. Approximately 50 birds were observed in the park portion of Peck Road Park Lake. A 100-foot trash assessment was conducted on the beach near the bathroom and parking lot. The area scored a 48/120 with some trash items found in the water. Because this area is more accessible to the public, it might lead to greater picnicking activities and trash littering (Figure 4-10).

Figure 4-10. **Picnic Area near Quantitative Assessment Location #1**

4.8.3.2 Industrial Area

Between 50-300 large pieces of trash were observed along 100 ft of shoreline in the industrial area surrounding Peck Road Park Lake. The area was too steep to appropriately conduct a quantitative trash assessment, but items observed from a distance included plastic bags, milk jugs, a tire, a cooler, metal cable, and industrial scraps. Figure 4-11 shows an example of the trash impairment along the northeastern shore of the lake. A chain link fence surrounds the industrial facilities, which acts as a buffer to trash entering the park. The trash accumulated near the fence does not appear to have been removed for a long period. Many dumpsters at the industrial sites were uncovered or overflowing with debris.

Some companies were notably tidier than others. A transient tarp shelter with over 100 pieces of large visible trash within 100 ft of the shelter was also noted.

Figure 4-11. **Evidence of Dumping near the Industrial Facilities**

4.8.3.3 Sawpit Wash

The second quantitative trash assessment was conducted near the inlet of Sawpit Wash. This area scored a 12/120 due to a heavy accumulation of trash, evidence of trash dumping, and much trash debris found in the water. Water levels in the past were probably higher (i.e., during storm events) as evidenced by trash being stuck higher in branches (Figure 4-12). Specific items found included a semiconductor, pepper spray, a spray paint can, cigarette butts, furniture, and Styrofoam and plastic pieces.

Figure 4-12. **A Bird Lives amongst Trash near the Sawpit Wash Inlet to Peck Road Park Lake**

4.8.3.4 Santa Anita Wash and Adjacent Area to the South

In general, the Santa Anita Wash area has a terraced grading. Visual assessment showed less than five larger pieces of trash per 100 ft. Residential homes, a school, and golf course were tidy and had fences enclosing their property. Dog excrement was observed along the bike trail. Although a large sediment buildup was observed next to a shopping cart, the amount of large visible trash was low near the lake inlet.

The third quantitative trash assessment was completed near Santa Anita Wash, which scored a 49/120. Grading was similar along most of the western shore except for a short beach area which was included in this assessment. Along this portion of the shore, a tree provided a physical space for trash to become entangled (Figure 4-13). Shorelines without any physical obstruction allowed trash to blow directly into the lake. Some trash items were observed in the water.

Locations of the three quantitative trash assessments are shown in Figure 4-14.

Note: Trash accumulates where physical space for entanglement such as branches are present, but likely blows directly into the lake along barren portions of the eastern shore of Peck Road Park Lake.

Figure 4-13. **Trash Accumulates near Santa Anita Wash**

Figure 4-14. **Quantitative Monitoring Locations at Peck Road Park Lake**

During a follow-up visit to Peck Road Park Lake on August 5, 2009, trash was similarly observed in the lake and on the shore. No quantitative surveys were conducted.

In summary, trash was present in and along the shore of Peck Road Park Lake during all visits. The main trash problems were near the park, industrial facilities, and storm drain outfalls.

4.8.4 Source Assessment

The major source of trash in Peck Road Park Lake is due to litter, which is intentionally or accidentally discarded in the lake and watershed. Potential sources can be categorized as point sources and nonpoint sources depending on the transport mechanisms. For example:

1. Storm drains: trash is deposited throughout the watershed and carried to various sections of the lake during and after rainstorms via storm drains. This is a point source.
2. Wind action: trash blown into the lake directly. This is a nonpoint source.
3. Direct disposal: direct dumping or littering into the lake. This is a nonpoint source.

Since the Peck Road Park Lake watershed includes residential areas, open space, parks, roads, and storm drains, both point and nonpoint sources contribute trash to the lake.

4.8.4.1 Point Sources

Trash conveyed by stormwater through storm drains to Peck Road Park Lake is evidenced by trash accumulation at the end of storm drains discharging to the lake.

Based on reports from similar watersheds, the amount and type of trash transported is a function of the surrounding land use. The city of Long Beach recorded trash quantity collected at the mouth of the Los Angeles River; the results suggest total trash amount is linearly correlated with precipitation (Figure 4-15, R^2=0.90, Signal Hill, 2006). A similar study found that the amount of gross pollutants entering the stormwater system is rainfall dependent but does not necessarily depend on the source (Walker and Wong, 1999). The amount of trash entering the stormwater system depends on the energy available to re-mobilize and transport deposited gross pollutants on street surfaces, rather than the amount of available gross pollutants deposited on street surfaces. Where gross pollutants exist, a clear relationship is established between the gross pollutant load in the stormwater system and the magnitude of the storm event. The limiting mechanism affecting the transport of gross pollutants, in the majority of cases, appears to be re-mobilization and transport processes (i.e., stormwater rates and velocities).

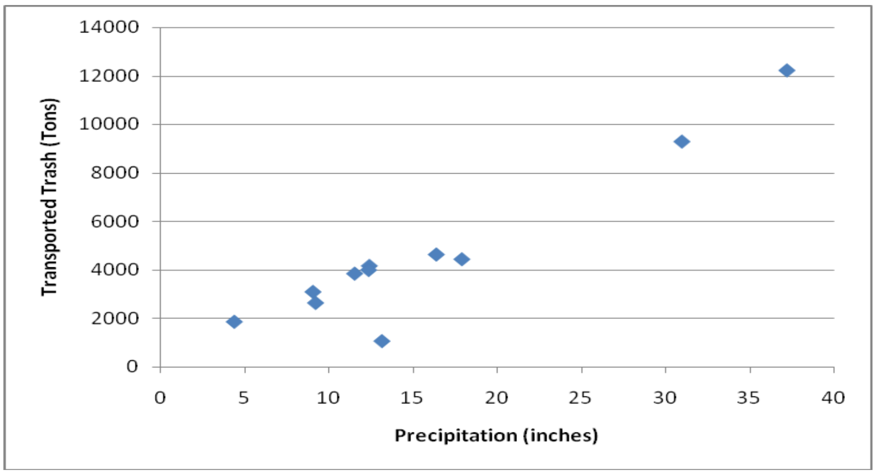

Figure 4-15. **Storm Debris Collection Summary for Long Beach (Signal Hill, 2006)**

In order to estimate trash generation rates, data from a comparable watershed were analyzed. The city of Calabasas completed a study on a Continuous Deflective Separation (CDS) unit installed to catch runoff from Calabasas Park Hills to Las Virgenes. The CDS unit is a hydrodynamic separator that uses vortex settling to remove sediment, trap debris and trash, and separate floatables such as oil and grease. It is assumed that this CDS unit prevented all trash from passing through. The calculated area drained by this CDS Unit is approximately 12.8 square miles. Regional Board staff estimated the waterbody's urbanized area to be 0.10 square miles. The results of this clean-out, which represents approximately half of the 1998-1999 rainy season, were 2,000 gallons of sludgy water and a 64-gallon bag two-thirds full of plastic food wrappers. Part of the trash accumulated in this CDS unit for over half of the rainy season is assumed to have decomposed due to the absence of paper products. Since the CDS unit was cleaned out after slightly more than nine months of use, it was assumed that this 0.10 square mile urbanized area produced a volume of 64 gallons of trash. Therefore, 640 gallons of trash were generated per square mile per year. This estimate is used to determine trash loads.

During the 1998/1999 and 1999/2000 rainy seasons, a Litter Management Pilot Study (LMPS) was conducted by Caltrans to evaluate the effectiveness of several litter management practices in reducing litter discharged from Caltrans stormwater conveyance systems. The LMPS employed four field study sites, each of which was measured with the amount of trash produced when separate BMPs were applied. The average total load for each site normalized by the total area of control catchments was 6,677 gallons/mi^2/yr. Other trash generation rates and studies exist, but the LMPS study is the most applicable to Peck Road Park Lake because of similar land use, population density, and average daily traffic conditions. Therefore, this analysis will use 6,677 gallons/mi^2/yr as the baseline estimate of trash for Caltrans roads.

Table 4-48 shows the current estimated volume of trash deposited within each of the responsible jurisdictions, in gallons per year, assuming a trash generation rate of 6,677 gallons of uncompressed trash/mi^2/yr for Caltrans and a trash generation rate of 640 gallons of uncompressed trash per square mile per year for other jurisdictions. For responsible jurisdictions that are only partially located in the watershed, the square mileage indicated is for the portion in the watershed only. The current loads need to be reduced 100 percent to meet the TMDL target of zero trash.

Table 4-48. Peck Road Park Lake Estimated Point Source Trash Loads

Responsible Jurisdictions	Point Source Area (mi^2)	Current Point Source Trash Load (gal/yr)
Arcadia	3.5	2300
Bradbury	0.79	500
CA DOT (Caltrans)	0.14	950
Duarte	1.7	1100
El Monte	0.077	49
Irwindale	0.78	500
County of Los Angeles	16	10000
Monrovia	13	8000
Sierra Madre	1.1	680

Note: For Caltrans: Current Point Source Trash Load (gal/yr) = Point Source Area (mi^2) * 6,677 (gal/ mi^2/yr). For all other jurisdictions: Current Point Source Trash Load (gal/yr) = Point Source Area (mi^2) * 640 (gal/ mi^2/yr)

4.8.4.2 Nonpoint Sources

Nonpoint source pollution is a source of trash in Peck Road Park Lake. Trash deposited in the lake from nonpoint sources is a function of transport via wind, wildlife, overland flow, and direct dumping.

Few studies have evaluated the relationship between wind strength and movement of trash from land surfaces to a waterbody. Lighter trash with a sufficient surface area to be blown in the wind, such as plastic bags, beverage containers, and paper or plastic food containers, are easily lifted and carried to waterbodies. Also, overland flow carries trash from the shoreline to waterbodies. Transportation of pollutants from one location to another is determined by the energy of both wind and overland stormwater flow.

Existing trash surrounding the lake is the fundamental cause of nonpoint source trash loading. Land use directly surrounding Peck Road Park Lake is low density single-family residential, industrial, and open space and recreational areas. Visitors may intentionally or accidentally discard trash to grass or trails in the park, which initiate the journey of trash to waterbodies via wind or overland water flow. Industrial facilities can contribute nonpoint sources of trash especially if dumpsters are overflowing and trash is not confined within a given area. Varying uses of the park are responsible for different degrees of trash impairment. For example, areas with picnic tables generate more trash than parking lots. Visitation rates are also likely linked to the amount of trash from nonpoint sources.

Table 4-49 summarizes the nonpoint source area and current estimate of nonpoint source trash loads for responsible jurisdictions (the park area and responsible jurisdictions are illustrated in Figure 4-16), assuming a trash generation rate of 640 gallons of uncompressed trash per square mile per year. The current loads need to be reduced 100 percent to meet the TMDL target of zero trash.

Table 4-49. Peck Road Park Lake Estimated Nonpoint Source Trash Loads

Responsible Jurisdictions	Nonpoint Source Area (mi^2)	Current Nonpoint Source Trash Load (gal/year)
Arcadia	0.18	118.0
El Monte	0.0048	3.1
Irwindale	0.00031	0.2
County of Los Angeles	0.00031	0.2
Monrovia	0.048	31

Note: Current Nonpoint Source Trash Load (gal/yr) = Nonpoint Source Area (mi^2) * 640 ($gal/mi^2/yr$)

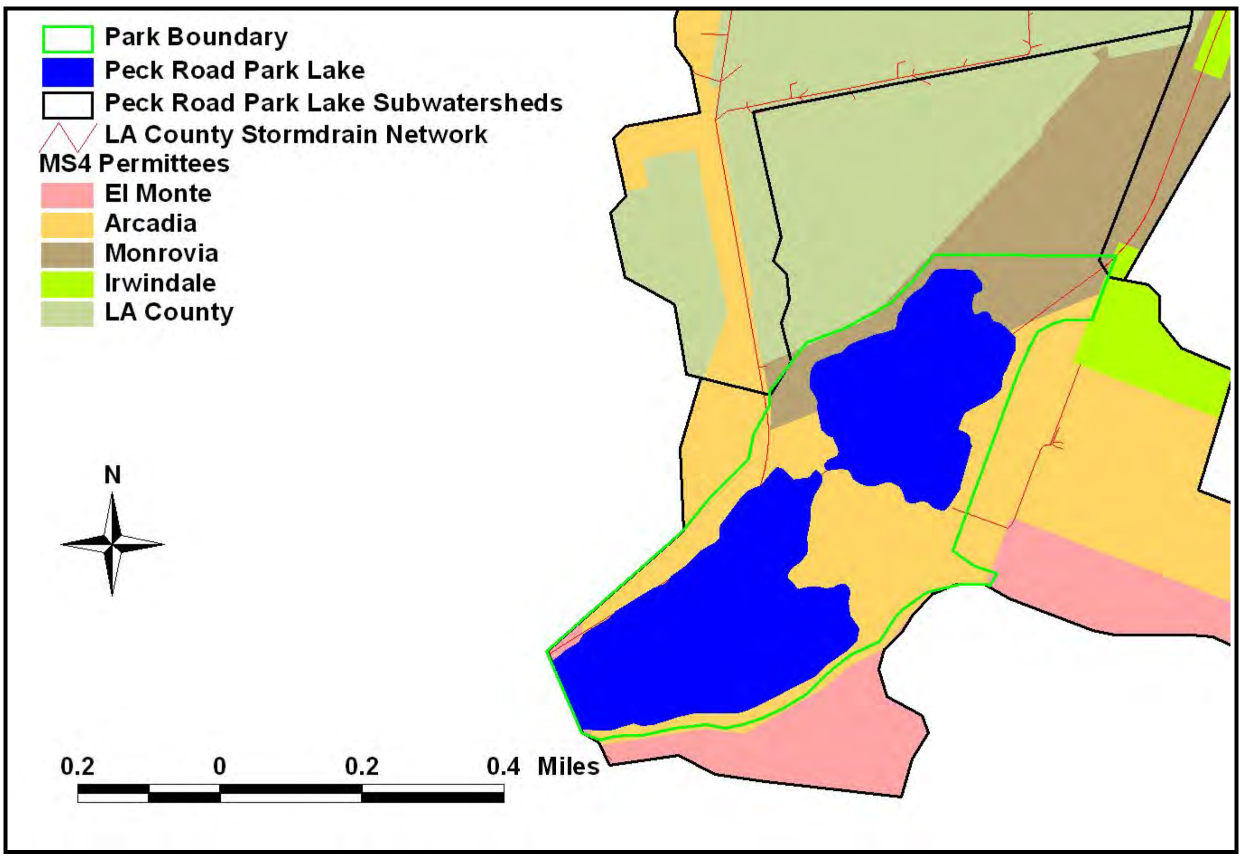

Figure 4-16. **Park Area Associated with Peck Road Park Lake**

4.8.5 Linkage Analysis

These TMDLs are based on numeric targets derived from narrative water quality objectives in the Los Angeles Basin Plan (LARWQCB, 1994) for floating materials and solid, suspended, or settleable materials. The narrative objectives state that waters shall not contain these materials in concentrations that cause nuisance or adversely affect beneficial uses. Since any amount of trash impairs beneficial uses, the loading capacity of Peck Road Park Lake is set to zero allowable trash.

4.8.6 TMDL Summary

Both point sources and nonpoint sources are identified as sources of trash in Peck Road Park Lake. For point sources, water quality standards are attained by assigning waste load allocations (WLAs) to permittees of the Los Angeles County Municipal Separate Storm Sewer System (MS4) Permit and Caltrans (hereinafter referred to as responsible jurisdictions); these WLAs will be implemented through permit requirements. For nonpoint sources, water quality standards are attained by assigning load allocations (LAs) to municipalities and agencies having jurisdictions over Peck Road Park Lake and its subwatershed. These LAs may be implemented through regulatory mechanisms that implement the State Board's 2004 Nonpoint Source Policy such as conditional waivers, waste discharge requirements, or prohibitions.

The TMDL of zero trash requires that current loads are reduced by 100%. Final WLAs and LAs are zero trash (Table 4-50).

Table 4-50. Peck Road Park Lake Trash WLAs and LAs

Peck Road Park Lake	Allocation
Trash WLA	0
Trash LA	0

4.8.6.1 Wasteload Allocations

The geographical boundary contributing to point sources is defined by watershed areas which contain conveyances discharging to the waterbodies of concern. Conveyances include, but are not limited to, natural and channelized tributaries, and stormwater drains and conveyances. Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs).

Wasteload allocations are set to zero allowable trash.

The permits affected are

- County of Los Angeles (includes all cities in Los Angeles County except Long Beach): Board Order 01-182 (as amended by Board Orders R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No 97-03-DWQ, CAS000001

4.8.6.2 Load Allocations

Nonpoint source areas refer to locations where trash may be carried by overland flow, wildlife, or wind to waterbodies. Due to the transportation mechanism by wind, wildlife, and overland flow to relocate trash from land to waterbodies, the nonpoint source area may be smaller than the watershed. In addition, trash loadings frequently occur immediately around or directly into the lake making the load allocation a significant source of trash. According to the study by the city of Calabasas, the trash generation rate is 640 gallons per square mile per year from nonpoint sources areas (including, but not limited to, schools, commercial areas, residential areas, public services, roads, and open space and parks areas). Current trash rates were calculated in the nonpoint source section.

Load allocations (LAs) for nonpoint sources are zero trash. Zero is defined as no allowable trash found in and on the lake, and along the shoreline. According to the Porter-Cologne Act, load allocations may be addressed by the conditional waivers of WDRs, or WDRs. Responsible jurisdictions should monitor the trash quantity deposited in the vicinities of the waterbodies of concern as well as on the waterbody to comply with the load allocation.

The area adjacent to Peck Road Park Lake or defined as nonpoint sources includes parking lots, recreational areas, picnic areas, hiking trails, residential, commercial, industrial, roads, public facilities, and open space areas. Assuming that trash within a reasonable distance from Peck Road Park Lake has a high potential to reach the waterbody, the nonpoint source jurisdictions are Arcadia, El Monte, Irwindale, the county of Los Angeles, and Monrovia. All load allocations are set to zero allowable trash.

4.8.6.3 Margin of Safety

A margin of safety (MOS) accounts for uncertainties in the TMDL analysis. The MOS can be expressed as an explicit mass load, or included implicitly in the WLAs and LAs that are allocated. Because this TMDL sets WLAs and LAs as zero trash, the TMDL includes an implicit MOS. Therefore, an explicit MOS is not necessary.

4.8.6.4 Critical Conditions/Seasonality

Critical conditions for Peck Road Park Lake are based on three conditions that correlate with loading conditions:

- Major storms
- Wind advisories issued by the National Weather Service
- High visitation – On weekends and holidays from May 15 to October 15.

Critical conditions do not affect wasteload or load allocations because zero trash is a conservative target. However, implementation efforts should be heightened during critical conditions in order to ensure that no trash enters the waterbody.

4.8.6.5 Future Growth

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

4.9 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize includes bacteria, nutrients, trash, metals and sediment. Reducing sediment loads would reduce OC pesticides and PCBs delivery to the lake in many instances. More information about this prioritization tool is available at: labmpmethod.org.

If necessary, these TMDLs may be revised as the result of new information (See Section 4.10 Monitoring Recommendations).

4.9.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 4-9, Table 4-18, Table 4-28, Table 4-37, Table 4-46, and Table 4-50 for nutrients, PCBs, chlordane, DDT, dieldrin, and trash, respectively.

4.9.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to MS4, Caltrans, and General Industrial Stormwater permits as well as the San Gabriel River Water Diversion. Wasteload allocations are expressed in Table 4-8, Table 4-16, Table 4-26, Table 4-36, Table 4-44, and Table 4-50 for nutrients, PCBs, chlordane, DDT, dieldrin, and trash, respectively. The concentration and mass-based wasteload allocations will be incorporated into the Caltrans and Los Angeles County MS4 permits. Concentration-based wasteload allocations will be incorporated into the General Industrial Stormwater permit.

4.9.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

Peck Road Park Lake has nutrient-related, chlordane, dieldrin, DDT, PCBs, and trash impairments. While there are some management strategies that would address multiple impairments (i.e., sediment removal BMPs in upland areas), their differences warrant separate implementation and monitoring discussions.

4.9.3.1 Nutrient-Related Impairments

To prevent degradation of this waterbody due to nutrient loading that may be associated with future land use changes, source reduction and pollutant removal BMPs, designed to reduce sediment loading, could be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to Peck Road Park Lake. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

Education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

4.9.3.2 Organochlorine Pesticides and PCBs Impairments

The manufacture and use of chlordane, DDT, dieldrin, and PCBs are currently banned in the U.S. except for certain limited uses of PCBs authorized by USEPA. Therefore, no additional allowances for future growth are needed in the TMDLs. Source control BMPs and pollutant removal are the most suitable courses of action to reduce OC pesticides and PCBs in Peck Road Park Lake. The TMDL calculations performed for each pollutant (described above in their individual sections) indicated internal lake storage as the greatest contributing source and driving factor affecting fish tissue concentrations. Additionally, the current watershed loads are a small fraction of the total loading that would be required to maintain the current sediment concentrations in the lake under steady-state conditions. This indicates that historic loading is causing the elevated fish tissue concentrations. It also suggests that concentrations in fish will decline over time. The most effective remedial actions and/or implementation efforts will focus on addressing the internal lake storage, such as capping or removal of contaminated lake sediments. For chlordane and dieldrin, the current watershed loads may not need any further reduction from current levels.

When properly conducted, removal of contaminated lake sediments, or dredging, can be an effective remediation option. The object of sediment dredging is to eliminate the pollutants that have accumulated in sediments at the lake bottom. Dredging is optimal in waterbodies with known spatial distribution of contamination because sediment removal can focus on problem areas. However, no spatial pattern of pollutant contamination was apparent in Peck Road Park Lake. Removal of the contaminated sediments reduces the pollutants available to in-lake cycling by discontinuing exposure to benthic organisms and reducing water column loading, resulting in reduced bioaccumulation in higher trophic level fish. Potential negative effects of dredging include increased turbidity and lowered dissolved oxygen concentrations in the short term, and disturbance to the benthic community and reactivation of buried sediment and any associated pollutants.

In some cases, sediment capping may be appropriate to sequester contaminated sediments below an uncontaminated layer of sediment, clay, gravel, or media material. Capping is effective in restricting the mobility of OC pesticides and PCBs; however, it is most useful in deep lakes and is likely not a viable solution for some parts of Peck Road Park Lake. Capping implementation should be restricted to areas with sediments that can support the weight of a capped layer, and to areas where hydrologic conditions of the waterbody will not disturb the cap.

The in-lake options for remediation are costly, but would be the only way to achieve full use support in a short timeframe. It is, however, also true that the OC pesticides and PCBs in question are no longer manufactured and will tend to decline in concentration due to dilution by clean sediment and natural attenuation. Natural attenuation includes the chemical, biological, and physical processes that degrade compounds, or remove them from lake sediments in contact with the food chain, and reduce the concentrations and bioavailability of contaminants. These processes occur naturally within the environment and do not require additional remediation efforts; however, the half-lives of OC pesticides and PCBs in the environment are long, and natural attenuation often requires decades before observing significant improvement.

Loading from the watershed can also be expected to decline over time due to natural attenuation and gradual reduction in atmospheric deposition rates. While reductions are called for in watershed loads, these loads are a small fraction of the historic loads already stored in the lakes. Limited sampling has not identified any hotspots of elevated loading under current conditions. It may, however, be necessary to further investigate potential sources of OC pesticides and PCBs loading in the watershed, such as active and abandoned industrial sites, waste disposal areas, former chemical storage areas, and other potential hotspots.

4.9.3.3 Trash Impairment

WLA may be complied with via full capture systems, partial capture systems, nonstructural BMPs, or any other lawful method which meets the target of zero trash. USEPA recommends the installation of full capture systems throughout the watershed. The Linear Radial, Inclined Screen, Baffle Box, and Catch Basin Insert are examples of full capture systems that fulfill the criteria of capturing all trash greater than 5 mm during flow less than the 1-year 1-hour storm. The Linear Radial utilizes a casing with louvers to serve as screens or mesh screen. Flows are routed through the louvers and into a vault. The Inclined Screen uses wedge-wire screen with the slotting perpendicular or parallel to the direction of flow. This device is configured with an influent trough to allow solids to settle. The Baffle Box applies a two-chamber concept: the first chamber utilizes an underflow weir to trap floatable solids, and the second chamber uses a bar rack to capture material. The catch basin has an opening cover screen which is a coarse mesh screen at street level that is paired with a catch basin insert, a 5 mm screen inside the catch basin which filters out smaller trash. USEPA recommends implementation plans be consistent with the Los Angeles River trash TMDL. A monitoring plan should be developed in order to understand the effectiveness of the implementation efforts.

LA may be complied with through the implementation of nonstructural BMPs or any other lawful methods which meet the target of zero trash. A minimum frequency of trash collection and assessment should be established at an interval that prevents trash from accumulating in deleterious amounts in between collections.

Trash should be prevented by providing effective public education about littering impacts. Signs dissuading littering and wildlife feeding along roadways and around the lake are recommended.

A city ban, tax, or incentive program reducing single-use plastic bags, Styrofoam containers, and other commonly discarded items which cannot decompose is recommended (Los Angeles County Department of Public Works, 2007).

Peck Road Park's grounds and facilities are maintained by the Los Angeles County Department of Parks and Recreation. Trash is currently collected and removed from the park twice a week. However, trash is not collected in locations unsafe to reach with court referral labor, such as steep slopes. The Los Angeles County Department of Parks and Recreation should continue to expand the current trash pickup program.

In particular, trash should be collected from all areas of the lake including shorelines with steeper slopes (e.g., northeastern region).

The Los Angeles County Flood Control District is responsible for the trash in the lake. Currently, no method exists to remove trash from the middle of the lake. Therefore, a regular in-lake trash pickup schedule should be implemented, in addition to reporting and scheduling immediate trash collection of dangerous items.

The prevention and removal of trash in Peck Road Park Lake will lead to enhanced aesthetics, improved water quality, and the protection of habitat.

4.10 MONITORING RECOMMENDATIONS

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations will result in compliance with the chlorophyll *a*, fish tissue and trash targets, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be: 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies, fish tissue organochlorine compounds concentrations and trash levels.

4.10.1 Nutrient Related Impairments

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. Deep lakes, such as Peck Road Park Lake, must meet the DO and pH targets in the water column from the surface to 0.3 meters above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the DO and pH targets must be met in the epilimnion, the portion of the water column above the thermocline. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. Wasteload allocations are assigned to stormwater inputs and the San Gabriel River Water Diversion. These sources should be measured near the point where they enter the lakes twice a year for at minimum: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient-response analysis for Peck Road Park Lake indicates that existing levels of nitrogen and phosphorus loading are resulting in attainment of the summer average chlorophyll *a* target concentration of 20 µg/L and are not significantly impacting DO levels in the waterbody. As an antidegradation measure, nitrogen and phosphorus TMDLs are allocated based on existing loading. As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels

(Table 4-7), the targeted concentrations of total phosphorus and total nitrogen may be 0.62 mg-P/L and 4.04 mg-N/L at the outlet of the eastern subwatershed and 0.54 mg-P/L and 3.85 mg-N/L at the outlet of the western subwatershed. Targeted concentrations in the runoff from the near lake subwatershed may be 0.62 mg-P/L and 4.13 mg-N/L. The targeted concentration for San Gabriel River diversion waters may be 0.12 mg-P/L and 3.24 mg-N/L. Assuming average precipitation depths, the targeted concentration of nitrogen in precipitation may be 0.182 mg-N/L. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

4.10.2 Organochlorine Pesticides and PCB Impairments

To assess compliance with the organochlorine compounds TMDLs, monitoring should include monitoring of fish tissue at least every three years as well as once yearly sediment and water column sampling. For the OC pesticides and PCBs TMDLs a demonstration that fish tissue targets have been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total suspended sediments, total PCBs, total chlordane, total DDTs, and dieldrin; as well as the following in-lake sediment parameters: total organic carbon, total PCBs, total chlordane, total DDTs and dieldrin. Environmentally relevant detection limits should be used (i.e. detection limits lower than applicable target), if available at a commercial laboratory. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Wasteload allocations are assigned to stormwater inputs and the San Gabriel River Water Diversion. These sources should be measured near the point where they enter the lakes once a year during a wet weather event. Sampling should be designed to collect sufficient volumes of suspended solids to allow for the analysis of at minimum: total organic carbon, total suspended solids, total PCBs, total chlordane, total DDTs and dieldrin. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken.

WLAs and LAs for each pollutant were assigned to the sediment-associated load from the watershed as well as the lake sediments. The concentration-based WLAs and LAs for chlordane, total DDTs, dieldrin, and total PCBs are 4.14 μg/kg dry weight, 5.28 g/dry g, 0.43 g/dry g, and 1.29 μg/kg dry weight, respectively. The associated reductions from the watershed load needed to meet the WLAs are 45.1 percent for total chlordane, 5.2 percent for total DDTs, and 91.6 percent for total PCBs. A quantitative percent reduction cannot be estimated for dieldrin because all sediment samples were below detection limits (which are greater than the TMDL target concentration); however, the needed reduction appears to be on the order of 53 percent.

4.10.3 Trash

Responsible jurisdictions should monitor the trash quantity deposited in the vicinity of Peck Road Park Lake as well as on the waterbody to comply with the load allocation and to understand the effectiveness of various implementation efforts. Quarterly monitoring using the Rapid Trash Assessment Method is recommended. The trash TMDL target is zero trash; a 100 percent reduction is required.

5 Lincoln Park Lake TMDLs

Lincoln Park Lake (#CAL4051501020000303205453) is listed for ammonia, eutrophication, lead, odor, organic enrichment/low dissolved oxygen, and trash (SWRCB, 2010). This section of the TMDL report describes the impairments and the TMDLs developed to address nutrients (Section 5.2) and trash (Section 5.4). Nutrient load reductions are required to achieve the chlorophyll *a* target; these reductions are also expected to alleviate pH, odor, DO and ammonia problems. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that lead is currently achieving numeric targets at Lincoln Park Lake; therefore, a TMDL is not included for this pollutant. Analyses for lead are presented below (Section 5.3).

5.1 Environmental Setting

Lincoln Park Lake is located in the Los Angeles River Basin (HUC 18070105) within the city of Los Angeles (Figure 5-1). The Urban Lakes Study (UC Riverside, 1994) reported that the area was dedicated for park purposes on August 18, 1883, and that the lake and surrounding park were developed sometime in the early 1890s. The small urban lake has a surface area of 4.9 acres (based on Southern California Association of Governments [SCAG] 2005 land use), an average depth of approximately four feet as estimated from 2009 sampling events and the Urban Lakes Study (UC Riverside, 1994), and a total volume of approximately 19.6 acre-feet (volume calculated from estimated depth and surface area estimated from land use data). The lake is filled primarily with potable water and the park restrooms are connected to the city sewer system. Recreation includes catch and release fishing and there is a fountain near the center of the lake (Figure 5-2). According to the California Department of Fish and Game, trout are stocked periodically (CDFG, 2009). Visitors are not allowed to boat or swim in the lake. Bird feeding is another recreational activity at Lincoln Park Lake, and heavy feeding has been observed during recent fieldwork, likely contributing to larger bird populations. Lake managers use algaecides to control algal growth in the lakes on an as needed basis. Additional characteristics of the watershed are summarized below.

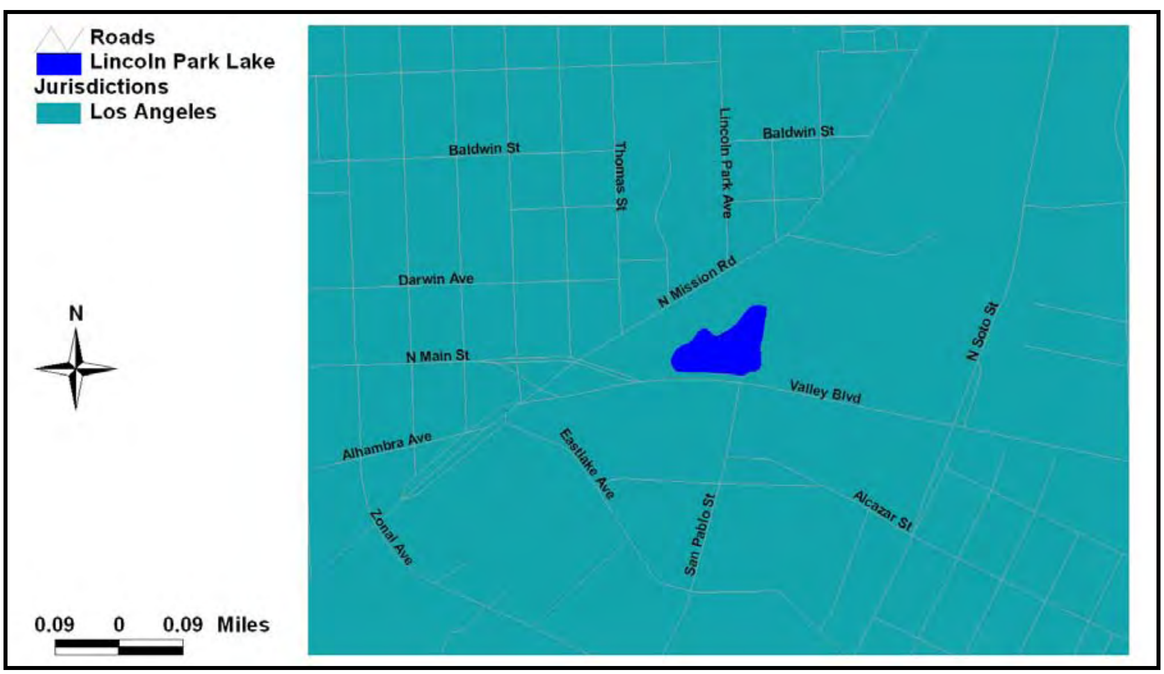

Figure 5-1. Location of Lincoln Park Lake

Figure 5-2. View of Lincoln Park Lake from the West Shore Boat Ramp

5.1.1 Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries

The Lincoln Park watershed is 37.1 acres and ranges in elevation from 104 meters to 132 meters (Figure 5-3). Though the lake appears to be connected to a storm drain network (coverage provided by the county of Los Angeles), this system actually passes under Lincoln Park Lake and does not discharge stormwater to the lake (personal communication, Shoukofe Marashi, city of Los Angeles, to Anna Sofranko, USEPA Region 9, September 25, 2009). Overflow from the lake discharges to the storm drain system (Figure 5-4). The subwatershed boundary for Lincoln Park Lake is comprised only of the surrounding parklands. The TMDL subwatershed boundary was manually delineated based on the digital elevation data. The resulting area is assigned load allocations for TMDL development; the supplemental water additions are assigned wasteload allocations.

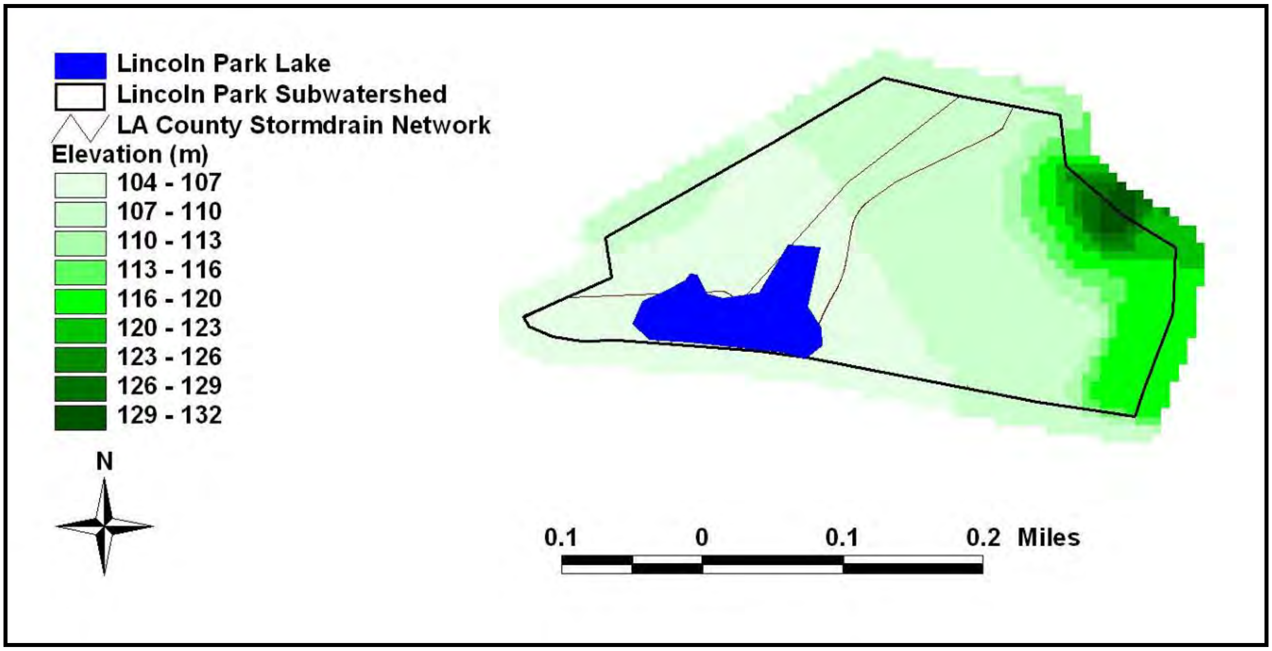

Figure 5-3. Elevation, Storm Drain Networks, and TMDL Subwatershed Boundary for Lincoln Park Lake

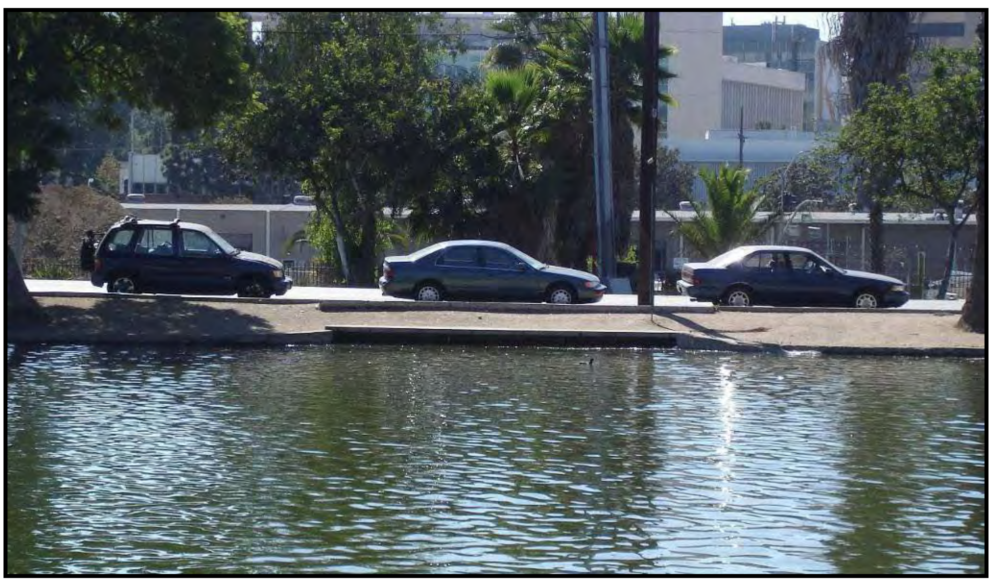

Figure 5-4. Lincoln Park Lake Outflow

5.1.2 MS4 Permittee

Figure 5-5 shows the municipal separate storm sewer system (MS4) stormwater permittee in the Lincoln Park Lake watershed. The watershed is entirely within the city of Los Angeles; however, the lake does not receive drainage from the MS4. The storm drain coverage was provided by the county of Los Angeles and is labeled accordingly.

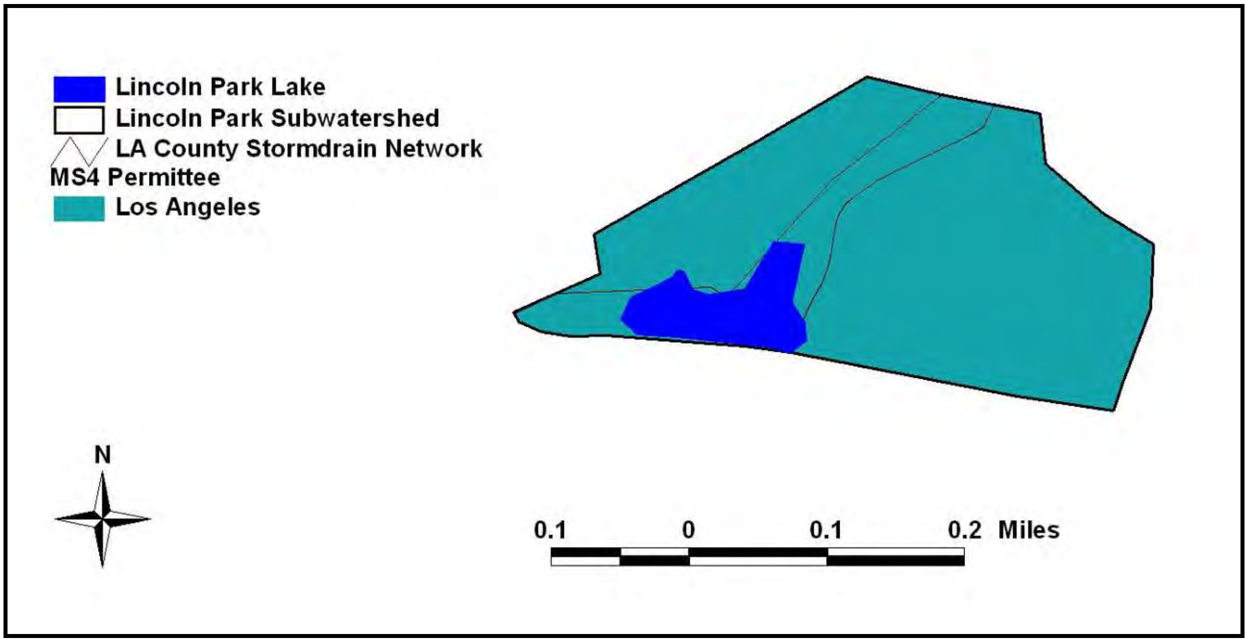

Figure 5-5. MS4 Permittee and the Storm Drain Network for the Lincoln Park Lake Subwatershed

5.1.3 Non-MS4 NPDES Dischargers

As of the writing of these TMDLs, there are no additional (non-MS4) NPDES permits in the Lincoln Park Lake watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

5.1.4 Land Uses and Soil Types

The analysis for the Lincoln Park Lake watershed includes source loading estimates obtained from the Los Angeles River Basin LSPC Model discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the Los Angeles River LSPC model are shown in Figure 5-6. The watershed is comprised of open space and industrial areas. Table 5-1 summarizes the land use areas for the subwatershed.

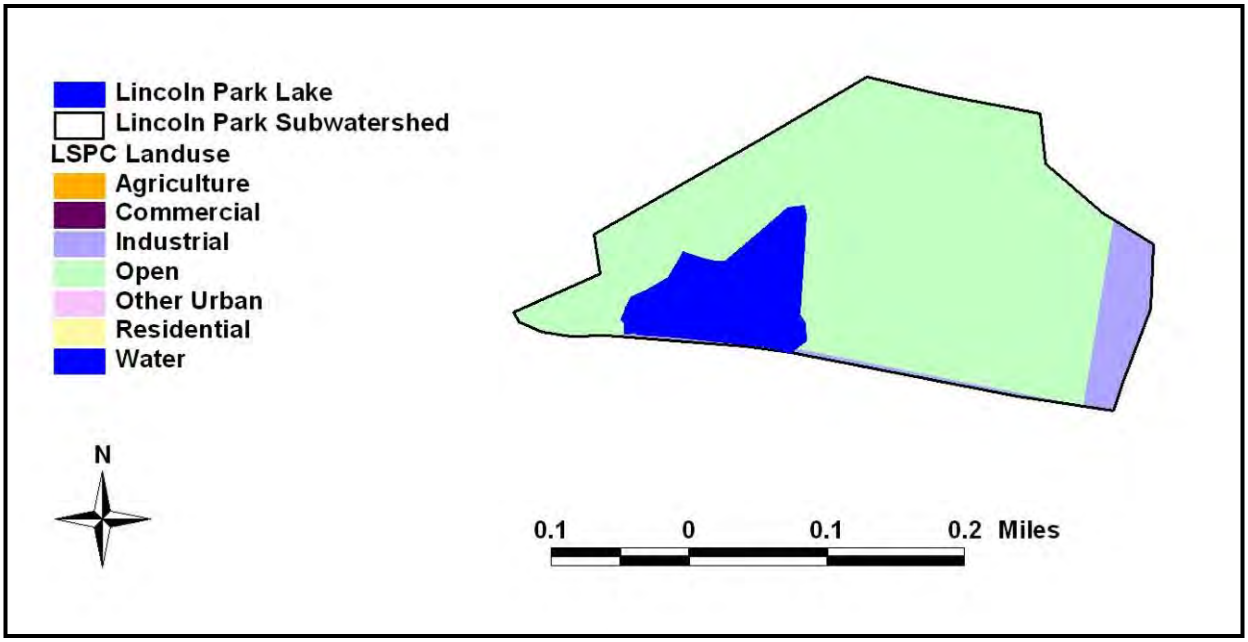

Figure 5-6. LSPC Land Use Classes for the Lincoln Park Lake Subwatershed

Table 5-1. Land Use Areas (ac) Draining from the Lincoln Park Lake Subwatershed

Land Use	Los Angeles
Agriculture	0
Commercial	0
Industrial	3.40
Open	33.7
Other Urban	0
Residential	0
Total	**37.1**

There are no Resource Conservation and Recovery Act (RCRA) contaminated industrial facilities located near the Lincoln Park Lake watershed. Figure 5-7 shows the predominant soils identified by STATSGO in the Lincoln Park Lake subwatershed. The soil type is identified as Urban land-Lithic Xerorthents-Hambright-Castaic (MUKEY 660489), a hydrologic group D soil, which has high runoff potential, very low infiltration rates, and consists chiefly of clay soils.

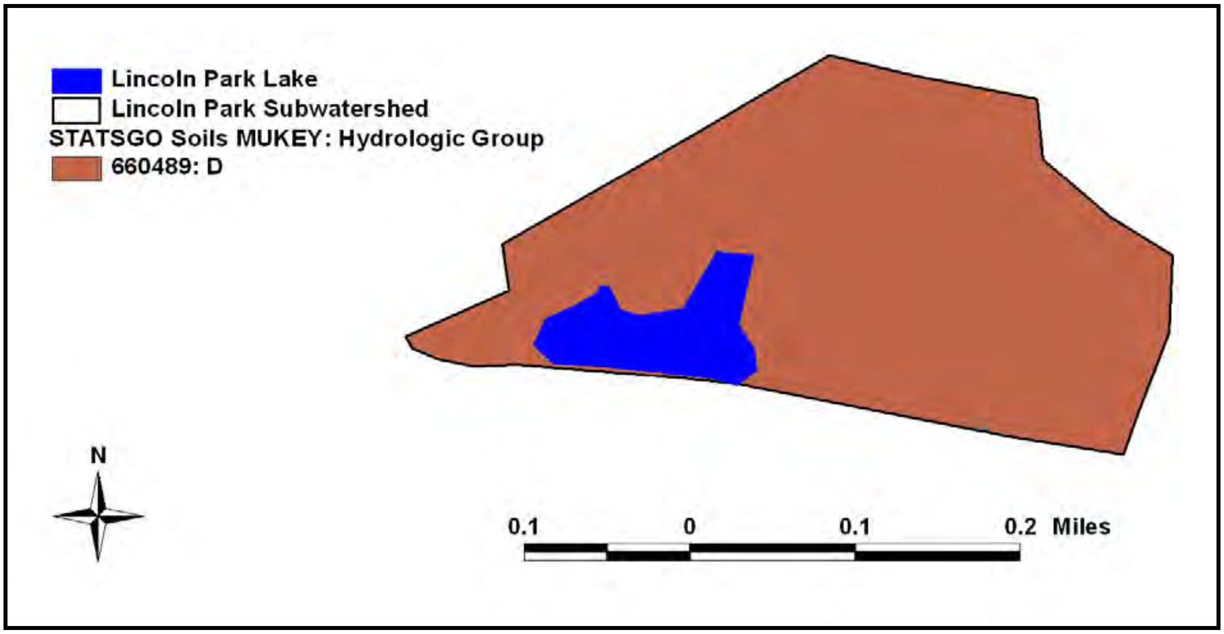

Figure 5-7. STATSGO Soil Types Present in the Lincoln Park Lake Subwatershed

5.1.5 Additional Inputs

Lincoln Park Lake receives supplemental flows from a potable water source to maintain lake levels and irrigate parkland. Two years of monthly usage data were used to estimate the average annual volume pumped into the lake (30.8 ac-ft/yr). An additional 1 foot of potable water is used annually to irrigate 32 acres of surrounding parkland. Some of this irrigation water may reach the lake (5.6 percent of the total irrigation volume is assumed to reach the lake).

5.2 Nutrient Related Impairments

A number of the assessed impairments for Lincoln Park Lake are associated with nutrients and eutrophication. Nutrient related impairments for Lincoln Park Lake include ammonia, eutrophication, odor, and organic enrichment/low dissolved oxygen (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algal photosynthesis removes carbon dioxide from the water, which can lead to elevated pH in poorly buffered systems. Respiration during nighttime hours may cause decreased dissolved oxygen (DO) concentrations. Algal blooms may also contribute to odor problems.

5.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Lincoln Park Lake include REC1, REC2, WARM, WILD, and MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, and WARM uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels and alter biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD and MUN uses could become impaired.

5.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Lincoln Park Lake. The following targets apply to the ammonia, eutrophication, odor, and organic enrichment/low dissolved oxygen impairments (see Section 2 for additional details and Table 5-2 for a summary):

- Ammonia toxicity to aquatic life is caused primarily by the un-ionized form (NH_3), while most ammonia in water is present in the ionic form of ammonium (NH_4^+). The Basin Plan expresses ammonia targets as a function of pH and temperature because these determine the un-ionized fraction. To assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for Lincoln Park Lake in these TMDLs, a median temperature of 19.0 ºC and a 95th percentile pH of 9.0 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 1.32 mg-N/L, the four-day average is 0.91 mg-N/L, and the 30-day average (chronic) target is 0.36 mg-N/L (Note: the median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target varies with the values determined during sample collection.).
- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient (e.g., nitrogen and phosphorous) concentrations in a waterbody can lead to nuisance effects such as overgrowth of algae, odors, and scum. The narrative objective specifies, “waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses.” The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Lincoln Park Lake; however, as described in Tetra Tech (2006), summer (May – September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and is also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target is specified at half of the Secchi depth during the summer (May – September) and annual averaging periods.
- The Basin Plan states that “waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses.”
- The Basin Plan states “at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations.” In addition, the Basin Plan states, “the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges.” Shallow, well-mixed lakes, such as Lincoln Park Lake, must meet the DO target in the water column from the surface to 0.3 meters above the bottom of the lake.
- The Basin Plan states that “the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge.” Shallow, well-mixed lakes, such as Lincoln Park Lake, must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake.

Nitrogen and phosphorus target concentrations are based on simulation of nutrient concentrations and chlorophyll *a* response with the NNE BATHTUB model (see Section 5.2.5). Based on the calibrated model for Lincoln Park Lake, the target nutrient concentrations consistent with achieving the mean chlorophyll *a* target within the lake are:

- 0.88 mg-N/L summer average (May – September) and annual average
- 0.088 mg-P/L summer average (May – September) and annual average

Table 5-2. Nutrient-Related Numeric Targets for Lincoln Park Lake

Parameter	Numeric Target	Notes
Ammonia[1]	1.32 mg-N/L acute (one-hour) 0.91 mg-N/L four-day average 0.36 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.88 mg-N/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model
Total Phosphorous	0.088 mg-P/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model

[1] The median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan.

5.2.3 Summary of Monitoring Data

This section summarizes the in-lake water quality data for Lincoln Park Lake related to the nutrient impairments. Shoreline sampling is not included in this discussion. Appendix G (Monitoring Data) provides more detail regarding sampling events and monitoring results.

In 1992 and 1993, the lake was sampled from a station located in the western half of the lake (UC Riverside, 1994). Sampling occurred from the surface to over 2 meters of depth on 12 sampling days. Total Kjeldahl nitrogen (TKN) ranged from 0.3 mg-N/L to 2.8 mg-N/L. Eight of 28 samples for ammonium were less than the detection limit (0.01 mg-N/L), and the maximum observed ammonium concentration was 1.1 mg-N/L which is less than the acute target assuming the analysis methodology converted all ammonia to ammonium. All nitrite samples were less than the detection limit (0.01 mg-N/L), and 17 of 28 nitrate samples were less than the detection limit (0.01 mg-N/L). The maximum nitrate concentration was 0.3 mg-N/L. Orthophosphate concentrations in 1992 were less than or equivalent to the detection limit (0.01 mg-P/L), while concentrations in 1993 ranged from 0.2 mg-P/L to 0.3 mg-P/L. Total phosphorus was also higher in 1993 with concentrations ranging from 0.2 mg-P/L to 0.5 mg-P/L compared to concentrations in 1992 of which nine samples were less than the detection limit (0.01 mg-P/L), and the maximum observed concentration was 0.2 mg-P/L. pH measurements ranged from 7.7 to 9.1 throughout the water column. Total organic carbon (TOC) ranged from 6.0 mg/L to 14.5 mg/L, with one outlier of 132 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from <1 µg/L to 97 µg/L with an average of 33 µg/L. For this data set, exceedances of the pH and chlorophyll *a* targets were observed.

The Water Quality Assessment Report (LARWQCB, 1996) states that DO was partially supporting the aquatic life use with 78 measurements of dissolved oxygen ranging from 0.1 mg/L to 13.7 mg/L. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonium samples were reported ranging from non-detect to 1.14 mg-N/L which is less than the acute target, but greater than the chronic target for total ammonia N (assuming the analytical method converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples.

In 2009, the city of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples approximately monthly at three locations in Lincoln Park Lake. The nitrate in the lake at all locations and sampling times was below the detection level (0.02 mg-N/L). Nitrite samples ranged from below the detection level (0.02 mg-N/L) to 0.13 mg-N/L. Ammonia samples ranged from below the detection limit (0.05 mg-N/L) to 0.27 mg-N/L, with all observations less than the chronic target. Chlorophyll *a* concentrations ranged from 13 µg/L to 47 µg/L and exceeded the average summer target with an average of 34 µg/L.

Vertical profile data using datasondes were also collected by the city of Los Angeles Bureau of Sanitation during 2003. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification. Dissolved oxygen concentrations ranged from 6.49 mg/L to 9.19 mg/L; pH ranged from 8.16 to 8.72. There were no exceedances of the DO target during these events; 20 percent of pH measurements exceeded the maximum allowable value (all were recorded on one sampling day in July over the entire lake depth).

On March 10, 2009, the Regional Board and USEPA sampled water quality in Lincoln Park Lake at two sites that were accessed by wading in from boat access ramps located on either side of the lake. Samples were collected 1 foot from the surface at each site and the total depth at each site was approximately 2.2 feet. Ammonia concentrations ranged from 1.2 mg-N/L to 1.26 mg-N/L; TKN was 2.2 mg-N/L at both stations. Nitrate and nitrite concentrations were 0.07 mg-N/L and 0.04 mg-N/L, respectively. Orthophosphate concentrations were approximately 0.08 mg-P/L at both stations, and total phosphorus concentrations were approximately 0.126 mg-P/L. Chlorophyll *a* concentrations at both sites were less than the detection limit of 1 µg/L. DO concentrations in the lake generally ranged from 5.9 mg/L to 6.2 mg/L with one reading of 7.0 mg/L from a surface sample. pH ranged from 6.7 to 7.0. The Secchi depth was greater than the total depth at both stations. No exceedances of targets for this lake were observed during this event. Field notes for the March 2009 sampling event indicate the presence of large numbers of birds (100 to 150 pigeons and ducks) and the presence of food left on the boat ramps by visitors to feed the birds.

Profile data were collected at one station on May 10, 2009. The DO concentration ranged from 8.32 to 10.19 mg/L over the depth of the lake. The total depth at this station was 1.7 meters, and the Secchi depth was 0.66 meters. The pH was approximately 9.1 at all depths, which exceeds the target for this parameter, but may not be due to waste discharges so may not represent an exceedance of the standard.

On August 4, 2009, USEPA and the Regional Board collected additional nutrient samples from Lincoln Park Lake. Ammonia, TKN, nitrite, and nitrate were all less than the detection limits of 0.03 mg-N/L, 0.456 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively. Orthophosphate was less than the detection limit (0.0075 mg-P/L), and total phosphorus was 0.182 mg-P/L. The chlorophyll *a* concentration was 27.3 μg/L. The chlorophyll *a* concentration exceeds the target value of 20 μg/L. At the time of this sampling event, the potable water input had been turned off for approximately 2.5 weeks due to water shortages and budget cuts. Field notes also indicate that submerged plants were visible.

In summary, exceedances of the pH and chlorophyll *a* targets have been observed in Lincoln Park Lake. The 1994 Urban Lakes Study suggested that the lake liner and aeration system appear to be effective in suppressing excessive algal growth in the lake; however, the lake did not meet the chlorophyll *a* target during that study (UC Riverside, 1994) nor during more recent sampling. DO concentrations do appear to be successfully managed by the aeration system and annual averages were greater than the target of 7 mg/L. No odors were observed during four recent sampling events by USEPA and/or the Regional Board. There were no exceedances of the acute or chronic ammonia criteria during any recent sampling events with associated pH and temperature measurements. The nutrient TMDLs for Lincoln Park Lake presented in Section 5.2.6 account for summer season critical conditions by assessing loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 μg/L. These reductions in nutrient loading are expected to alleviate any pH, odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication.

5.2.4 Source Assessment

The source assessment for Lincoln Park Lake includes load estimates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading) including irrigation (5.6 percent of the total irrigation volume is assumed to reach the lake), potable water used for supplemental water additions to the lake (Appendix F, Dry Weather Loading), and atmospheric deposition (Appendix E, Atmospheric Deposition). In addition to these sources, there are other sources of loading to Lincoln Park Lake associated with the parkland area for which loading estimates were not available (Appendix F, Dry Weather Loading). These include excessive fertilization relative to product recommendations, internal loading from lake sediments, natural wildlife populations, excessive bird populations caused by the improper disposal of food waste (Figure 5-8), and pet wastes. Loads in the additional parkland loading category were quantified using the NNE BATHTUB model by increasing the inputs until simulated concentrations of total phosphorus and total nitrogen matched those observed (see Section 5.2.5). For this waterbody, the additional parkland loading comprises 56 percent of the total phosphorus load and 35 percent of the total nitrogen load. All existing loads to Lincoln Park Lake are summarized in Table 5-3.

Precise bird counts for Lincoln Park Lake are not available; however, field notes indicate excess bird populations which are likely a significant portion of the nutrient loading associated with additional parkland areas. At Echo Park Lake, total phosphorus and total nitrogen loads of 78 lb-P/yr and 780 lb-N/yr were estimated for the approximately 1,000 birds observed to reside at that lake (Black and Veatch, 2010). The bird population at Lincoln Park Lake is likely one-half to one-quarter of that. Thus total phosphorus loads due to the bird population at Lincoln Park Lake likely range from 19.5 lb-P/yr to 39 lb-P/yr; total nitrogen loads range from 195 lb-N/yr to 390 lb-N/yr. The estimated loading from the resident bird population at Lincoln Park Lake is greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the bird population at Lincoln Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading. If the resident bird population is reduced to 100 birds their total phosphorus loads would be only 7.8 lb-P/yr and 78 lb-N/yr.

Table 5-3. Summary of Average Annual Flows and Nutrient Loading to Lincoln Park Lake

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
City of Los Angeles	Runoff	4.15	4.72 (13.6)	46.1 (23.3)
City of Los Angeles	Supplemental Water Additions (Potable Water)	30.8	9.88 (28.4)	74.6 (37.7)
City of Los Angeles	Parkland Irrigation	1.80	0.577 (0.02)	4.36 (2.20)
City of Los Angeles	Additional Parkland Loading	NA	19.6 (56.3)	70 (35.4)
	Atmospheric Deposition (to the lake surface)*	6.25	NA	3.10 (1.57)
Total		**43.1**	**34.8**	**198**

* Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

Figure 5-8. Inappropriate Bird Feeding Maintains an Excessive Bird Population at Lincoln Park Lake

5.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on Lincoln Park Lake, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE)

BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediment. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Lincoln Park Lake, the following inputs apply: surface area of 4.9 acres, average depth of 4 ft, and volume of 19.6 ac-ft. Based on the turnover ratio for the limiting nutrient for this lake (nitrogen) (Walker, 1987), the annual averaging period is most appropriate (i.e., annual loads are input to the model rather than summer season loads).

The NNE BATHTUB Tool was calibrated to average summer season water quality data observed over twice the Secchi depth (2*0.66 m = 1.32 m). Both nitrogen and phosphorus concentrations were underpredicted when the calibration factors were adjusted within normal range. To predict the average summer concentrations of total phosphorus (0.14 mg-P/L) and total nitrogen (1.29 mg-N/L), loads from additional parkland sources were increased to 23.5 lb-P/yr and 70 lb-N/yr, respectively with calibration factors on the sedimentation rates set to 1. The amount of the additional parkland loading of phosphorus due to internal recycling was calculated with the method discussed in Appendix A (Nutrient TMDL Development) and is 3.93 lb-P/yr. This portion of the phosphorus load was subtracted out of the additional parkland sources category, and the model was recalibrated with a loading of 19.6 lb-P/yr. The resulting calibration factor on the net phosphorus settling rate is 0.82 which allows the model to account for internal loading implicitly. Though internal loading is not explicitly assigned a load allocation, reductions in external loading of phosphorous will ultimately result in reductions of internal cycling processes. Internal loading of nitrogen was not calculated because 1) internal loading is typically insignificant relative to external loading, and 2) empirical relationships for the estimation of internal nitrogen loading have not been developed. Thus, the additional parkland source loading and calibration factor for nitrogen were not changed. To simulate the average observed chlorophyll *a* concentration, the calibration factor on concentration was set to 0.62 for a predicted concentration of 32.6 µg/L.

5.2.6 TMDL Summary

A waterbody’s loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Following calibration of the NNE BATHTUB Tool (Section 5.2.5), the allowable loading combinations of nitrogen and phosphorus were calculated using Visual Basic's GoalSeek function (Appendix A, Nutrient TMDL Development). The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 0.88 mg-N/L summer average (May – September) and annual average
- 0.088 mg-P/L summer average (May – September) and annual average

The loading capacities for total nitrogen and total phosphorus are 120 lb-N/yr and 17.0 lb-P/yr, respectively. These loading capacities can be further broken down into wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load (divided among WLAs and LAs) is 54.5 percent of the existing load of 198 lb-N/yr, or 108 lb-N/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. WLAs and LAs are developed assuming an equal percent load reductions in all sources. The resulting TMDL equation for total nitrogen is then:

120 lb-N/yr = 40.7 lb-N/yr + 67.4 lb-N/yr + 12.0 lb-N/yr

For total phosphorus, the allocatable load (divided among WLAs and LAs) is 44.0 percent of the existing load of 34.8 lb-P/yr, or 15.3 lb-P/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. The resulting TMDL equation for total phosphorous is then:

17.0 lb-P/yr = 4.34 lb-P/yr + 10.9 lb-P/yr + 1.70 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on simulation of allowable loads with the NNE BATHTUB model (see Section 5.2.5). These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

5.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs and alternative WLAs for total phosphorous and total nitrogen.

The alternative WLAs will be effective and supersede the WLAs listed in Table 5-4 if the conditions described in Section 5.2.6.1.2 are met.

Under either wasteload allocation scheme responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on the Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

5.2.6.1.1 Wasteload Allocations

There are no MS4 discharges to Lincoln Park Lake and no other (non-MS4) permitted dischargers in the watershed. The supplemental water addition used to maintain the lake level is the only source of nutrient loading to Lincoln Park Lake that is assigned a WLA (Table 5-4). Total phosphorus WLAs represent a 56.0 percent reduction in existing loading, and total nitrogen WLAs represent a 45.5 percent reduction in existing loading. These loading values (in pounds per year) represent the TMDLs wasteload allocations. The wasteload allocations must be met at the point of discharge.

Table 5-4. Wasteload Allocations of Phosphorus and Nitrogen Loading to Lincoln Park Lake

Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Wasteload Allocation Total Phosphorus[1] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Wasteload Allocation Total Nitrogen[1] (lb-N/yr)
City of Los Angeles	Supplemental Water Additions	9.88	4.34	74.6	40.7
Total		**9.88**	**4.34**	**74.9**	**40.7**

[1] The wasteload allocation must be met at the point of discharge.

5.2.6.1.2 Alternative "Approved Lake Management Plan Wasteload Allocations"

Concentration-based WLAs not exceeding the concentrations listed in Table 5-5 are effective and supersede corresponding WLAs for the City of Los Angeles in Table 5-4 if:

1. The City of Los Angeles requests that concentration-based wasteload allocations not to exceed the concentrations established in Table 5-5 apply to it,
2. The City of Los Angeles provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 5-2. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by

improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The City of Los Angeles may use monitoring data and modeling to show that the water quality criteria, targets and requested WLAs will be met,

3. The Regional Board Executive Officer approves the request and applies concentration-based wasteload allocations for total nitrogen and total phosphorus. These wasteload allocations are not to exceed the concentrations in Table 5-5 as a summer average (May-September) and annual average, and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

The concentration-based WLAs must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 5-5. Alternative Wasteload Allocations of Phosphorus and Nitrogen in Lincoln Park Lake if an Approved Lake Management Plan Exists

Responsible Jurisdiction	Input	Maximum Allowable Wasteload Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Wasteload Allocation Total Nitrogen[1] (mg-N/L)
City of Los Angeles	Supplemental Water Additions	0.1	1.0

[1] The concentration-based wasteload allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

5.2.6.2 Load Allocations

These TMDLs establish load allocations (LAs) and alternative LAs for total phosphorous and total nitrogen. The alternative LAs will be effective and supersede the LAs listed in Table 5-6 if the conditions described in Section 5.2.6.2.2 are met.

5.2.6.2.1 Load Allocations

There are no storm drains that discharge runoff flows into Lincoln Park Lake. Therefore, all loads associated with the surrounding drainage area are assigned LAs (Table 5-6). Atmospheric deposition and additional parkland loading are also assigned LAs. Total phosphorus LAs represent a 56.0 percent reduction in existing loading, and total nitrogen LAs represent a 45.5 percent reduction in existing loading. LAs are provided for each responsible jurisdiction and input and must be met at the point of discharge. These loading values (in pounds per year) represent the TMDLs load allocations.

Table 5-6. Load Allocations of Phosphorus and Nitrogen Loading to Lincoln Park Lake

Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Load Allocation Total Phosphorus[1] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Load Allocation Total Nitrogen[1] (lb/yr)
City of Los Angeles	Runoff	4.72	2.07	46.1	25.1
City of Los Angeles	Parkland Irrigation	0.577	0.254	4.36	2.38
City of Los Angeles	Additional Parkland Loading	19.6	8.62	70	38.2
	Atmospheric Deposition (to the lake surface)[2]	NA	NA	3.1	1.69
Total		**24.9**	**10.9**	**124**	**67.4**

[1] Each load allocation must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

5.2.6.2.2 Alternative "Approved Lake Management Plan Load Allocations"

Concentration-based load allocations not exceeding the concentrations listed in Table 5-7 are effective and supersede corresponding load allocations for the City of Los Angeles in Table 5-6 if:

1. The City of Los Angeles requests that concentration-based load allocations not to exceed the concentrations established in Table 5-7 apply to it;
2. The City of Los Angeles provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 5-2. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The City of Los Angeles may use monitoring data and modeling to show that the water quality criteria, targets and requested load allocations will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based load allocations for total nitrogen and total phosphorus. These load allocations are not to exceed the concentrations in Table 5-7 as a summer average (May-September) and annual average; and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based LA must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 5-7. Alternative Load Allocations of Phosphorus and Nitrogen Loading to Lincoln Park Lake if an Approved Lake Management Plan Exists

Responsible Jurisdiction	Input	Maximum Allowable Load Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Load Allocation Total Nitrogen[1] (mg-N/L)
City of Los Angeles	Runoff	0.1	1.0
City of Los Angeles	Parkland Irrigation	0.1	1.0
City of Los Angeles	Additional Parkland Loading	0.1	1.0

[1] Each concentration-based load allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

5.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. To account for the uncertainties concerning the relationship between nutrient loading and the resultant in-lake chlorophyll *a* an explicit MOS is included in these TMDLs. This explicit MOS is set at 10 percent of the loading capacity for total phosphorus and total nitrogen.

5.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are expected to alleviate any odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication. These TMDLs therefore protect for critical conditions.

5.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). The majority of nutrient loading to Lincoln Park Lake comes from the supplemental water additions. Estimated maximum daily loads from this source are determined. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

The maximum daily loads from the supplemental water additions were calculated from the largest metered monthly water volume and the long-term average concentration consistent with meeting the

TMDLs. For the supplemental water additions, the allowable loads of nitrogen and phosphorus are 40.7 lb-N/yr and 4.34 lb-P/yr (Table 5-4), respectively. The volume of water discharged from this source is approximately 30.8 ac-ft/yr. The allowable concentrations from this source are 0.486 mg-N/L and 0.052 mg-P/L. The maximum metered monthly flow rate is 5.81 ac-ft/mo or 0.187 ac-ft/d (5.81 ac-ft/mo divided by 31 d/mo). The maximum daily nutrient loads from this source are 0.247 lb-N/d and 0.026 lb-P/d.

As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

5.2.6.6 Future Growth/Conditions

The Lincoln Park Lake watershed is comprised entirely of parkland with a small section of adjacent industrial area. It is not likely that the watershed will be developed and it is expected to remain as open space. No load allocation has been set aside for future growth, and it is unlikely that any dischargers will be permitted in the watershed.

The city of Los Angeles would like to use a reclaimed/recycled water source to supplement water levels at Lincoln Park Lake instead of the potable water source that is currently used. Recent monitoring data performed by the City indicate that total nitrogen and total phosphorus concentrations from the potential reclaimed water source are approximately 8.82 mg-N/L and 1.93 mg-P/L. If the City were to use this reclaimed source, this would add an additional 664 lb-N/yr and 152 lb-P/yr relative to existing conditions. Unless BMPs are implemented at the lake to provide treatment of the reclaimed water source, the use of this source will not meet the requirements of these TMDLs. It is advisable that alternative solutions and BMPs be investigated during the implementation planning for this lake.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

5.3 LEAD IMPAIRMENT

Lincoln Park Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 40 additional samples (11 wet weather) between October 2008 and December 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 40 samples (Appendix G, Monitoring Data). USEPA also collected one sediment sample in September 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Lincoln Park Lake meets lead water quality standards and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Lincoln Park Lake not be identified as impaired by lead in California’s next 303(d) list.

5.4 TRASH IMPAIRMENT

5.4.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Lincoln Park Lake include REC1, REC2, WARM, and WILD. Descriptions of these uses are listed in Section 2 of this TMDL report. Trash can potentially impair the REC1, REC2, WARM, and WILD in a variety of ways, including causing toxicity to aquatic organisms, damaging habitat, impairing aesthetics, and impeding recreation.

5.4.2 Numeric Targets

The numeric target is derived from the narrative water quality objective in the Los Angeles Basin Plan (LARWQCB, 1994) for floating material:

> *"Waters shall not contain floating materials, including solids, liquids, foams, and scum, in concentrations that cause nuisance or adversely affect beneficial uses";*

and for solid, suspended, or settleable materials:

> *"Waters shall not contain suspended or settleable material in concentrations that cause nuisance or adversely affect beneficial uses."*

The numeric target for the Lincoln Park Lake Trash TMDL is 0 (zero) trash in or on the water and on the shoreline. Zero trash is defined as no allowable trash discharged into the waterbody of concern, shoreline, and channels. No information has been found to justify any value other than zero that would fully support the designated beneficial uses. Furthermore, court rulings have found that a numeric target of zero trash is legally valid (*City of Arcadia et al. v. Los Angeles Regional Water Quality Control Board et al. (2006) 135 Cal.App.4th 1392*). The numeric target was used to calculate the waste load allocations for point sources and load allocations for nonpoint sources, as described in the following sections of this report.

5.4.3 Summary of Monitoring Data

The existing beneficial uses are impaired by the accumulation of suspended and settled debris. Common items that have been observed include plastic bags, plastic pieces, paper items, Styrofoam, bottle caps, and cigarette butts.

According to California's 2008-2010 303(d) Impaired Waterbodies list, trash is causing water quality problems in Lincoln Park Lake. USEPA and Regional Water Quality Control Board staff confirmed the trash impairment during a site visit to Lincoln Park Lake on March 9, 2009. Staff conducted quantitative trash assessments and documented the trash impairment with photographs. Trash was observed in the lake and along the shorelines.

Although some trash management practices were in place at Lincoln Park, improvements could be added. Many uncovered trash cans were observed throughout the park so trash may be transported from the cans via animals or wind; for example, two open dumpsters were observed near the school. Field staff did not observe any fences between the street and the lake, and between neighboring residences and the street. Over 100 birds were observed in and near this small lake, leading to unnaturally large amounts of bird droppings in and around the lake. The cause of the unnaturally large bird population is likely due to people feeding the birds and birds eating from uncovered trash cans.

Trash observed in the lake was predominantly found in sharp corners of the lake where the water was stagnant (Figure 5-9).

Figure 5-9. Scum and Trash Accumulate in the Sharp Corners of Lincoln Park Lake

Two quantitative trash assessments were conducted according to the Rapid Trash Assessment protocol which gives each shoreline a numeric score out of a possible 120 points (SWAMP, 2007). Higher scores correspond to cleaner areas, with 120 points representing a clean area. The severity of the trash problem was scored based upon the condition of the following parameters: level of trash, actual number of trash items found, threat to aquatic life, threat to human health, illegal dumping and littering, and accumulation of trash. Trash assessments were conducted within a 100 feet long by 10 feet wide area. The site visit evaluated different land use types surrounding Lincoln Park Lake, including recreational uses near a roadway and near picnic tables.

5.4.3.1 Near Valley Boulevard

The trash assessment conducted on the shore near Valley Boulevard (Figure 5-10) scored 91/120. Field staff observed two uncovered trash cans which may lead to trash transported by animals or wind. This is a highly accessible portion of the lake due to its close proximity to on-street parking and a sidewalk. Trash is likely transported from the road and people picnicking along the shore. Some trash was found in the water but no accumulation of trash was observed.

Figure 5-10. Shoreline Along Valley Boulevard

5.4.3.2 Picnic Tables

The second trash assessment was conducted on the eastern shore near the palm tree island, a park path, and picnic tables (Figure 5-11). This area scored a 93/120 and may have been recently cleaned due to the presence of an orderly pile of trash along the shore and almost empty trash cans. Trash is likely transported from people littering in the picnic area and along the path, and from uncovered trash cans. Some items were found in the water.

Figure 5-11. Location of the Second Quantitative Trash Assessment with Trash Cans and Picnic Tables Nearby

5.4.3.3 After School Program

An after school program organized by a non-profit organization, Plaza De La Raza, takes place on the northern shore. The school is completely fenced off and no trash was observed within the school yard's deck area. The school is an unlikely source of trash.

5.4.3.4 Wildlife Feeding

Bird feeding was observed the following day, March 10, 2009. Large piles of rice were observed near Valley Boulevard and on the eastern boat ramp (Figure 5-12). This food was likely left by visitors to feed the birds. Human food is unhealthy for wildlife and the massive amounts discarded can cause an overabundance of birds to inhabit this area. An unnaturally large bird population leads to greater excrement quantities which add to the nutrient problem in the lake.

Figure 5-12. Food is Trash and Encourages an Overabundance of Birds to Live in the Area

Locations of the quantitative monitoring sites are shown in the map below (Figure 5-13).

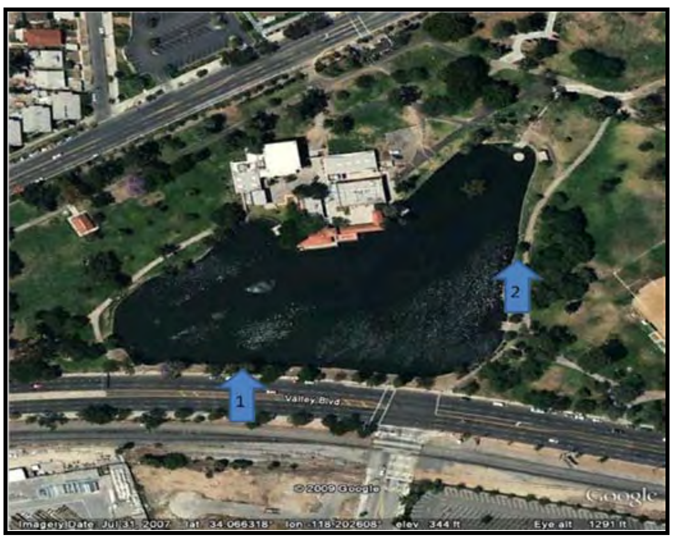

Figure 5-13. Quantitative Trash Assessment Locations

During a follow-up visit to Lincoln Park Lake on August 4, 2009, trash was similarly observed in the lake and on the shore. No quantitative surveys were conducted.

In summary, trash was present in and along the shore of Lincoln Park Lake during all visits. The prevalence of trash was evenly distributed around the lake. The main trash problems were caused by feeding wildlife and small trash items, such as cigarette butts.

5.4.4 Source Assessment

The major source of trash in Lincoln Park Lake results from litter, which is intentionally or accidentally discarded to the lake and watershed. Potential sources can be categorized as nonpoint sources with the following transport mechanisms:

1. Wind action: trash that is blown into the lake directly.
2. Direct disposal: direct dumping or litter into the lake.

Since the Lincoln Park Lake watershed primarily includes open space and parks, only nonpoint sources contribute trash to the lake.

5.4.4.1 Point Sources

There are no point sources of trash to Lincoln Park Lake. The area directly surrounding the waterbody is designated as nonpoint source. Therefore, it is included in the load allocation section.

5.4.4.2 Nonpoint Sources

Based on reports from similar watersheds, the amount and type of trash transported is a function of the surrounding land use. The city of Long Beach recorded trash quantity collected at the mouth of the Los Angeles River; the results suggest total trash amount is linearly correlated with precipitation (Figure 5-14, R^2=0.90, Signal Hill, 2006). A similar study found that the amount of gross pollutants entering the stormwater system is rainfall dependent but does not necessarily depend on the source (Walker and Wong, 1999). The amount of trash entering the stormwater system depends on the energy available to re-mobilize and transport deposited gross pollutants on street surfaces, rather than the amount of available gross pollutants deposited on street surfaces. Where gross pollutants exist, a clear relationship is established between the gross pollutant load in the stormwater system and the magnitude of the storm event. The limiting mechanism affecting the transport of gross pollutants, in the majority of cases, appears to be re-mobilization and transport processes (i.e., stormwater rates and velocities). In order to estimate trash generation rates, data from a comparable watershed was analyzed.

The city of Calabasas completed a study on a Continuous Deflective Separation (CDS) unit installed to catch runoff from Calabasas Park Hills to Las Virgenes. The CDS unit is a hydrodynamic separator that uses vortex settling to remove sediment, trap debris and trash, and separate floatables such as oil and grease. It is assumed that this CDS unit prevented all trash from passing through. The calculated area drained by this CDS Unit is approximately 12.8 square miles. Regional Board staff estimated the waterbody's urbanized area to be 0.10 square miles. The results of this clean-out, which represents approximately half of the 1998-1999 rainy season, were 2,000 gallons of sludgy water and a 64-gallon bag two-third full of plastic food wrappers. Part of the trash accumulated in this CDS unit for over half of the rainy season is assumed to have decomposed due to the absence of paper products. Since the CDS unit was cleaned out after slightly more than nine months of use, it was assumed that this 0.10 square mile urbanized area produced a volume of 64 gallons of trash. Therefore, 640 gallons of trash were generated per square mile per year. This estimate is used to determine trash loads.

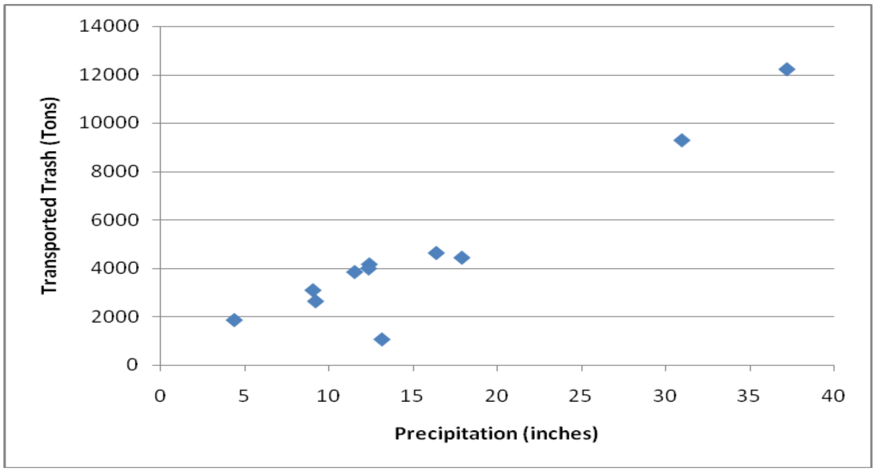

Figure 5-14. Storm Debris Collection Summary for Long Beach (Signal Hill, 2006)

Nonpoint source pollution is the primary source of trash in Lincoln Park Lake. Trash deposited in the lake from nonpoint sources is a function of transport via wind, wildlife, and overland flow, and direct dumping.

Few studies have evaluated the relationship between wind strength and movement of trash from land surfaces to a waterbody. Lighter trash with a sufficient surface area to be blown in the wind, such as plastic bags, beverage containers, and paper or plastic food containers, are easily lifted and carried to waterbodies. Also, overland flow carries trash from the shoreline to waterbodies. Transportation of pollutants from one location to another is determined by the energy of both wind and overland stormwater flow.

Existing trash surrounding the lake is the fundamental cause of nonpoint source trash loading. The land use directly surrounding Lincoln Lake is recreational and educational. Visitors may intentionally or accidentally discard trash to the grass or trails in the park, which initiate the journey of trash to waterbodies via wind or overland water flow. The after-school facilities can contribute nonpoint source trash especially if dumpsters are overflowing and trash is not confined within a given area. Varying uses of the park are responsible for different degrees of trash impairment. For example, areas with picnic tables generate more trash than parking lots. Visitation rates are also likely linked to the amount of trash from nonpoint sources.

Table 5-8 summarizes the nonpoint source area and current estimate of nonpoint source trash loads for responsible jurisdictions, assuming a trash generation rate of 640 gallons of uncompressed trash per square mile per year. The current loads need to be reduced 100% to meet the TMDL target of zero trash.

Table 5-8. Lincoln Park Lake Estimated Nonpoint Source Trash Loads

Responsible Jurisdictions	Nonpoint Source Area (Mile^2)	Current Nonpoint Source Trash Load (Gal/year)
City of Los Angeles	0.058	37

Note: Current Nonpoint Source Trash Load (gal/yr) = Nonpoint Source Area (mi2) * 640 (gal/ mi2/yr)

5.4.5 Linkage Analysis

These TMDLs are based on numeric targets derived from narrative water quality objectives in the Los Angeles Basin Plan (LARWQCB, 1994) for floating materials and solid, suspended, or settleable materials. The narrative objectives state that waters shall not contain these materials in concentrations that cause nuisance or adversely affect beneficial uses. Since any amount of trash impairs beneficial uses, the loading capacity of Lincoln Park Lake is set to zero allowable trash.

5.4.6 TMDL Summary

Nonpoint sources are identified as the only source of trash in Lincoln Park Lake. For nonpoint sources, water quality standards are attained by assigning load allocations (LAs) to municipalities and agencies having jurisdictions over Peck Road Park Lake and its subwatershed. These LAs may be implemented through regulatory mechanisms that implement the State Board's 2004 Nonpoint Source Policy such as conditional waivers, waste discharge requirements, or prohibitions.

The TMDL of zero trash requires that current loads are reduced by 100 percent. Final LAs are zero trash (Table 5-9).

Table 5-9. Lincoln Park Lake Trash LAs

Lincoln Park Lake	Allocation
Trash LA	0

5.4.6.1 Wasteload Allocations

Since there are no point sources in the Lincoln Park Lake watershed, wasteload allocations are not provided. If a point source is added to the watershed in the future, its wasteload allocation will be zero allowable trash.

5.4.6.2 Load Allocations

Nonpoint source areas refer to locations where trash may be carried by overland flow, wildlife, or wind to waterbodies. Due to the transportation mechanism by wind, wildlife, and overland flow to relocate trash from land to waterbodies, the nonpoint source area may be smaller than the watershed. In addition, trash loadings frequently occur immediately around or directly into the lake making the load allocation a significant source of trash. According to the study by the city of Calabasas, the trash generation rate is 640 gallons per square mile per year from nonpoint sources areas (including, but not limited to, schools, commercial areas, residential areas, public services, road, and open space and parks areas). Current trash rates were calculated in the nonpoint source section.

Load allocations (LAs) for nonpoint sources are zero trash. Zero is defined as no allowable trash found in and on the lake, and along the shoreline. According to the Porter-Cologne Act, load allocations may be addressed by the conditional Waivers of WDRs, or WDRs. Responsible jurisdictions should monitor the trash quantity deposited in the vicinities of the waterbodies of concern as well as on the waterbody to comply with the load allocation.

The area adjacent to Lincoln Park Lake or defined as nonpoint sources includes parking lots, recreational areas, picnic areas, walking trails, and an educational institution. Assuming that trash within a reasonable distance from Lincoln Park Lake has a high potential to reach the waterbody, the nonpoint source jurisdiction is the city of Los Angeles. All load allocations are set to zero allowable trash.

5.4.6.3 Margin of Safety

A margin of safety (MOS) accounts for uncertainties in the TMDL analysis. The MOS can be expressed as an explicit mass load, or included implicitly in the WLAs and LAs that are allocated. Because this

TMDL sets WLAs and LAs as zero trash, the TMDL includes an implicit MOS. Therefore, an explicit MOS is not necessary.

5.4.6.4 Critical Conditions/Seasonality

Critical conditions for Lincoln Park Lake are based on three conditions that correlate with loading conditions:

- Major storms
- Wind advisories issued by the National Weather Service
- High visitation – On weekends and holidays from May 15 to October 15.

Critical conditions do not affect wasteload or load allocations because zero trash is a conservative target. However, implementation efforts should be heightened during critical conditions in order to ensure that no trash enters the waterbody.

5.4.6.5 Future Growth

If any sources, currently assigned load allocations, are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality based effluent limitations pursuant to 40 CFR 122.44(d)(1).

5.5 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

If necessary, these TMDLs may be revised as the result of new information (See Section 5.6 Monitoring Recommendations).

5.5.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 5-6 and Table 5-9 for nutrients and trash, respectively.

5.5.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to supplemental water additions (Table 5-4). These mass-based waste load allocations will be implemented by the Regional Board.

5.5.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

Lincoln Park Lake has both nutrient-related and trash impairments. While there are some management strategies that would address both of these impairments (i.e., discouraging bird feeding), their differences warrant separate implementation and monitoring discussions.

5.5.3.1 Nutrient-Related Impairments

To address nutrient-related impairments, source reduction and pollutant removal BMPs designed to reduce sediment loading could be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to Lincoln Park Lake. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

Education of lake maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations. The NNE BATHTUB model indicated Additional Parkland Loading is present in Lincoln Park Lake. This lake is heavily frequented by bird feeders and the additional bird feces produced by bird feeding contributes to this load; loads linked to trash and associated food scraps would also be reduced.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

5.5.3.2 Trash Impairment

LA may be complied with through the implementation of nonstructural BMPs or any other lawful methods which meet the target of zero trash. USEPA recommends implementation plans be consistent with the Los Angeles River trash TMDL. A minimum frequency of trash collection and assessment should be established at an interval that prevents trash from accumulating in deleterious amounts in between collections. Trash should be prevented by providing effective public education about littering impacts. Signs dissuading littering and wildlife feeding along roadways and around the lake are recommended.

A city ban, tax, or incentive program reducing single-use plastic bags, Styrofoam containers, and other commonly discarded items which cannot decompose is recommended (Los Angeles County Department of Public Works, 2007).

Lincoln Park's grounds and facilities are maintained by the city of Los Angeles. Trash is currently collected and removed from the park daily. USEPA recommends continuation and expansion of the current trash pick-ups by the city of Los Angeles, including the collection of small trash items, such as cigarette butts.

The city of Los Angeles is also responsible for the trash in the lake. Currently trash is removed from the middle of the lake if a problem is reported. A more frequent in-lake trash removal program should be established to prevent the accumulation of small trash pieces in the waterbody.

The prevention and removal of trash in Lincoln Park Lake will lead to enhanced aesthetics, improved water quality, and the protection of habitat.

5.6 MONITORING RECOMMENDATIONS

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations result in compliance with the chlorophyll *a* and trash targets, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be: 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies and trash levels.

5.6.1 Nutrient-Related Impairments

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. At Lincoln Park Lake the only wasteload allocation is to supplemental water additions. This source should be monitoring once a year during the summer months (the critical condition) for at minimum; ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient TMDLs for Lincoln Park Lake conclude that a 56.0 percent reduction in total phosphorus loading and a 45.5 percent reduction in total nitrogen loading are needed to maintain a summer average chlorophyll *a* concentration of 20 µg/L. As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 5-3), target concentrations in supplemental water additions may be 0.0519 mg-P/L and 0.486 mg-N/L. Similarly, target concentrations associated may be 0.184 mg-P/L and 2.23 mg-N/L in the city of Los Angeles runoff , 0.0518 mg-P/L and 0.486 mg-N/L in the parkland irrigation return flows, and, assuming an average precipitation depth, the target concentration associated with precipitation may be 0.112 mg-N/L (note: the flows associated with the additional parkland loading are unknown, so target concentrations cannot be estimated). As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

5.6.2 Trash Impairments

Responsible jurisdictions should monitor the trash quantity deposited in the vicinity of Lincoln Park Lake as well as on the waterbody to comply with the load allocation and to understand the effectiveness of various implementation efforts. Quarterly monitoring using the Rapid Trash Assessment Method is recommended. The trash TMDL target is zero trash; a 100 percent reduction is required.

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6 Echo Park Lake TMDLs

Echo Park Lake (#CAL4051501020000228155002) is listed as impaired by algae, ammonia, copper, eutrophication, lead, odor, PCBs, pH, and trash (SWRCB, 2010). In addition, chlordane and dieldrin impairments have been identified by new data analyses since the 2008-2010 303(d) list data cut off. This section of the TMDL report describes the impairments, and the TMDLs developed to address them: nutrients (see Section 6.2), organochlorine (OC) pesticides and PCBs (Section 6.5 through Section 6.7), and trash (Section 6.8). Nutrient TMDLs are identified here based on existing conditions since nitrogen and phosphorus levels are achieving the chlorophyll *a* target level. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that copper and lead are currently achieving numeric targets at Echo Park Lake; therefore, TMDLs are not included for these pollutants. Analyses are presented below for lead (Section 6.3) and copper (Section 6.4).

6.1 Environmental Setting

Echo Park Lake is located in the Los Angeles River (HUC 18070105) (Figure 6-1). The waterbody was originally constructed as the Arroyo de los Reyes reservoir in 1898 and became Echo Park Lake in 1907. The lake now has a surface area of 14.1 acres (based on Southern California Association of Governments [SCAG] 2005 land use), an average depth of five feet (estimated from 2009 sampling events and the Urban Lakes Study [UC Riverside, 1994]), and a volume of 70.5 ac-ft (calculated from the land use estimated surface area and estimated average depth). Two primary storm drains provide inflows to the lake; the lake then discharges to a storm drain that ultimately reaches the Los Angeles River.

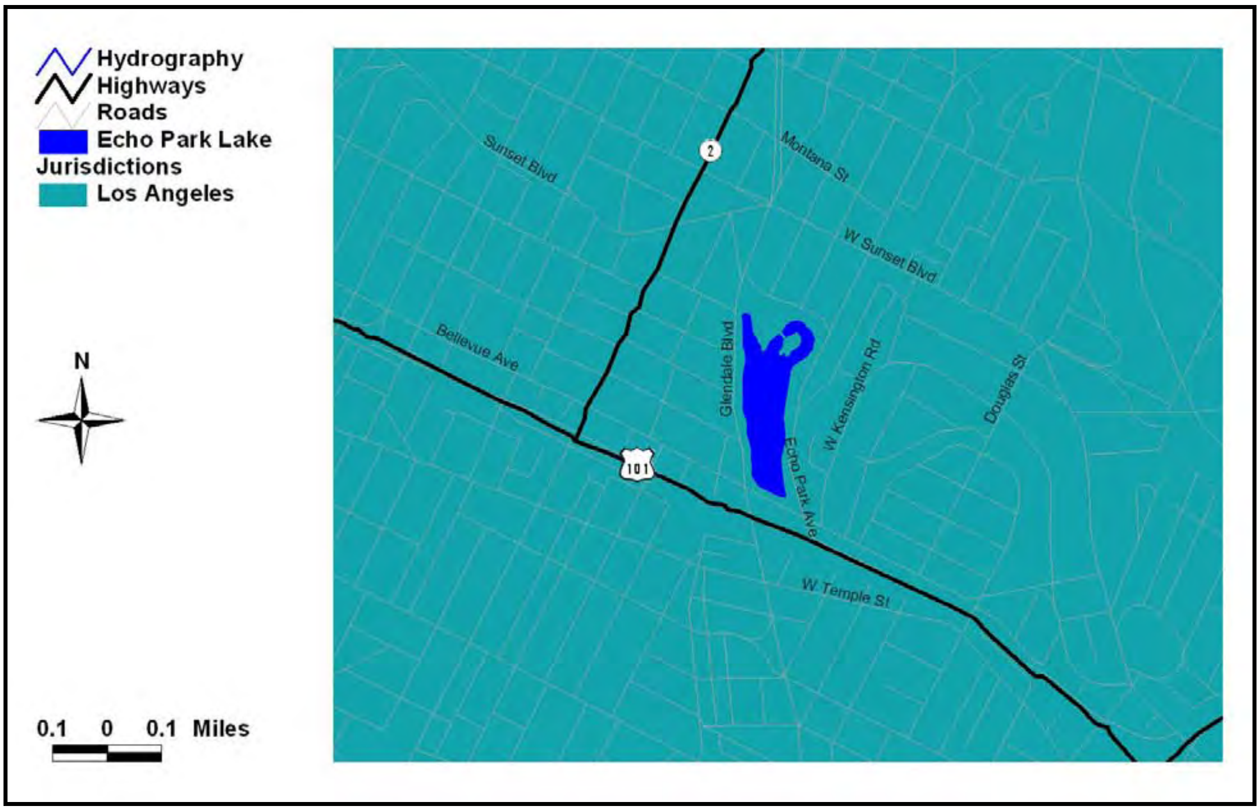

Figure 6-1. Location of Echo Park Lake

Mixing and aeration of the lake is currently performed by a mechanical aeration system, including the lake's notable fountain located near the tip of the western peninsula. Objectives of aeration include increasing dissolved oxygen and decreasing nuisance surface scum and algal growth. In addition to aeration, four floating hydroponic wetlands were constructed for additional water quality treatment. An island, managed by the city of Los Angeles, located in the northeastern lobe of the lake, also provides habitat for waterfowl and turtles. Figure 6-2 shows the fountain and one of the hydroponic islands in the lake; Figure 6-3 shows the bubbles that result from one of the aerators.

Figure 6-2. Fountain and Hydroponic Island at Echo Park Lake

Figure 6-3. An Aerator North of the Bridge at Echo Park Lake

Echo Park Lake harbors a historically and culturally significant population of lotus beds; it is believed that the current population is a descendent of lotus plants imported in 1920. Once believed to be the largest population in the western United States, recent decline of the lotus beds has been attributed to buildup of hydrogen-sulfide in the sediment. Due to the stress associated with the hydrogen-sulfide, it is not expected that the existing-historic lotus beds will reestablish. For this reason, a lotus restoration plan, completed in 2009, will be vital to the future sustainability of the lotus beds (Black & Veatch, 2009). A critical feature to reduce the concentration of hydrogen sulfide and augment the success of the lotus beds is proper lake circulation and improved aeration.

A small strip of parkland surrounds the lake, offering a slight buffer from the surrounding roads and dense residential development. The park provides public access to the lake and restrooms located in the park are connected to the city sewer system. According to California Department of Fish and Game, trout are periodically stocked (CDFG, 2009). Catch and release fishing and paddle boating are the primary recreational uses (Figure 6-4). Bird feeding is another recreational activity at Echo Park Lake and heavy feeding has been observed during recent fieldwork, likely contributing to larger resident bird populations. Visitors are not allowed to swim in the lake. Lake managers use algaecides to control algal growth in the lake on an as-needed basis.

Note: recreational uses include catch and release fishing and paddle boating.

Figure 6-4. Echo Park Lake Recreational Uses

Additional characteristics of the watershed are summarized below.

6.1.1 Elevation, Storm Drain Networks, and Subwatershed Boundaries

The Echo Park Lake watershed is 784 acres in size and ranges in elevation from 115 meters to 229 meters (Figure 6-5). The TMDL subwatershed boundaries selected for Echo Park Lake were based on boundaries obtained from the county of Los Angeles and are labeled on the figure accordingly. The county of Los Angeles southern-subwatershed was sub-delineated based on a digital elevation model to remove the drainage area downstream of the lake. The subwatershed draining the northern part of the watershed is 614 acres, and the southern subwatershed drains 170 acres. The majority of wet weather and dry weather flows from the northwestern and northeastern storm drains are diverted around the lake (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading). Because both subwatersheds drain to a storm drain system and because many storm drains drain to the lake, all allocations except atmospheric deposition will be wasteload allocations. The trash TMDL includes load allocations due to direct dumping of trash along the shoreline and in the water by park visitors in the area indicated in Figure 6-6.

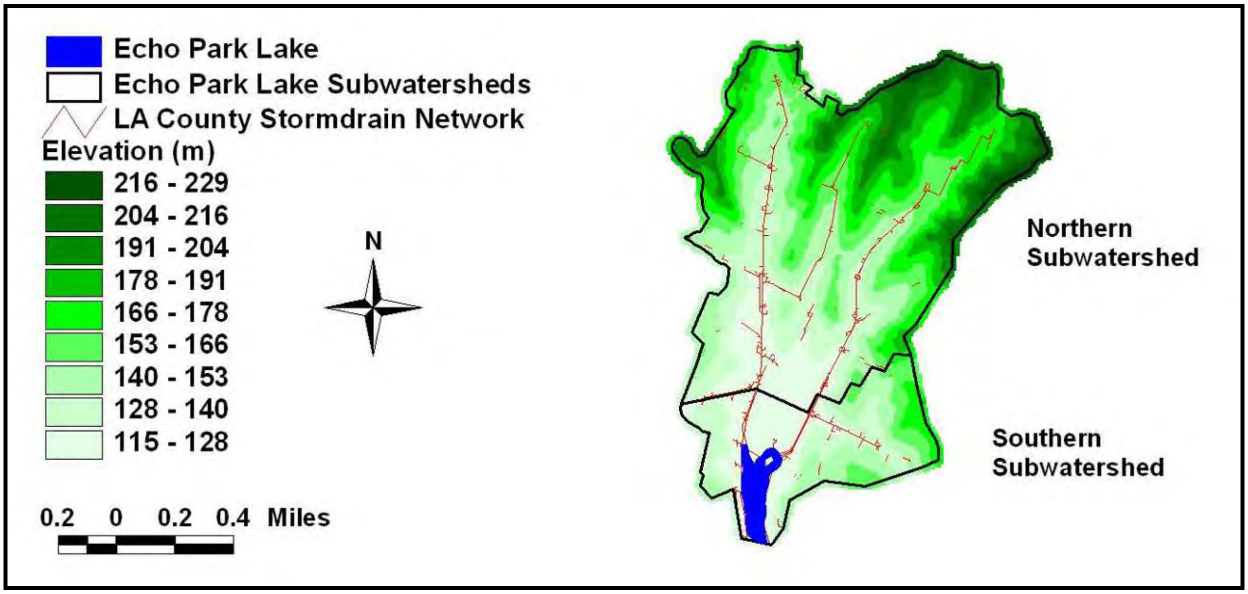

Figure 6-5. Elevation, Storm Drain Network, and TMDL Subwatershed Boundaries for Echo Park Lake

6.1.2 MS4 Permittees

Figure 6-6 shows the MS4 stormwater permittees in the Echo Park Lake watershed. Both subwatersheds are located entirely within the city of Los Angeles with a small portion in Caltrans area. Figure 6-7 shows one of the main storm drain inlets at the lake. The park is comprised of 15.5 acres of land adjacent to the lake.

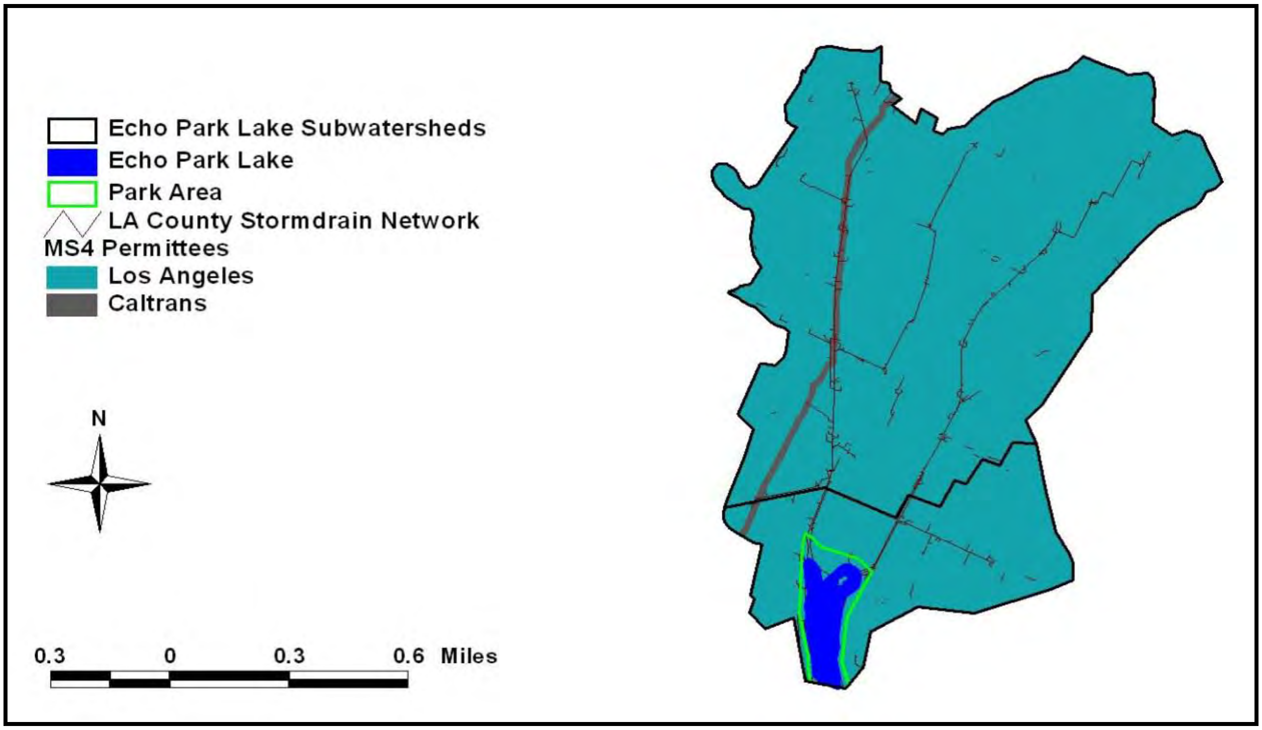

Figure 6-6. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Echo Park Lake Subwatersheds

Figure 6-7. Echo Park Lake Northeast Storm Drain Input

6.1.3 Non-MS4 NPDES Dischargers

The primary permitted discharger in the watershed is the county of Los Angeles MS4 system. There is one additional NPDES permitted discharger (non-MS4) in the Echo Park Lake watershed (Table 6-1 and Figure 6-8) that is a discharger covered under a general industrial stormwater permit (see Section 3.1 for a detailed discussion of this permit type). This permit was identified by querying excel files of permits from the Regional Board website (Excel files for each watershed are available from this link, www.waterboards.ca.gov/losangeles/water_issues/programs/regional_program/index.shtml#watershed, accessed on October 5, 2009). This permittee is located in the city of Los Angeles in the northern subwatershed (Section 6.1.1) and has two disturbed acres. The disturbed area associated with this permit drains to the northwestern storm drain which is diverted around the lake in most cases except during high flow events. Loads from this permittee were therefore not calculated; however, concentration-based wasteload allocations for this permittee are included in the TMDLs.

Table 6-1. Non-MS4 Permits in the Echo Park Lake Subwatersheds

Type of NPDES Permit	Number of Permits	Subwatershed	Jurisdiction	Disturbed Area
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000003)	1	Northern	City of Los Angeles	2 acres

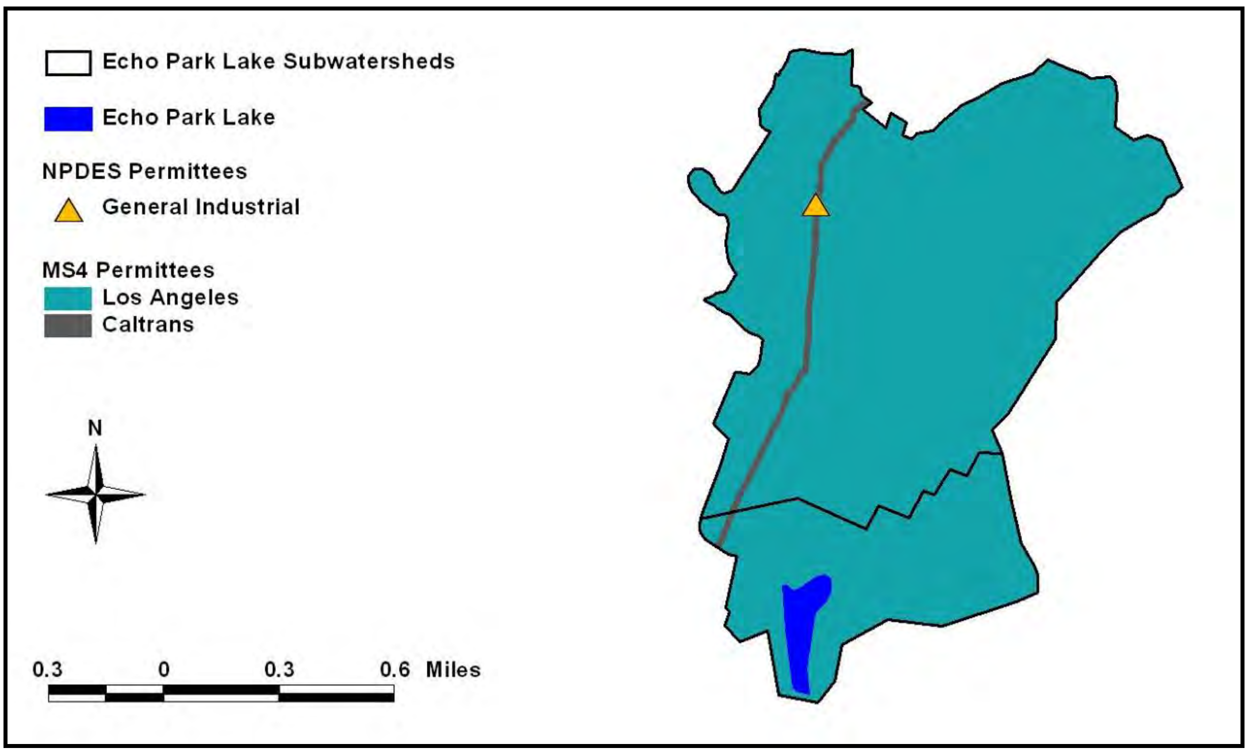

Figure 6-8. Non-MS4 Permits in the Echo Park Lake Subwatersheds

6.1.4 Land Uses and Soil Types

The analysis for this watershed includes source loading estimates obtained from the Los Angeles River Basin LSPC Model discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the Los Angeles River Basin LSPC model are shown in Figure 6-9. The watershed is comprised primarily of residential development as well as commercial, other urban, industrial, and open space areas. Table 6-2 and Table 6-3 summarize the land use areas by TMDL subwatershed and jurisdiction.

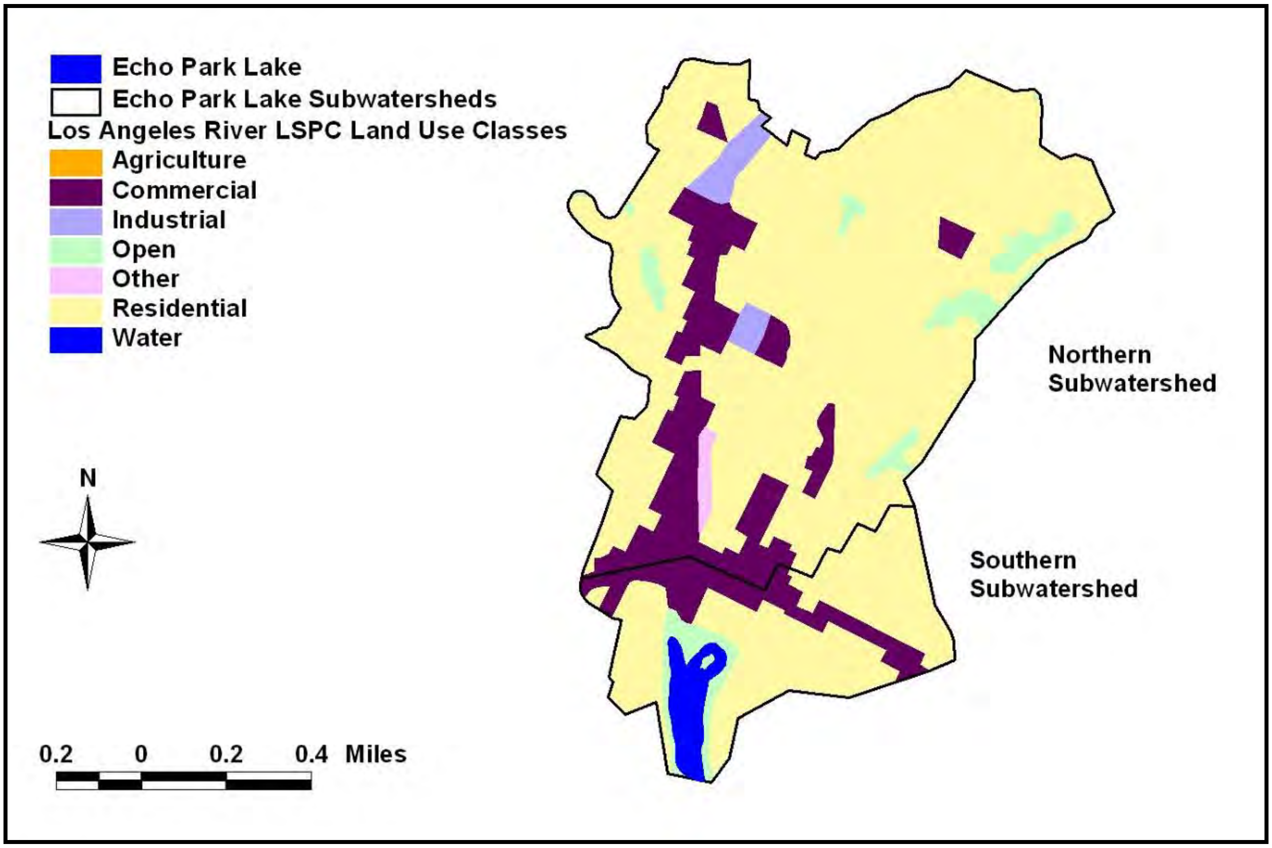

Figure 6-9. LSPC Land Use Classes for the Echo Park Lake Subwatersheds

Table 6-2. Land Use Areas (ac) Draining to Echo Park Lake from the Northern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	78.4	0	**78.4**
Industrial	12.2	13.0	**25.2**
Open	27.5	0	**27.5**
Other Urban	4.67	0	**4.67**
Residential	479	0	**479**
Total	**601**	**13.0**	**614**

Table 6-3. Land Use Areas (ac) Draining to Echo Park Lake from the Southern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	31.6	0	**31.6**
Industrial	0	1.10	**1.10**
Open	15.5	0	**15.5**
Other Urban	0	0	**0**
Residential	122	0	**122**
Total	**169**	**1.10**	**170**

There are no Resource Conservation and Recovery Act (RCRA) contaminated industrial facilities located near the Echo Park Lake watershed. The USDA STATSGO state soils coverage identifies all soils within the Echo Park Lake watershed as Urban Land – Lithic Xerorthents – Hambright – Castaic (MUKEY 660489). These soils are classified as belonging to soil hydrologic group D, which is characterized by high runoff potential, very low infiltration rates, and generally high clay content.

6.1.5 Additional Inputs

In addition to stormwater runoff, a natural spring exists in the center of Echo Park Lake (UC Riverside, 1994); however, the addition of potable water is required to maintain the lake level. A potable water source at Echo Park Lake is used for both supplemental water additions to the lake and irrigation of surrounding parklands (Figure 6-10). According to a hydrologic study of the park lake conducted by Black & Veatch (2008), 162 ac-ft/yr of potable water is pumped annually for these purposes. Staff at Echo Park indicate that a portion of the pumped water is used to irrigate approximately 9 acres in the vicinity of the lake at a rate of approximately 1 foot per year. Some of this irrigation water may reach the lake (4.6 percent of the total irrigation volume is assumed to reach the lake).

Figure 6-10. Echo Park Lake Potable Water Source and Northwestern Storm Drain Input

6.2 NUTRIENT RELATED IMPAIRMENTS

A number of the assessed impairments for Echo Park Lake are associated with nutrients and eutrophication. Nutrient-related impairments for Echo Park Lake include algae, ammonia, eutrophication, odor, and pH (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algal photosynthesis removes carbon dioxide from the water, which can lead to elevated pH in poorly buffered systems. Algal blooms may also contribute to odor problems.

6.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Echo Park Lake include REC1, REC2, WARM, WILD, and MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, and WARM uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels, and alter biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD and MUN uses could become impaired.

6.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Echo Park Lake. The following targets apply to the algae, ammonia, eutrophication, odor, and pH impairments (see Section 2 for additional details and Table 6-4 for a summary):

- The Basin Plan expresses ammonia targets as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life. In order to assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for Echo Park Lake in these TMDLs, a median temperature of 19.7 ºC and a 95th percentile pH of 9.1 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 1.14 mg-N/L, the four-day average is 0.76 mg-N/L, and the 30-day average (chronic) target is 0.30 mg-N/L (Note: The median temperature and 95th percentile pH values were calculated from the observed surface depth data and used in the calculation of ammonia targets. These are presented as example calculations since the actual target varies with the temperature and pH values determined during sample collection).
- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient (e.g., nitrogen and phosphorous) concentrations in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Echo Park Lake; however, as described in Tetra Tech (2006), summer (May to September) mean and annual mean chlorophyll *a* concentration of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and is also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target must be met at half of the Secchi depth during the summer (May – September) and annual averaging periods.

- The Basin Plan states that "waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses."
- The Basin Plan states "at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges." Shallow, well-mixed lakes, such as Echo Park Lake, must meet the DO target in the water column from the surface to 0.3 meters above the bottom of the lake.
- The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." Shallow, well-mixed lakes, such as Echo Park Lake, must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake.

Nitrogen and phosphorus target concentrations within the lake are based on existing conditions as explained in Sections 6.2.5 and 6.2.6:

- 1.2 mg-N/L summer average (May – September) and annual average
- 0.12 mg-P/L summer average (May – September) and annual average

Table 6-4. Nutrient-Related Numeric Targets for Echo Park Lake

Parameter	Numeric Target	Notes
Ammonia[1]	1.14 mg-N/L acute (one-hour) 0.76 mg-N/L four-day average 0.30 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	1.2 mg-N/L summer average (May – September) and annual average	Conservatively based on existing conditions, which are maintaining chlorophyll *a* levels below the target of 20 µg/L

Parameter	Numeric Target	Notes
Total Phosphorous	0.12 mg-P/L summer average (May – September) and annual average	Conservatively based on existing conditions, which are maintaining chlorophyll *a* levels below the target of 20 µg/L

[1] The median temperature and 95th percentile pH values were calculated from the observed surface depth data and used in the calculation of ammonia targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan..

6.2.3 Summary of Monitoring Data

Water quality monitoring has occurred in Echo Park Lake in 1992, 1993, and 2003 through 2009. This section summarizes the monitoring data relevant to the nutrient impairments. Additional details regarding monitoring are discussed in Appendix G (Monitoring Data).

During the 1992/1993 Urban Lakes Study, sampling occurred near the center of the lower half of the lake (UC Riverside, 1994). Total Kjeldahl nitrogen (TKN) concentrations during this sampling period ranged from 0.9 mg-N/L to 1.9 mg-N/L. Ammonium concentrations were less than the reporting limit for 22 of 31 samples, and the maximum observed ammonium concentration was 0.7 mg-N/L which is less than the acute target assuming the analysis methodology converted all ammonia to ammonium. Nitrite concentrations were less than the detection limit (0.1 mg-N/L) in all samples and 24 of 31 nitrate samples were less than the detection limit (0.1 mg-N/L). The maximum observed nitrate concentration was 0.2 mg-N/L. Orthophosphate concentrations were generally less than or equivalent to the detection limit (0.1 mg-P/L) with some observations of 0.2 mg-P/L. Total phosphorus concentrations ranged from less than the detection limit (0.1 mg-P/L) to 0.3 mg-P/L. pH measurements ranged from 7.7 to 9.4 throughout the water column, and TOC ranged from 4.8 mg/L to 7.6 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 6 µg/L to 66 µg/L with an average of 24 µg/L. For this period, exceedances of the pH and chlorophyll *a* targets were observed. The report stated that aquatic weeds were present near the fountain, lotus plants were located at the northwest end of the lake, and algal blooms were observed during the summer. A strong odor resulting from duck feces was also reported. Nutrient levels were generally low during the study period and it was reported that the level of algae in the lake was not problematic.

There were no stations in Echo Park Lake or its drainage area in the Regional Board Water Quality Assessment Database. The Water Quality Assessment Report, however, states that pH was not supporting the contact recreation use and partially supporting the aquatic life use: 69 measurements of pH were collected which ranged from 7.0 to 9.4. Thirty-one ammonium samples were collected with values ranging from non-detect to 0.71 mg-N/L; ammonia was listed as not supporting the aquatic life and contact recreation uses. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Odor and algae were both listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

In 2003, the City of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples from Echo Park Lake at three in-lake stations. Of the 84 samples collected during this period, 38 were non-detect for ammonia (less than 0.1 mg-N/L); the maximum ammonia concentration was 0.93 mg-N/L which does not exceed the acute or chronic ammonia criteria based on the associated pH and temperature measurements. Organic nitrogen concentrations ranged from 0.28 mg-N/L to 3.14 mg-N/L. Thirty-five nitrate samples were below the detection limit (0.02 mg-N/L), and the maximum observed nitrate concentration was 1.0 mg-N/L. Fifty-five of the nitrite samples were below the detection limit (0.02 mg-N/L); the other two samples had concentrations of 0.02 mg-N/L and 0.09 mg-N/L. Total nitrogen concentrations, calculated from the sum of ammonia, organic nitrogen, nitrate, and nitrite, ranged from 0.28 mg-N/L to 3.48 mg-N/L. Total phosphate measurements generally ranged from 0.06 mg-P/L to 0.51 mg-P/L with three measurements less than detection (0.05 mg-P/L). No chlorophyll *a* data were reported.

Vertical profile data using datasondes were also collected by the City of Los Angeles Bureau of Sanitation during 2003. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification. Dissolved oxygen concentrations ranged from 5.62 mg/L to 15.9 mg/L; pH ranged from 7.46 to 9.04 throughout the water column. Twenty-seven percent of pH measurements exceeded the maximum allowable value.

In 2008, the Regional Board sampled Echo Park Lake on two occasions. As the lake is relatively shallow and well mixed by wind action and aerators, the sampling team collected analytical samples from the lake surface only. On June 25, 2008, ammonia concentrations in Echo Park Lake were fairly similar at all three sampled locations and ranged from 0.131 mg-N/L to 0.136 mg-N/L. TKN at the lake midpoint and near the hydroponic island ranged from 1.38 mg-N/L to 1.49 mg-N/L; the concentration was higher in the lotus beds at 4.72 mg-N/L. Concentrations of nitrate, nitrite, orthophosphate, and total phosphate were all less than the reporting limits of 0.1 mg-N/L, 0.1 mg-N/L, 0.4 mg-P/L, and 0.5 mg-P/L, respectively. Dissolved oxygen concentrations ranged from 4.95 mg/L to 9.82 mg/L, and pH ranged from 8.21 to 8.56. The pH levels showed slight exceedances relative to the target. The DO target for waters designated WARM is 5 mg/L and after rounding to the appropriate decimal place the lowest observed measurement of 4.95 mg/L meets the target. Note that the pH meter was not producing calibration results within the acceptable range and that exceedances of the pH target were only observed along the shoreline near two storm drain outlets. Chlorophyll *a* samples generally ranged from 10.9 μg/L to 26.7 μg/L. There were two outlier chlorophyll *a* concentrations of 0.8 μg/L and 53.6 μg/L. The average concentration in the lake on this sampling day, including the outliers, was 17.3 μg/L. A description of the methodology or equipment used to measure chlorophyll *a* concentrations in the field was not provided.

Regional Board also collected samples on December 18, 2008 from five shoreline locations at a depth of approximately 4 inches. pH ranged from 7.7 to 8.1. No exceedances of the acute ammonia target or chlorophyll *a* target were observed on this day. These samples are not discussed in detail in this section as shoreline samples may not be reflective of conditions in the lake as a whole.

On March 10, 2009, USEPA and the Regional Board sampled Echo Park Lake at three locations. Ammonia concentrations ranged from 0.04 mg-N/L to 0.06 mg-N/L, and TKN ranged from 0.7 mg-N/L to 1.3 mg-N/L. Nitrate was approximately 0.15 mg-N/L at each station, and nitrite was less than the detection limit (0.01 mg-N/L). Orthophosphate was less than the detection limit (0.008 mg-P/L) at each station, and total phosphorus generally ranged from 0.033 mg-P/L to 0.071 mg-P/L. One total phosphorus sample measured 0.762 mg-P/L, though the field duplicate had a value of 0.071 mg-P/L. Chlorophyll *a* measurements in the lake ranged from 14.2 μg/L to 15.2 μg/L.

Two in-lake stations were sampled by USEPA and the Regional Board on August 4th, 2009. All nitrogen parameters (ammonia, TKN, nitrate, and nitrite) were below detection limits (0.03 mg-N/L, 0.456 mg-N/L, 0.01 mg-N/L, 0.01 mg-N/L, respectively) at both sites. Total phosphorus measurements were 0.196 mg-P/L and 0.195 mg-P/L. The orthophosphate concentrations were 0.0850 mg-P/L and 0.0917 mg-P/L. The chlorophyll *a* measurements were 15.0 µg/L and 15.5 µg/L.

Profile data were collected in Echo Park Lake during both USEPA/Regional Board sampling events. On both days the lake appeared well-mixed both vertically and spatially. On March 10th, DO concentrations in the lake generally ranged from 7.0 mg/L to 8.6 mg/L with one reading of 10.0 mg/L from a surface sample; pH ranged from 7.5 to 7.9. On August 4th, DO concentrations in the lake ranged from 6.4 mg/L to 7.6 mg/L. The pH ranged from 8.3 to 8.6 throughout the water column and therefore exceeded the allowable range during the August 4th sampling event. Potable water measured during the August 4th sampling event was 7.54 pH units.

In summary, recent samples show the chlorophyll *a* target is being met. The 1994 Urban Lakes Study suggested that the fountain and aeration system were effective in managing DO concentrations (UC Riverside, 1994). That appears to be the case today as well, as the DO measurements are above 5 mg/L and averaged greater than the target of 7 mg/L. No odors were observed during five recent sampling events by USEPA and/or Regional Board. It is unlikely that the source of the odor reported at Echo Road Park Lake is due to elevated nutrient and algal biomass levels. They are likely associated with the trash impairment addressed in Section 6.8.

6.2.3.1 Summary of pH Non-Impairment

The Basin Plan states *"The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge."* There were nine elevations of pH in 36 recent samples. All elevations occurred during dry weather and therefore are not due to stormwater flow. Potable water which accounts for 89 percent of influent water measured 7.54 pH units. There are no other waste discharges that could be elevating the pH. Therefore, the elevated pH levels are meeting the water quality objective. In addition, the chlorophyll *a* target is being met, so nutrient loading is not elevating pH. Based on these multiple lines of evidence, Echo Park Lake is attaining beneficial uses and meets pH water quality standards. USEPA concludes that preparing a TMDL for pH is unwarranted at this time. USEPA recommends that Echo Park Lake not be identified as impaired by pH in California's next 303(d) list.

6.2.3.2 Summary of Ammonia Non-Impairment

Echo Park Lake was listed as impaired for ammonia in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 35 additional samples (7 wet weather) between May 2003 and February 2010 to evaluate current water quality conditions. There was one ammonia exceedance in 35 samples (Appendix G, Monitoring Data). Therefore, Echo Park Lake meets ammonia water quality standards and USEPA concludes that preparing a TMDL for ammonia is unwarranted at this time. USEPA recommends that Echo Park Lake not be identified as impaired for ammonia in California's next 303(d) listing.

6.2.4 Source Assessment

The source assessment for Echo Park Lake includes load estimates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading) including irrigation (4.6 percent of the total irrigation volume is assumed to reach the lake), potable water used supplementing lake levels (Appendix F, Dry Weather Loading), and atmospheric deposition (Appendix E, Atmospheric Deposition). Loads generated from upland areas located in the city of Los Angeles in the northern and southern watersheds contribute 29 percent of the total phosphorus load and 28 percent of the total nitrogen load (the majority of runoff from these areas is diverted downstream of the lake). The potable water used for supplemental water additions contributes 46 percent of the total phosphorus load and 64 percent of the total nitrogen load to Echo Park Lake. In addition to these sources, there are other sources of loading to Echo Park Lake for which loading estimates were not available (Appendix F, Dry Weather Loading). These may include excessive fertilization relative to product recommendations, internal loading from lake sediments, natural wildlife populations, excessive resident bird populations caused by the improper disposal of food waste, and pet wastes. During calibration of the NNE BATHTUB model, loads in the category, "Additional Parkland Loading," were increased until simulated concentrations of total phosphorus and total nitrogen matched those observed (see Section 6.2.5). For this waterbody, these additional sources of loading comprise 24 percent of the total phosphorus load and 5.5 percent of the total nitrogen load. All existing loads to Echo Park Lake are summarized in Table 6-5.

Table 6-5. Summary of Average Annual Flows and Nutrient Loading to Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Northern	Caltrans	State Highway Stormwater[1]	0.385	0.608 (0.6)	4.77 (0.7)
Northern	City of Los Angeles	MS4 Stormwater[1]	13.2	24.7 (22.7)	156 (21.3)
Southern	Caltrans	State Highway Stormwater[1]	0.033	0.051 (0.05)	0.403 (0.06)
Southern	City of Los Angeles	MS4 Stormwater[1]	4.16	6.99 (6.4)	48.4 (6.6)
Southern	City of Los Angeles	Supplemental Water Additions (Potable Water)	153	50.8 (46.6)	471 (64.4)
Southern	City of Los Angeles	Parkland Irrigation	0.418	0.139 (0.1)	1.29 (0.2)
Southern	City of Los Angeles	Additional Parkland Loading	NA	26.1 (23.9)	40 (5.4)
Lake Surface		Atmospheric Deposition[2]	18.0	NA	9.0 (1.2)
Total			**188**	**109**	**731**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

A significant portion of loading from the additional parkland sources is likely due to excessive resident bird populations. According to a recent water quality modeling study conducted by Black and Veatch (2010), there is a year-round, resident bird population of approximately 1,000 Rock Doves and American Coots. Estimates of nutrient loading from these birds were based on literature values and an assumption that all waste generated by the birds would reach the lake (i.e., no uptake or trapping on adjacent areas). The estimated total phosphorus loading from these birds is 78 lb-P/yr, and the estimated total nitrogen loading is 780 lb-N/yr. Both loading estimates are greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the year-round bird population at Echo Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading.

6.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on Echo Park Lake, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Echo Park Lake, the following inputs apply: surface area of 14.1 acres, average depth of 5 ft, and volume of 70.5 ac-ft. Based on the turnover ratios for both nitrogen and phosphorus (Walker, 1987), the annual averaging period is most appropriate (i.e., annual loads are input to the model rather than summer season loads). Based on the results of a recent exfiltration and flow monitoring study of the lake (Black and Veatch, 2008), exfiltration losses through the lake liner are approximately 52.6 ac-ft/yr. Loads of nitrogen and phosphorus associated with these losses were estimated from average in-lake water quality data multiplied by the annual rate of exfiltration.

The NNE BATHTUB Tool was calibrated to average summer season water quality data observed over twice the Secchi depth (2*0.8 m = 1.6 m). Because simulated phosphorus concentrations could not be calibrated within the default range specified in the BATHTUB User's Manual (Walker, 1987), loads from additional parkland sources were increased to predict the average summer concentrations of total phosphorus (0.115 mg-P/L) and total nitrogen (1.16 mg-N/L), leaving the net sedimentation rates at 1.0 for both nutrients. Additional loading associated with parkland areas is 40 lb-N/yr and 40 lb-P/yr. The amount of the additional parkland loading of phosphorus due to internal recycling was calculated with the method discussed in Appendix A (Nutrient TMDL Development) and is 13.9 lb-P/yr. This portion of the phosphorus load was subtracted out of the additional parkland sources category, and the model was recalibrated with a loading of 26.1 lb-P/yr. The resulting calibration factor on the net phosphorus settling rate is 0.74, which allows the model to account for internal loading implicitly. Though internal loading is not explicitly assigned a load allocation, reductions in external loading of phosphorus will ultimately result in reductions of internal cycling processes. Internal loading of nitrogen was not calculated because 1) internal loading is typically insignificant relative to external loading, and 2) empirical relationships for the estimation of internal nitrogen loading have not been developed. Thus, the additional parkland source loading and calibration factor for nitrogen were not changed. To simulate the average observed summer chlorophyll *a* concentration, the calibration factor on chlorophyll *a* concentration was set to 0.45 for a predicted concentration of 17.8 µg/L.

Because of the way Echo Park Lake is currently managed (fountain, aeration system, hydroponic islands, etc.), the density of algae is typically below the target summer average concentration (20 µg/L). However pH and chlorophyll *a* exceedances have occurred. To be adequately protective, nutrient TMDLs are allocated at existing levels as an antidegradation measure to ensure that future loading does not increase the chlorophyll *a* concentration.

6.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Based on observed levels of chlorophyll *a* and DO in Echo Park Lake, existing levels of nitrogen and phosphorus loading result in attainment of both the chlorophyll *a* and DO targets. Monitoring data indicate that the average in-lake total nitrogen concentration is 1.16 mg-N/L (Appendix G, Monitoring Data). Because the majority of in-lake phosphorous samples have been less than the detection limits for the analytical laboratory, the phosphorus target concentration is based on an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10. This ratio was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 1.2 mg-N/L summer average (May – September) and annual average
- 0.12 mg-P/L summer average (May – September) and annual average

To prevent degradation of this waterbody, nutrient TMDLs will be allocated based on existing loading. These TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation. Note that the MOS is zero because the TMDLs are equal to the existing load.

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load is equal to the existing load and is divided among WLAs and LAs. The resulting TMDL equation for total nitrogen is then:

731 lb-N/yr = 682 lb-N/yr + 49.0 lb-N/yr + 0 lb-N/yr

For total phosphorus, the allocatable load is equal to the existing load and allocated to WLAs only; LAs are zero as explained in Section 6.2.6.2. The resulting TMDL equation for total phosphorous is then:

109 lb-P/yr = 83.3 lb-P/yr + 26.1 lb-P/yr + 0 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined for the lake based on recent and historical monitoring data (see Section 6.2.3). These in-lake concentrations reflect internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

6.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs and alternative WLAs for total phosphorous and total nitrogen. The alternative WLAs will be effective and supersede the WLAs listed in Table 6-6 if the conditions described in Section **Error! Reference source not found.** are met.

Under any of the wasteload allocation schemes responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention and treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. In the case of Echo Park Lake, the City of Los Angeles has already modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from such best management practices and construction is currently underway on a major lake rehabilitation project.

Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

The Echo Park Lake watershed drains to a series of storm drains prior to discharging to the lake. Therefore, all nutrient loads associated with the surrounding drainage area are assigned WLAs (Note: the loading associated with irrigation is included in the City of Los Angeles' WLA). The potable water input used for supplemental water addition to the lake discharges at a single point and is also assigned a WLA. Relevant permit numbers are

- County of Los Angeles (including the city of Los Angeles): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003

Note that WLAs are equal to existing loading rates because no reductions in loading are required. WLAs are presented in Table 6-6. These loading values (in pounds per year) represent the TMDLs wasteload allocations (Table 6-6). All responsible jurisdictions must meet the WLAs at the point of discharge as a mass load except for stormwater permittees under the general industrial stormwater permit that are receiving concentration-based WLAs. In Table 6-6 below, stormwater permittees under the general industrial stormwater permit must meet the concentration values to achieve compliance with the WLAs. The phosphorous and nitrogen WLA concentrations were calculated by dividing the allowable load (in lbs/yr; Table 6-6) by total inflow volume (**Error! Reference source not found.**). Each wasteload allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 6-6. Wasteload Allocations of Phosphorus and Nitrogen Loading to Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation Total Phosphorus (lb-P/yr)[4]	Wasteload Allocation Total Nitrogen (lb-N/yr)[4]
Northern	Caltrans	State Highway	0.608	4.77

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation Total Phosphorus (lb-P/yr)[4]	Wasteload Allocation Total Nitrogen (lb-N/yr)[4]
		Stormwater[1]		
Northern	City of Los Angeles	MS4 Stormwater[1]	24.7	156
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[3]	General Industrial Stormwater[1]	0.16 mg/L P[2]	1.33 mg/L N[2]
Southern	Caltrans	State Highway Stormwater[1]	0.051	0.403
Southern	City of Los Angeles	MS4 Stormwater[1]	7.129	49.69
Southern	City of Los Angeles	Supplemental Water Additions	50.8	471
Total			**83.3**	**682**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] For these responsible jurisdictions, the concentration-based WLA will be use to evaluate compliance.

[4] Each wasteload allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained. In assessing compliance with wasteload allocations, responsible jurisdictions assigned both northern and southern subwatershed allocations may combine allocations.

6.2.6.2 Load Allocations

Atmospheric deposition of nitrogen to the lake surface is a nonpoint source and is assigned a load allocation (LA). Table 6-7 presents the LA for atmospheric deposition, which is equivalent to existing loading rates because no reductions in loading are required. Atmospheric deposition does not contribute significant loads of phosphorus (Appendix E, Atmospheric Deposition). LAs are provided for each responsible jurisdiction and input. These loading values (in pounds per year) represent the TMDLs load allocations (Table 6-7). Each load allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 6-7. Load Allocations of Phosphorus and Nitrogen Loading to Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation Total Phosphorus (lb-P/yr)[1]	Load Allocation Total Nitrogen (lb-N/yr)[1]
Southern	City of Los Angeles	Additional Parkland Loading	26.1	40
Lake Surface		Atmospheric Deposition[2]	NA	9.0
Total			**26.1**	**49.0**

[1] Each load allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained. In assessing compliance with wasteload allocations, responsible jurisdictions assigned both northern and southern subwatershed allocations may have their allocations combined.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

6.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This lake is currently achieving the in-lake chlorophyll *a* target and TMDLs are being established at the existing loads. This conservative anti-degradation measure is the implicit margin of safety for these TMDLs.

6.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are based on existing conditions as an anti-degradation measure since nitrogen and phosphorus levels are currently achieving the chlorophyll *a* target level. These TMDLs therefore protect for critical conditions.

6.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). The majority of nutrient loading to Echo Park Lake comes from the supplemental water additions. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

The maximum daily loads from the supplemental water additions were calculated from average daily water volume and the long-term average concentration consistent with meeting the TMDLs. For the supplemental water addition, the allowable concentrations are 1.13 mg-N/L and 0.122 mg-P/L (Section 6.2.6.1). The daily average flow rate is 0.419 ac-ft/d (153 ac-ft/yr divided by 365 d/yr). The maximum daily nutrient loads from this source are 1.29 lb-N/d and 0.139 lb-P/d.

As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

6.2.6.6 Future Growth/Conditions

The Echo Park Lake watershed is nearly fully developed, with the exception of small park areas that are not likely to be converted in the near future. If land use changes do occur in the watershed, BMPs will be required such that loading rates are consistent with the allocations established by these TMDLs. Therefore, no load allocation has been set aside for future growth.

Though future growth is not expected to impact conditions in Echo Park Lake, the city of Los Angeles is in the process of designing and constructing a large scale rehabilitation project at the park, which will impact the conditions of the lake system. In addition to treating runoff flows with a hydrodynamic separator and constructed wetland system, the City is considering the use of reclaimed/recycled water for supplemental water additions to the lake rather than the potable water source that is currently used.

The design engineers indicate that the rehabilitation project will have the following impacts on the system (personal communication, James Rasmus, Black and Veatch, April 16, 2010):

- Wet weather flows to the lake from the storm drain system will increase from 16.7 ac-ft/yr to 131 ac-ft/yr. Dry weather flows to the lake from the storm drain system will increase from 0 ac-ft/yr to 123 ac-ft/yr.
- Exfiltration losses through the lake liner will decrease to 0.896 ac-ft/yr.
- The vortex and constructed wetland treatment system will treat 121 ac-ft/yr of wet weather flows, 123 ac-ft/yr of dry weather flows, and all water used for supplementing lake levels. Lake water will be recirculated through the constructed wetland system at a rate of 600 gpm.
- The vortex/constructed wetland system will remove 68 percent of the total nitrogen and 77 percent of the total phosphorus loads from treated flows. Recirculation of lake water will increase reduction efficiencies to 80 percent for total nitrogen and 86 percent for total phosphorus. These values represent updated efficiencies from the City of Los Angeles (personal communication, City of Los Angeles, June 2010).

To simulate the impacts of the rehabilitation project on lake water quality, the following conservative assumptions were made:

- Reclaimed water from the Glendale Water Reclamation Plant will be used for irrigation of park areas and supplemental water additions (see Appendix G [Monitoring Data] for water quality data for this source).
- The volume of reclaimed water used for supplemental water additions will be 15.5 ac-ft/yr based on a worst case scenario of evaporative losses of 55,000 gpd for three months straight with no wet or dry weather flows to offset these losses.

Simulating this future scenario for Echo Park Lake with the calibrated NNE BATHTUB model yields a total nitrogen concentration of 0.79 mg-N/L, a total phosphorus concentration of 0.10 mg-P/L, and a chlorophyll *a* concentration of 12µg/L. These simulated in-lake concentrations are based on the reduction efficiencies reported for the vortex/constructed wetland/recirculation system. If reductions are based on the vortex/constructed wetland system without recirculation, the simulated in-lake total phosphorus concentration is not predicted to meet the target of 0.12 mg-P/L regardless of the assumptions regarding supplemental water additions (potable versus reclaimed, with or without supplemental water additions, etc.). If the rehabilitation project does not result in the assumed reduction efficiencies of 80 percent for total nitrogen and 86 percent for total phosphorus, pre-treatment or additional treatment of the wet weather and dry weather flows may be necessary to meet the in-lake target concentrations.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

6.3 Lead Impairment

Echo Park Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), USEPA and local agencies collected 61 additional samples (12 wet weather) between November 2004 and March 2010 to evaluate current water quality conditions. There were only four dissolved lead exceedances in 61 samples (Appendix G, Monitoring Data). Therefore, Echo Park Lake meets lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Echo Park Lake not be identified as impaired by lead in California’s next 303(d) list.

6.4 Copper Impairment

Echo Park Lake was listed as impaired for copper in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), USEPA and local agencies collected 60 additional samples (12 wet weather) between November 2004 and March 2010 to evaluate current water quality conditions. There were only four dissolved copper exceedances in 60 samples (Appendix G, Monitoring Data). Therefore, Echo Park Lake meets copper water quality standards, and USEPA concludes that preparing a TMDL for copper is unwarranted at this time. USEPA recommends that Echo Park Lake not be identified as impaired by copper in California’s next 303(d) list.

6.5 PCB Impairment

Polychlorinated biphenyls (PCBs) consist of a family of many related congeners. The individual congeners are often referred to by their “BZ” number. Environmental analyses may address individual congeners, homologs (groups of congeners with the same number of chlorine atoms), equivalent concentrations of the commercial mixtures of PCBs known by the trade name Aroclors, or total PCBs. The environmental measurements and targets described in this section are in terms of total PCBs, defined as the “sum of all congener or isomer or homolog or Aroclor analyses” (CTR, 40 CFR 131.38(b)(1) footnote v).

The PCB impairment of Echo Park Lake affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. PCBs are no longer in production. While some loading of PCBs continues to occur in watershed runoff, the primary source of PCBs in the water column and aquatic life in Echo Park Lake is from historic loads stored in the lake sediments. Like other organochlorine compounds, PCBs accumulate in aquatic organisms and biomagnify in the food chain. As a result, low environmental exposure concentrations can result in unacceptable levels in higher trophic level fish in the lake.

6.5.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Echo Park Lake include REC1, REC2, WARM, WILD, and MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of PCBs are currently impairing the REC1, REC2, and WARM uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

6.5.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of PCBs in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by the OEHHA (2008) for fish consumption. The numeric targets used for PCBs are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for PCBs in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0005 mg/L, or 0.5 µg/L, total PCBs in water. The Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this report is designated WARM, at a minimum, and must meet this requirement. A chronic criterion for the sum of PCB compounds in freshwater systems to protect aquatic life is included in the CTR as 0.014 µg/L (USEPA, 2000a). The CTR also provides a human health-based water quality criterion for the consumption of both water and organisms and organisms only of 0.00017 µg/L (0.17 ng/L). The human health criterion of 0.17 ng/L is the most restrictive applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in Macdonald et al. (2000) for the threshold effects concentration (TEC) for total PCBs in sediment is 59.8 µg/kg (ng/dry g) dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The existing sediment PCB concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for PCBs defined by the OEHHA (2008) is 3.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For total PCBs, the corresponding sediment concentration target determined using the BSAF is 1.77 µg/kg dry weight, as described in detail in Section 6.5.5. All applicable targets are shown below in Table 6-8. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 6-8. PCB Targets Applicable to Echo Park Lake

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	3.6
Sediment (µg/kg dry weight)	Consensus-based TEC	59.8
Sediment (µg/kg dry weight)	BSAF-derived target	1.77
Water (ng/L)	CTR	0.17

Note: Shaded cells represent the selected targets for this TMDL.

6.5.3 Summary of Monitoring Data

This section summarizes the monitoring data for Echo Park Lake related to the PCB impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data). For PCBs, as well as other organochlorine compounds, sample analyses include both a detection limit and a method reporting limit. For example, a typical detection limit for total PCBs in sediment reported by UCLA is 0.53 µg/kg dry weight, while the reporting limit is 15 µg/kg dry weight.

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008. In all three samples PCB congeners were detected, but below reporting limits of 15 ng/L. Water samples from Echo Park Lake were also collected by the Regional Board on December 18, 2008 at four stations. PCBs at all stations were below the detection limit of 1 ng/L. A summary of the water column data is shown in Table 6-9.

Table 6-9. Summary of Water Column Samples for PCBs in Echo Park Lake

Station	Average Water Concentration (ng/L)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
NE near LA City Storm Drain	(0.5)	1	0	0
W near County Storm Drain	(0.5)	1	0	0
South	[2.72]	3	2	2
North, Lotus Bed	[4.47]	2	1	1
Northeast	(0.5)	1	0	
In-Lake Average[2]	[1.74]			
CTR Water Column Target	0.17			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Echo Park Lake samples from summer 2008 were analyzed for pollutant concentrations associated with suspended sediments in the lake. Samples were analyzed at two stations with detection limits ranging from 3.19 µg/kg to 10.05 µg/kg dry weight, and reporting limits ranging from 31.95 µg/kg to 100.5 µg/kg dry weight. In one sample, PCB congener BZ-31 was detected at 117 µg/kg dry weight, while congener BZ-153 was also detected, but not above the reporting limit.

UCLA collected bed sediment samples at four locations in Echo Park Lake in summer and fall 2008. Samples related to tributaries were collected in the lake near the tributary outfalls. Several PCB congeners were detected in the summer 2008 sediment samples, with only one station with all congeners below detection limits.

Sediment sampling was also conducted by the Regional Board at three stations on December 1, 2009. PCBs were quantified at all three stations. PCB congeners BZ-18, BZ-95, BZ-101, and BZ-110 were quantified at all locations. Other congeners were also quantified at one or two locations. A summary of the sediment data is shown in Table 6-10. The lake-wide average of 40.29 µg/kg dry weight is greater than the concentration near outfalls (24.16 µg/kg dry weight), and both are less than the consensus-based TEC of 59.8 µg/kg dry weight.

Table 6-10. Summary of Sediment Samples for PCBs in Echo Park Lake, 2008-2009

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
NE near LA City Storm Drain	2.98	3	3	2
W near County Storm Drain	31.41	2	1	0
South	29.85	1	1	0
North, Lotus Bed	70.01	2	1	0
Northeast	(0.30)	1	0	0
NW Arm near outfall	38.10	1	1	0
Center Lake	72.55	2	2	0
Center Lake South	77.10	1	1	0
In-Lake Average[2]	40.29			
Influent Average	24.16			
Consensus-based TEC	59.8			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Four fish samples (composites of filets from five fish) were collected and analyzed for PCBs as Aroclor equivalents between 1987 and 1991. In 1987, a largemouth bass and bullhead sample reported 84 ppb and 50 ppb wet weight, respectively. Another largemouth bass sample was analyzed in 1991 and reported as 0 ppb (the detection limits for the historical fish samples are not reported). In 1992, the PCB concentration in a largemouth bass composite sample was 60 ppb. The average reported PCB concentration in all samples from the 1980s and 1990s was 48.5 ppb, including the reported zero. Results from the individual samples are shown in Appendix G (Monitoring Data).

Considering only data collected in the past 10 years, the average concentration of total PCBs in largemouth bass was 49.0 ppb wet weight, based on the two largemouth bass composite samples collected by SWAMP in the summer of 2007 with an average lipid fraction of 0.396 percent and an additional sample from April 2010 with a lipid fraction of 0.315 percent. Three composite samples of bottom-feeding carp (Trophic Level 3) were also analyzed. These yielded an average total PCB concentration of 81.8 ppb wet weight with an average lipid fraction of 1.263 percent. The recent fish-tissue data for Echo Park Lake are summarized in Table 6-11.

Table 6-11. Summary of Recent Fish Tissue Samples for PCBs in Echo Park Lake

Sample Date	Fish Species	Total PCBs (ppb wet weight)[1]
11 June 2007	Largemouth Bass	64.7
11 June 2007	Largemouth Bass	31.5
13 April 2010	Largemouth Bass	50.9
11 June 2007	Common Carp	119.0
11 June 2007	Common Carp	82.6
13 April 2010	Common Carp	43.9
2007 - 2010 Average – Largemouth Bass		49.0
2007 - 2010 Average – Common Carp		81.8
FCG		3.6

[1] Composite sample of filet from five individuals.

In sum, recent fish tissue samples collected from Echo Park Lake are all elevated above the OEHHA fish consumption guidelines for total PCBs. Concentrations in sediment are, on average, below the consensus-based TEC, although individual samples exceed this value. Concentrations in water have not been quantified; however, several 2008 samples were above detection limits that exceed the CTR criterion, although less than the reporting limit.

6.5.4 Source Assessment

PCBs in Echo Park Lake are primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed PCB concentrations on sediment near inflows to the lake. Watershed loads of PCBs may arise from spills from industrial and commercial uses, improper disposal, and atmospheric deposition. Industrial and commercial spills will tend to be associated with specific land areas, such as older industrial districts, junk yards, and transformer substations. Improper disposal could have occurred at various locations (indeed, waste PCB oils were sometimes used for dust control on dirt roads in the 1950s). Atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated PCB load within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas. Although supplemental water additions of potable water makes up a significant amount of the flow to Echo Park Lake it does not contribute sediment load and is considered to not contribute significantly to PCB loading (total suspended sediment measured non-detect in two samples collected August 4th 2009).

The average concentration of PCBs on incoming sediment was estimated to be 24.16 µg/kg dry weight and the estimated annual sediment load to Echo Park Lake is 1.32 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load of PCBs is approximately 0.029 g/yr. Table 6-12 shows the annual PCB load estimated from each jurisdiction.

Table 6-12. Total PCB Loads Estimated for Each Jurisdiction and Subwatershed in the Echo Park Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total PCB Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	0.044	0.0010	3.35%
Northern	City of Los Angeles	MS4 Stormwater[1]	0.98	0.021	74.24%
Southern	Caltrans	State Highway Stormwater[1]	0.0037	0.0001	0.28%
Southern	City of Los Angeles	MS4 Stormwater[1]	0.29	0.0064	22.13%
Total Load from Watershed			1.32	0.029	100%

[1] This input includes effluent from storm drain systems during both wet and dry weather.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of PCBs directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

6.5.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of PCBs into Echo Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of PCBs in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. PCBs are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of PCBs will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Echo Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment PCB concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to additional uptake of organochlorine compounds from the sediment by bottom feeding fish. The existing fish tissue concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of PCBs are likely to have declined steadily since the cessation of production and use of the chemical. For PCBs in Echo Park Lake the ratios of the FCG to existing concentrations are:

TL4: 3.6/49.0 = 0.0735

TL3: 3.6/81.8 = 0.0440

The lower ratio, obtained for the TL3 fish, corresponds to the trophic level requiring the greatest reductions to achieve the fish tissue target. This ratio is applied to the observed in-lake sediment concentration of 40.29 µg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 1.77 µg/kg dry weight (Table 6-13).

Table 6-13. Fish Tissue-Based Total PCB Concentration Targets for Sediment in Echo Park Lake

Total PCB Concentration	Sediment (µg/kg dry weight)
Existing	40.29
BSAF-derived Target	1.77
Required Reduction	95.6%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 59.8 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.17 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 3,230 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 0.76 g/yr, or 0.02 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of PCBs in Echo Park Lake is primarily due to the storage of historic loads of PCBs in the lake sediment.

6.5.6 TMDL Summary

Because PCB impairment in Echo Park Lake is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake

The PCB TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 1.77 µg/kg dry weight total PCBs. The wasteload allocations and load allocations are also equal to 1.77 µg/kg dry weight total PCBs in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

6.5.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for total PCBs ("Alternative WLAs if the Fish Tissue Target is Met") described in Section 0. The alternative wasteload allocations will supersede the wasteload allocations in Section 6.5.6.1.1 if the conditions described in Section 0 are met.

6.5.6.1.1 Wasteload Allocations

The entire watershed of Echo Park Lake is contained in an MS4 jurisdiction, and watershed loads are therefore assigned WLAs. Relevant permit numbers are

- County of Los Angeles (including the city of Los Angeles): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003

PCBs in water flowing into Echo Park Lake are below detection limits, and most PCB load is expected to move in association with sediment. Therefore, suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for PCBs in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved PCBs and PCBs associated with suspended sediment. The existing average concentration of sediment entering the lake is 24.16 µg/kg dry weight. Therefore, a reduction of (24.16 – 1.77)/24.16 = 92.7 percent is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 6-14 and each wasteload allocation must be met at the point of discharge.

Table 6-14. Wasteload Allocations for Total PCBs in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	1.77	0.17
Northern	City of Los Angeles	MS4 Stormwater[1]	1.77	0.17
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	1.77	0.17
Southern	Caltrans	State Highway Stormwater[1]	1.77	0.17
Southern	City of Los Angeles	MS4 Stormwater[1]	1.77	0.17

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

6.5.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 6-14 will be superseded, and the wasteload allocations in Table 6-15 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350 mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 6-15, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 6-15. Alternative Wasteload Allocations for Total PCBs in Echo Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	59.8	0.17
Northern	City of Los Angeles	MS4 Stormwater[1]	59.8	0.17
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	59.8	0.17
Southern	Caltrans	State Highway Stormwater[1]	59.8	0.17
Southern	City of Los Angeles	MS4 Stormwater[1]	59.8	0.17

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

6.5.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for total PCBs ("Alternative LAs if the Fish Tissue Target is Met") described in Section 6.5.6.2.2. The alternative load allocations will supersede the load allocations in Section 6.5.6.2.1 if the conditions described in Section 6.5.6.2.2 are met.

6.5.6.2.1 Load Allocations

No part of the watershed of Echo Park Lake is outside MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to atmospheric deposition of PCBs. The legacy PCB stored in lake sediment is the major cause of use impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (City of Los Angeles) should achieve a PCB concentration of 1.77 µg/kg dry weight in lake bottom sediments (Table 6-16).

Table 6-16. Load Allocations for Total PCBs in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	1.77

6.5.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 6-16 will be superseded, and the load allocations in Table 6-17 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 6-17, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Table 6-17. Alternative Load Allocations for Total PCBs in Echo Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	59.8

6.5.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

6.5.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate PCBs, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

6.5.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the PCB WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99th percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Echo Park Lake watershed. USGS Station 11102000, Mission Creek near Montebello, CA, was selected as a surrogate for flow determination. This gage is the closest USGS StreamStats gage in the Los Angeles River Basin with a relatively small drainage area (2,662 acres). The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for Mission Creek (30.2 cfs) (Wolock, 2003). To estimate the peak flow to Echo Park Lake, the 99th percentile flow for Mission Creek was scaled down by the ratio of drainage areas (784 acres/2,662 acres; Echo Park Lake watershed area/Mission Creek watershed area at the gage). The resulting peak flow estimate for Echo Park Lake is 8.89 cfs.

The event mean concentration of sediment in stormwater (55.8 mg/L) was calculated from the estimated existing watershed sediment load of 1.32 tons/yr (Table 6-12) divided by the total storm flow volume entering the lake (17.4 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (8.98 cfs) yields a daily maximum sediment load from stormwater of1226 kg/d (1.35 tons/d). Applying the wasteload allocation concentration of 1.77 µg total PCBs per dry kg of sediment yields the stormwater daily maximum allowable load of 0.0022 g/d of total PCBs. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

6.5.6.6 Future Growth

USEPA regulates PCBs under the Toxic Substances Control Act (TSCA), which generally bans the manufacture, use, and distribution in commerce of the chemicals in products at concentrations of 50 parts per million or more, although TSCA allows USEPA to authorize certain uses, such as to rebuild existing electrical transformers during the transformers' useful life. Therefore, no additional allowance is made for future growth in the PCB TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

6.6 Chlordane Impairment

Total chlordane consists of a family of related chemicals, including cis- and trans-chlordane, oxychlordane, trans-nonachlor, and cis-nonachlor. Observations and targets discussed in this section all refer to total chlordane. Chlordane was used as a pesticide in field, commercial, and residential uses. Chlordane is no longer in production, but persists in the environment from legacy loads.

The chlordane impairment of Echo Park Lake affects beneficial uses related to recreation, municipal water supplies, wildlife health, and fish consumption. While some loading of chlordane continues to occur in watershed runoff, the primary source of chlordane in the water column and aquatic life in Echo Park Lake is from historic loads stored in the lake sediments. Chlordane, like other OC pesticides and PCBs, accumulates in aquatic organisms and biomagnifies in the food chain. As a result, low environmental concentrations can result in unacceptable levels in higher trophic level fish in the lake. The approach for chlordane is similar to that for PCBs.

6.6.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Echo Park Lake include REC1, REC2, WARM, WILD, and MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of chlordane are currently impairing the REC1,REC2 and WARM uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

6.6.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of chlordane listed in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), for chlordane defined by the Office of Environmental Health Hazard Assessment (OEHHA) for fish consumption. The numeric targets used for chlordane are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for chlordane in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0001 mg/L, or 0.1 µg/L. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Acute and chronic criterion for chlordane in freshwater systems are defined by the California Toxics Rule as 2.4 µg/L and 0.0043 µg/L, respectively (USEPA, 2000a). The CTR also includes human health criteria for the consumption of water and organisms and for the consumption of organisms only as 0.00057 µg/L and 0.00059 µg/L, respectively (USEPA, 2000a). For Echo Park Lake, the Regional Board has determined that the appropriate human health criterion is 0.00059 µg/L (0.59 ng/L) as the MUN use is not an existing use and may be removed.

For sediment, the consensus-based sediment quality guidelines provided in Macdonald et al. (2000) for the threshold effects concentration (TEC) for chlordane is 3.24 µg/kg (µg/kg dry weight) dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The existing sediment chlordane concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for chlordane defined by the OEHHA (2008) is 5.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For chlordane, the corresponding sediment concentration target determined using the BSAF is 2.10 µg/kg dry weight, as described in Section 6.6.5. All applicable targets are shown below in Table 6-18. For sediment the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 6-18. Total Chlordane Targets for Echo Park Lake

Media	Source	Target
Fish (ppb wet weight)	OEHHA FCG	5.6
Sediment (ng /dry g)	Consensus-based TEC	3.24
Sediment (µg/kg dry weight)	BSAF-derived target	2.10
Water (ng/L)	CTR	0.59

Note: Shaded cells represent the selected targets for this TMDL.

6.6.3 Summary of Monitoring Data

This section summarizes the monitoring data for Echo Park Lake related to the chlordane impairment. Addition details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 at two locations within Echo Park Lake. These analyses measured cis- and trans-chlordane, but not oxychlordane or nonachlor. All water column samples were less than the detection limit for chlordane (1.5 ng/L; the detection limit for chlordane is higher than the water column criterion of 0.59 ng/L). No additional water column sampling for chlordane has been conducted in Echo Park Lake.

A summary of the water column data is shown in Table 6-19.

Table 6-19. Summary of Water Column Samples for Total Chlordane in Echo Park Lake

Station	Average Water Concentration(ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
South	(0.75)[2]	2	0
North, Lotus Bed	(0.75)	1	0
In-Lake Average[3]	(0.75)		
CTR Criterion	0.59		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no chlordanes were quantified in any of the collected samples.

[3] Overall average is the average of individual station averages.

Concentrations of chlordane on suspended sediment were also analyzed at two in-lake stations during the summer of 2008 by UCLA; both were less than the detection limits (3.19 µg/kg to 10.05 µg/kg dry weight). Porewater was sampled by UCLA in both the fall and spring of 2008. Specifically, chlordane concentrations in the porewater sampled at four sites during the summer of 2008 were all less than the detection limit of 15 ng/L; both sites sampled during the fall of 2008 were also below detection limits of 15 ng/L to 1,500 ng/L.

UCLA also collected sediment samples at five locations in Echo Park Lake during summer and fall 2008. As with the water column analyses by UCLA, these report cis- and trans-chlordane, but not oxychlordane or nonachlor. Of the nine total samples, all but one resulted in chlordane concentrations below the detection limit (which ranged from 0.44 µg/kg to 1.23 µg/kg dry weight). One sediment sample collected during summer 2008 resulted in a sample average concentration of 4.14 µg/kg dry weight, which is greater than the consensus-based TEC of 3.24 µg/kg dry weight. Three in-lake locations were sampled by the Regional Board and USEPA on December 1, 2009, resulting in reportable concentrations of 4.1 µg/kg to 22.25 µg/kg dry weight. These analyses do include oxychlordane and nonachlor.

All lake stations were averaged to estimate an exposure concentration for total chlordane in Echo Park Lake sediments of 4.43 µg/kg dry weight (with non-detects included at one-half the detection limit for each sample). Stations located near outfalls are taken as an estimate of the concentrations on incoming sediment. A summary of the sediment data is shown in Table 6-20.

Table 6-20. Summary of Sediment Samples for Total Chlordane in Echo Park Lake

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
NE near LA City Storm Drain	(0.44)	3	0	0
W near County Storm Drain	2.25	2	1	0
South	(0.46)	1	0	0
North, Lotus Bed	(0.53)	2	0	0
Northeast	(0.30)	1	0	0
NW Arm, near outfall	22.25	1	1	0

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
Center Lake	5.15	2	1	0
Center Lake S	4.10	1	1	0
In-Lake Average[2]	4.43			
Influent Average	8.31			
Consensus-based TEC	3.24			

[1] Total chlordane in a sample represents the sum of all reported measurements for alpha and gamma chlordane, oxychlordane, and cis- and trans-nonachlor, including results reported below the method reporting limit. If all components were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no chlordane quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations of total chlordane from Echo Park Lake have been analyzed in largemouth bass, common carp, and bullhead (SWAMP and TSMP). Four fish samples (composites of filets from five fish) were collected and analyzed for total chlordane between 1987 and 1991. In 1987, concentrations in a largemouth bass and a bullhead composite sample were reported at 17.8 and 66 ppb wet weight, respectively. Two additional largemouth bass samples were analyzed in 1991, with concentrations reported as 0 ppb (the detection limits for the historical fish samples are not reported).

Considering only data collected in the past 10 years, the average concentration of chlordane in largemouth bass was 4.70 ppb wet weight, based on the three largemouth bass composite samples collected in the summer of 2007 and April 2010 with an average lipid fraction of 0.37 percent. Three composite samples of bottom-feeding common carp (Trophic Level 3) were also analyzed. These yielded an average total chlordane concentration of 11.85 ppb wet weight with an average lipid fraction of 1.26 percent. The recent fish-tissue data for Echo Park Lake are summarized in Table 6-21.

Table 6-21. Summary of Recent Fish Tissue Samples for Total Chlordane in Echo Park Lake

Sample Date	Fish Species	Total Chlordane (ppb wet weight)[1]
11 June 2007	Largemouth Bass	8.534
11 June 2007	Largemouth Bass	2.037
13 April 2010	Largemouth Bass	2.517
11 June 2007	Common Carp	18.41
11 June 2007	Common Carp	12.92
13 April 2010	Common Carp	4.216
2007 - 2010 Average – Largemouth Bass		4.70
2007 - 2010 Average – Common Carp		11.85
FCG		5.6

[1] Composite samples of filet from five individuals.

In sum, recent fish tissue concentrations in Echo Park Lake are above the FCG in two of three samples for common carp, and in one of three largemouth bass composite samples. The average concentration in sediment is below the consensus-based TEC, although individual samples exceed the TEC. Water column samples have all been below detection limits.

6.6.4 Source Assessment

Chlordane in Echo Park Lake is primarily due to historical loading and storing within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed chlordane concentrations on sediment near inflows to the lake. Watershed loads of chlordane may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas. Although supplemental water additions of potable water makes up a significant amount of the flow to Echo Park Lake it does not contribute sediment load and is considered to no contribute significantly to chlordane loading (total suspended sediment measured non-detect in two samples collected August 4th 2009).

The average concentration of total chlordane on incoming sediment is estimated to be 8.31 µg/kg dry weight (Table 6-20) and the annual sediment load to Echo Park Lake is 1.32 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load of chlordane is approximately 0.0099 g/yr (Table 6-22).

Table 6-22. Total Chlordane Loads Estimated for Each Jurisdiction and Subwatershed in the Echo Park Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total Chlordane Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	0.044	0.0003	3.35%
Northern	City of Los Angeles	MS4 Stormwater[1]	0.98	0.0074	74.24%
Southern	Caltrans	State Highway Stormwater[1]	0.0037	0.00003	0.28%
Southern	City of Los Angeles	MS4 Stormwater[1]	0.29	0.0022	22.13%
Total Load from Watershed			**1.32**	**0.0099**	**100.00%**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of total chlordane directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

6.6.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of total chlordane into Echo Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of total chlordane in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Chlordanes are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of total chlordane will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Echo Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment chlordane concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to additional uptake of organochlorine compounds from the sediment by bottom feeding fish. The existing fish tissue concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of chlordane are likely to have declined steadily since the cessation of production and use of the chemical. For chlordane in Echo Park Lake the ratios of the FCG to existing concentrations are:

TL4: 5.6/4.70 = 1.191

TL3: 5.6/11.85 = 0.473

The lower ratio, obtained for the TL3 fish, corresponds to the trophic level requiring the greatest reductions to achieve the fish tissue target. This ratio is applied to the observed sediment concentration of 4.43 µg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 2.10 µg/kg dry weight (Table 6-23).

Table 6-23. Fish Tissue-Based Chlordane Concentration Targets for Sediment in Echo Park Lake

Total Chlordane Concentration	Sediment (µg/kg dry weight)
Existing	4.43
BSAF-derived Target	2.10
Required Reduction	52.8%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 3.24 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.59 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 63.8 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 0.0099 g/yr, or 0.02 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of chlordane in Echo Park Lake is primarily due to the storage of historic loads of chlordane in the lake sediment.

6.6.6 TMDL Summary

Because chlordane impairment in Echo Park Lake is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake.

The chlordane TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 2.10 µg/kg dry weight chlordane. The wasteload allocations and load allocations are also equal to 2.10 µg/kg dry weight chlordane in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

6.6.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for chlordane (“Alternative WLAs if the Fish Tissue Target is Met”) described in Section 6.6.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 6.6.6.1.1 if the conditions described in Section 6.6.6.1.2 are met.

6.6.6.1.1 Wasteload Allocations

The entire watershed of Echo Park Lake is contained in an MS4 jurisdiction, and therefore receives WLAs. Relevant permit numbers are

- County of Los Angeles (including the city of Los Angeles): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003

Total chlordane concentrations in water flowing into Echo Park Lake are below detection limits, and most chlordane load is expected to move in association with sediment. Therefore, the suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for chlordane in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved chlordane and chlordane associated with suspended sediment. The existing concentration of sediment entering the lake is 8.31 µg/kg dry weight. Therefore, a reduction of (8.31 – 2.10)/8.31 = 74.7 percent is required on the sediment-associated load from the watershed. The reduction in watershed load is slightly greater than the reduction needed for in-lake sediments because the estimated concentration on influent sediment is greater than the lake-wide average.

The wasteload allocations are shown in Table 6-24 and each wasteload allocation must be met at the point of discharge.

Table 6-24. Wasteload Allocations for Total Chlordane in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	2.10	0.59
Northern	City of Los Angeles	MS4 Stormwater[1]	2.10	0.59
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	2.10	0.59
Southern	Caltrans	State Highway Stormwater[1]	2.10	0.59
Southern	City of Los Angeles	MS4 Stormwater[1]	2.10	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

6.6.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 6-24 will be superseded, and the wasteload allocations in Table 6-25 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 6-25, and
3. USEPA does not object to the Regional Board's determination within sixty days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 6-25. Alternative Wasteload Allocations for Total Chlordane in Echo Park Lake if Fish Tissue Targets are Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	3.24	0.59
Northern	City of Los Angeles	MS4 Stormwater[1]	3.24	0.59
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	3.24	0.59
Southern	Caltrans	State Highway Stormwater[1]	3.24	0.59
Southern	City of Los Angeles	MS4 Stormwater[1]	3.24	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

6.6.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for chlordane ("Alternative LAs if the Fish Tissue Target is Met") described in Section 6.6.6.2.2. The alternative load allocations will supersede the load allocations in Section 6.6.6.2.1 if the conditions described in Section 6.6.6.2.2 are met.

6.6.6.2.1 Load Allocations

No part of the watershed of Echo Park Lake is outside MS4 jurisdiction; therefore no LAs are assigned to the watershed loads. No load is allocated to atmospheric deposition of chlordane. The legacy chlordane stored in lake sediment is the major cause of impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (city of Los Angeles) should achieve a total chlordane concentration of 2.10 µg/kg dry weight in lake bottom sediments (Table 6-26).

Table 6-26. Load Allocations for Total Chlordane in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	2.10

6.6.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 6-26 will be superseded, and the load allocations in Table 6-27 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 6-27, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Table 6-27. Alternative Load Allocations for Total Chlordane in Echo Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	3.24

6.6.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

6.6.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate chlordane, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

6.6.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the total chlordane WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99th percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

The daily maximum allowable load in Echo Park Lake is calculated from the estimated 99th percentile flow to the Lake multiplied by the event mean concentration consistent with achieving the long-term loading targets, described above in the PCBs section. USGS Station 11102000, Mission Creek near Montebello, CA, was selected as a surrogate for flow determination for flow to the lake, as described in the PCBs section (Section 6.5.6.5).

The event mean concentration of sediment in stormwater (55.8 mg/L) was calculated from the estimated existing watershed sediment load of 1.32 tons/yr (Table 6-22) divided by the total storm flow volume reaching the lake (17.4 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (8.98 cfs) yields a daily maximum sediment load from stormwater of 1226 kg/d (1.35 tons/d). Applying the wasteload allocation concentration of 2.10 µg total chlordane per dry kg of sediment yields the stormwater daily maximum allowable load of 0.0026 g/d of total chlordane. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

6.6.6.6 Future Growth

The manufacture and use of chlordane is currently banned. Therefore, no additional allowance is made for future growth in the chlordane TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

6.7 DIELDRIN IMPAIRMENT

Dieldrin is a chlorinated insecticide originally developed as an alternative to DDT and was in wide use from the 1950s to the 1970s. Dieldrin in the environment also arises from use of the insecticide aldrin. Aldrin is not itself toxic to insects, but is metabolized to dieldrin in the insect body. The use of both dieldrin and aldrin was discontinued in the 1970s.

The dieldrin impairment of Echo Park Lake affects beneficial uses related to recreation, municipal water supplies, wildlife health, and fish consumption. Dieldrin, like PCBs and chlordane, is an organochlorine compound that is strongly sorbed to sediment and is no longer in production. As such, the approach for dieldrin impairment is similar to that for PCBs and chlordane.

6.7.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Echo Park Lake include REC1, REC2, WARM, WILD, and MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of dieldrin are currently impairing the REC1,REC2 and WARM uses by causing toxicity to aquatic organisms, raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories), and impair sport fishing recreational uses. At high enough concentrations WILD and MUN uses could become impaired.

6.7.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of dieldrin in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by the OEHHA (2008) for fish consumption. The numeric targets for dieldrin are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column targets for dieldrin in the Basin Plan are associated with a specific beneficial use. The Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this TMDL is designated WARM, at a minimum, and must meet this requirement. Acute and chronic criteria for the protection of aquatic life and wildlife in freshwater systems are included in the CTR for dieldrin as 0.24 µg/L and 0.056 µg/L, respectively (USEPA, 2000a). The CTR also provides a human health-based water quality criterion for the consumption of organisms only and the consumption of water and organisms as 0.00014 µg/L (USEPA, 2000a). The human health criterion of 0.00014 µg/L (0.14 ng/L) is the most restrictive of the applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) of dieldrin in sediment is 0.46 µg/kg (µg/kg dry weight). The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider “the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans).” The estimated existing sediment dieldrin concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for dieldrin defined by the OEHHA (2008) is 0.46 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For dieldrin, the corresponding sediment concentration target estimated using the BSAF approach is 0.80 µg/kg dry weight, as described in Section 6.7.5. All applicable targets are shown below in Table 6-28. For sediment the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 6-28. Dieldrin Targets for Echo Park Lake

Media	Source	Target
Fish (ppb wet weight)	OEHHA FCG	0.46
Sediment (µg/kg dry weight)	Consensus-based TEC	1.90
Sediment (µg/kg dry weight)	BSAF-derived target	0.80
Water (ng/L)	CTR	0.14

Note: Shaded cells represent the selected targets for this TMDL.

6.7.3 Summary of Monitoring Data

This section summarizes the monitoring data for Echo Park Lake related to the dieldrin impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the summer of 2008 with three samples at two locations within Echo Park Lake. All three water column samples were less than the detection limit for dieldrin (3 ng/L; the detection limit for dieldrin is higher than the water column criterion of 0.14 ng/L). No additional water column sampling for dieldrin has been conducted in Echo Park Lake.

A summary of the water column data is shown in Table 6-29.

Concentrations of dieldrin in suspended sediment were also analyzed at two in-lake stations during the summer of 2008 by UCLA, both were less than the detection limits (6.39 µg/kg to 20.10 µg/kg dry weight). Porewater was sampled by UCLA in both the summer and fall of 2008; dieldrin concentrations in all samples were less than the detection limits of 30 ng/L to 3,000 ng/L.

Table 6-29. Summary of Water Column Samples for Dieldrin in Echo Park Lake

Station	Average Water Concentration (ng/L)[1]	Number of Samples	Number of Samples Above Detection Limits
South	(1.50)[2]	2	0
North, Lotus Bed	(1.50)	1	0
In-Lake Average[2]	(1.50)		
CTR Criterion	0.14		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit. Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no dieldrin was quantified in any of the collected samples.

[2] Overall average is the average of individual station averages.

UCLA collected bed sediment samples at five locations in Echo Park Lake in summer and fall 2008. All nine samples analyzed by UCLA resulted in dieldrin concentrations below the detection limit (which ranged from 0.83 µg/kg to 2.46 µg/kg dry weight). Since the upper end of this range is greater than the consensus-based TEC for dieldrin sediment (1.9 µg/kg dry weight), exceedances cannot be ruled out. Three in-lake locations were sampled by the Regional Board and USEPA on December 1, 2009; all were below the detection limit (1 µg/kg dry weight). Stations located near outfalls are taken as an estimate of the concentrations on incoming sediment. Because dieldrin does appear in fish at levels greater than the FCG, and because these body burdens of dieldrin are believed to arise from the sediment, EPA decided to represent statistical estimates for the sediment concentrations of dieldrin by setting the concentration of non-detected samples to the detection limit. The estimated lake-wide average of < 1.39 µg/kg dry weight is less than the consensus-based TEC of 1.90 µg/kg dry weight. A summary of the sediment sampling is provided in Table 6-30.

Table 6-30. Summary of Sediment Samples for Dieldrin in Echo Park Lake

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples Above Detection Limits
NE near LA City Storm Drain	(1.76)	3	0
W near County Storm Drain	(1.19)	2	0

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples Above Detection Limits
South	(1.83)	1	0
North, Lotus Bed	(2.13)	2	0
Northeast	(1.20)	1	0
NW Arm, near outfall	(1.00)	1	0
Center Lake	(1.00)	1	0
Center Lake S	(1.00)	1	0
In-Lake Average[2]	(1.39)		
Influent Average	(1.32)		
Consensus-based TEC	1.90		

[1] Non-detect samples are included in reported averages at the detection limit. Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no dieldrin was detected in any of the collected samples.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations of dieldrin from Echo Park Lake have been analyzed in largemouth bass, common carp, and bullhead (SWAMP and TSMP). Four fish samples (composites of filets from five fish) were collected and analyzed for total dieldrin between 1987 and 1991. In 1987, concentrations in a largemouth bass and a bullhead composite sample were reported at 0 and 7 ppb wet weight, respectively. Two additional largemouth bass samples were analyzed in 1991, with concentrations reported as 0 ppb (the detection limits for the historical fish samples are not reported).

Considering only data collected in the past 10 years, the average concentration of dieldrin in largemouth bass was 0.716 ppb wet weight, based on the three largemouth bass composite samples collected by SWAMP in the summer of 2007 and April 2010 with an average lipid fraction of 0.37 percent. Three composite samples of bottom-feeding common carp (Trophic Level 3) were also analyzed. These yielded an average dieldrin concentration of 0.935 ppb wet weight with an average lipid fraction of 1.26 percent. The recent fish-tissue data for Echo Park Lake are summarized in Table 6-31.

Table 6-31. Summary of Recent Fish Tissue Samples for Dieldrin in Echo Park Lake

Sample Date	Fish Species	Dieldrin (ppb wet weight)[1]
11 June 2007	Largemouth Bass	0.848
11 June 2007	Largemouth Bass	0.585
13 April 2010	Largemouth Bass	[0.453][2]
11 June 2007	Common Carp	1.08
11 June 2007	Common Carp	0.79
13 April 2010	Common Carp	0.538
2007 - 2010 Average – Largemouth Bass		0.650

Sample Date	Fish Species	Dieldrin (ppb wet weight)[1]
2007 - 2010 Average – Common Carp		0.803
FCG		0.46

[1] Composite samples of filet from five individuals.

[2] Values in square brackets are reported concentrations below the practical reporting limit and are included in the averages.

In sum, five of six recent fish tissue concentrations in Echo Park Lake are above the FCG for dieldrin in both common carp and largemouth bass composite samples. Sediment and water column concentrations have all been below detection limits; however, the maximum detection limit in sediment is less than the consensus-based TEC.

6.7.4 Source Assessment

Dieldrin in Echo Park Lake is suspected to be primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading and direct atmospheric deposition to the lake are considered negligible sources of dieldrin. Stormwater loads from the watershed could not be directly estimated because all sediment and water samples were below detection limits. Watershed loads of dieldrin may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources within the watershed at this time. Therefore, an average concentration of sediment is applied to all contributing areas. Although supplemental water additions of potable water makes up a significant amount of the flow to Echo Park Lake it does not contribute sediment load and is considered to not contribute significantly to dieldrin loading (total suspended sediment measured non-detect in two samples collected August 4th 2009).

An upper-bound analysis for dieldrin is performed using the sediment load and detection limit to determine the maximum potential loading rate of dieldrin from the watershed. The dieldrin sediment concentration is assigned based on the estimate of concentration on influent sediment from sample detection limits of 1.32 µg/kg dry weight and the annual sediment load to Echo Park Lake is 1.32 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated upper bound on the wet weather load from the watershed is 0.0016 g/yr or less (Table 6-32).

Table 6-32. Maximum Potential Dieldrin Loads for Each Jurisdiction and Subwatershed in the Echo Park Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Upper-Bound Potential Current Dieldrin Load (g/yr)
Northern	Caltrans	State Highway Stormwater[1]	<0.00005
Northern	City of Los Angeles	MS4 Stormwater[1]	<0.00117
Southern	Caltrans	State Highway Stormwater[1]	<0.00000
Southern	City of Los Angeles	MS4 Stormwater[1]	<0.00035
Total Load from Watershed			<0.0016

This input includes effluent from storm drain systems during both wet and dry weather. As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of dieldrin directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

6.7.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of dieldrin into Echo Park Lake consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of dieldrin in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Dieldrin is strongly sorbed to sediments and has a long half-life in sediment and water. Incoming loads of dieldrin will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Echo Park Lake are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The estimated existing sediment dieldrin concentrations in Echo Park Lake are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to additional uptake of OC pesticides and PCBs from the sediment by bottom feeding fish. The existing fish tissue concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of dieldrin are likely to have declined steadily since the cessation of production and use of the chemical. For dieldrin in Echo Park Lake the ratios of the FCG to existing concentrations are:

TL4: 0.46/0.650 = 0.708

TL3: 0.46/0.803 = 0.573

The lower ratio, obtained for the TL3 fish, corresponds to the trophic level requiring the greatest reductions to achieve the fish tissue target. This ratio is applied to the estimated in-lake sediment concentration. Analyses of sediment concentrations are, however, below detection limits. Using an estimated concentration of 1.39 µg/kg dry weight based on the sample detection limits, the resulting target concentration would be 0.80 µg/kg dry weight to obtain FCGs. Calculation with a literature-based BSAF (Appendix G, Monitoring Data) suggests that even lower concentrations might be needed. However, the literature-based BSAF is highly uncertain and may not be directly applicable to conditions in Echo Park Lake. Therefore, the target based on the detection limits is used, with acknowledgment that the estimate may need to be refined if additional data are collected at lower detection limits. The resulting fish tissue based target concentration of dieldrin in the sediment of Echo Park Lake is shown in Table 6-33.

Table 6-33. Fish Tissue-Based Dieldrin Concentration Targets for Sediment in Echo Park Lake

Dieldrin Concentration	Sediment (µg/kg dry weight)
Existing	< 1.39
BSAF-derived Target	0.80
Required Reduction	< 50.7%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 1.90 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.14 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

6.7.6 TMDL Summary

Because the dieldrin impairment in Echo Park Lake is most likely due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake

The dieldrin TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 0.80 µg/kg dry weight dieldrin. The wasteload allocations and load allocations are also equal to 0.80 µg/kg dry weight dieldrin in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

6.7.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for dieldrin (“Alternative WLAs if the Fish Tissue Target is Met”) described in Section 6.7.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 6.7.6.1.1 if the conditions described in Section 6.7.6.1.2 are met.

6.7.6.1.1 Wasteload Allocations

The entire watershed of Echo Park Lake is contained in an MS4 jurisdiction, and therefore receives WLAs. Relevant permit numbers are

- County of Los Angeles (including the city of Los Angeles): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001

- Caltrans: Order No 99-06-DWQ, CAS000003

Dieldrin concentrations in sediment and water flowing into Echo Park Lake are below detection limits, but most dieldrin load is expected to move in association with sediment. Therefore, suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for dieldrin in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved dieldrin and dieldrin associated with suspended sediment. Comparing the sediment concentration target to the average detection limit for the influent samples of 1.32 µg/kg dry weight suggests that a reduction of approximately 39 percent in dieldrin loads is needed. The wasteload allocations are shown in Table 6-34 and each wasteload allocation must be met at the point of discharge.

Table 6-34. Wasteload Allocations for Dieldrin in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	0.80	0.14
Northern	City of Los Angeles	MS4 Stormwater[1]	0.80	0.14
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	0.80	0.14
Southern	Caltrans	State Highway Stormwater[1]	0.80	0.14
Southern	City of Los Angeles	MS4 Stormwater[1]	0.80	0.14

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

6.7.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 6-34 will be superseded, and the wasteload allocations in Table 6-35 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 6-35, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 6-35. Alternative Wasteload Allocations for Dieldrin in Echo Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	1.90	0.14
Northern	City of Los Angeles	MS4 Stormwater[1]	1.90	0.14
Northern	General Industrial Stormwater Permittees (in the City of Los Angeles)[2]	General Industrial Stormwater[1]	1.90	0.14
Southern	Caltrans	State Highway Stormwater[1]	1.90	0.14
Southern	City of Los Angeles	MS4 Stormwater[1]	1.90	0.14

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The discharges governed by the general industrial stormwater permit are currently in the City of Los Angeles. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

6.7.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for dieldrin ("Alternative LAs if the Fish Tissue Target is Met") described in Section 6.7.6.2.2. The alternative load allocations will supersede the load allocations in Section 6.7.6.2.1 if the conditions described in Section 6.7.6.2.2 are met.

6.7.6.2.1 Load Allocations

None of the watershed of Echo Park Lake is outside MS4 jurisdiction; therefore no LAs are assigned to watershed loads. No load is allocated to atmospheric deposition of dieldrin. The legacy dieldrin stored in lake sediment is believed to be the major cause of impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (city of Los Angeles) should achieve a dieldrin concentration of 0.80 µg/kg dry weight in lake bottom sediments (see Table 6-36).

Table 6-36. Load Allocations for Dieldrin in Echo Park Lake

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	0.80

6.7.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 6-36 will be superseded, and the load allocations in Table 6-37 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 6-37, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Table 6-37. Alternative Load Allocations for Dieldrin in Echo Park Lake if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Lake Surface	City of Los Angeles	Lake bottom sediments	1.90

6.7.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

6.7.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate dieldrin, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

6.7.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the dieldrin WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

The daily maximum allowable load in Echo Park Lake is calculated from the estimated 99th percentile flow to the Lake multiplied by the event mean concentration consistent with achieving the long-term loading targets, described above in the PCBs section. USGS Station 11102000, Mission Creek near Montebello, CA, was selected as a surrogate for flow determination for flow to the lake, as described in the PCBs section (Section 6.5.6.5).

The event mean concentration of sediment in stormwater (55.8 mg/L) was calculated from the estimated existing watershed sediment load of 1.32 tons/yr (Table 6-12) divided by the total storm flow volume reaching the lake (17.4 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (8.98 cfs) yields a daily maximum sediment load from stormwater of 1226 kg/d (1.35 tons/d). Applying the wasteload allocation concentration of 0.80 µg dieldrin per dry kg of sediment yields the stormwater daily maximum allowable load of 0.00098 g/d of dieldrin. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

6.7.6.6 Future Growth

The manufacture and use of dieldrin is currently banned. Therefore, no additional allowance is made for future growth in the dieldrin TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

6.8 TRASH IMPAIRMENT

6.8.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Echo Park Lake include REC1, REC2, WARM, and WILD. Descriptions of these uses are listed in Section 2 of this TMDL report. Trash can potentially impair the REC1, REC2, WARM, and WILD in a variety of ways, including causing toxicity to aquatic organisms, damaging habitat, impairing aesthetics, and impeding recreation.

6.8.2 Numeric Targets

The numeric target is derived from the narrative water quality objective in the Los Angeles Basin Plan (LARWQCB, 1994) for floating material:

> *“Waters shall not contain floating materials, including solids, liquids, foams, and scum, in concentrations that cause nuisance or adversely affect beneficial uses”;*

and for solid, suspended, or settleable materials:

> *“Waters shall not contain suspended or settleable material in concentrations that cause nuisance or adversely affect beneficial uses.”*

The numeric target for the Echo Park Lake Trash TMDL is 0 (zero) trash in or on the water and on the shoreline. Zero trash is defined as no allowable trash discharged into the waterbody of concern, shoreline, and channels. No information has been found to justify any value other than zero that would fully support the designated beneficial uses. Furthermore, court rulings have found that a numeric target of zero trash is legally valid (*City of Arcadia et al. v. Los Angeles Regional Water Quality Control Board et al. (2006) 135 Cal.App.4th 1392*). The numeric target was used to calculate the waste load allocations for point sources and load allocations for nonpoint sources, as described in the following sections of this report.

6.8.3 Summary of Monitoring Data

The existing beneficial uses are impaired by the accumulation of suspended and settled debris. Common items observed include plastic pieces, paper items, Styrofoam, food waste, glass pieces, aluminum foil, and cigarette butts.

According to California's 2006 303(d) Impaired Waterbodies list, trash is causing water quality problems in Echo Park Lake. USEPA and Regional Water Quality Control Board staff confirmed the trash impairment during a site visit to Echo Park Lake on March 9, 2009. Staff conducted quantitative trash assessments and documented the trash impairment with photographs. Trash was observed in the lake, along the shorelines, and at the outlet of storm drains discharging into the lake.

Two quantitative trash assessments were conducted according to the Rapid Trash Assessment protocol which gives each shoreline a numeric score out of a possible 120 points (SWAMP, 2007). Higher scores correspond to cleaner areas, with 120 points representing a clean area. The severity of the trash problem was scored based upon the condition of the following parameters: level of trash, actual number of trash items found, threat to aquatic life, threat to human health, illegal dumping and littering, and accumulation of trash. Trash assessments were conducted within a 100 feet long by 10 feet wide area. The site visit evaluated different land use types surrounding Echo Park Lake, including recreational uses near a roadway and near picnic tables.

Echo Park has many visitors and is located in a densely populated urban area surrounded by busy streets. The lake is down a short steep slope from the streets which delineates the nonpoint source subwatershed boundary. Echo Park Lake has a shallow lotus bed on the northwest side, an inaccessible island on the northeast side, multiple small wetlands in the center, and a large fountain. The Park includes picnic tables near the lake, a playground on the northern shore, paddle boats for rent along the eastern shore, a fence along the southern corner, and a paved path around the entire lake, used for jogging and walking. Uncovered trash cans are located along the park path approximately every 100 feet, potentially leading to the transport of trash by wildlife or wind. Staff also observed approximately 300 birds in this small lake resulting in excessive bird droppings. Scum and small floatable pieces of trash were observed to accumulate in corners of the lake with stagnant water (Figure 6-11 and Figure 6-12).

Figure 6-11. Trash Accumulation in the Lotus Bed Section of Echo Park Lake

Figure 6-12. Floating Debris Observed on December 2, 2009

6.8.3.1 Picnic Area

A 100 ft. trash assessment was conducted near the playground and picnic tables on the northern shore of the lake. This area scored a 95/120. Only small trash items were observed. Trash was likely transported due to people littering in the picnic area and along the path. Some items were found in the water but no accumulation of trash was observed.

6.8.3.2 Near Glendale Boulevard

A trash assessment, conducted on the western shore near Glendale Boulevard, scored a 95/120. Trash was likely transported from the road and people littering along the park path.

6.8.3.3 Wildlife Feeding

Dumping of food waste, such as piles of rice or whole loaves of bread, to feed the birds was observed. Human food is unhealthy to wildlife and the massive quantities discarded cause an overabundance of birds to inhabit this area. An unnaturally large bird population leads to greater excrement quantities, which can worsen the nutrient problem in the lake.

Locations of the quantitative monitoring sites are shown in the map below (Figure 6-13).

Figure 6-13. Quantitative Trash Assessment Locations at Echo Park Lake

During a follow-up visit to Echo Park Lake on August 4, 2009, trash was similarly observed in the lake and on the shore. No quantitative surveys were conducted.

In summary, trash was present in and along the shore of Echo Park Lake during all visits. The prevalence of trash was evenly distributed around the lake. The main trash problems were caused by feeding wildlife and small trash items, such as cigarette butts.

6.8.4 Source Assessment

The major source of trash in Echo Park Lake results from litter, which is intentionally or accidentally discarded to the lake and watershed. Potential sources are categorized as point and nonpoint sources, depending on the transport mechanisms. For example:

1. Storm drains: trash deposited throughout the watershed and carried to various sections of the lake during and after rainstorms via storm drains. This is a point source.
2. Wind action: trash blown into the lake directly. This is a nonpoint source.
3. Direct disposal: direct dumping or littering into the lake. This is a nonpoint source.

Since the Echo Park Lake watershed includes residential areas, open space, parks, roads, and storm drains, both point and nonpoint sources contribute trash to the lake.

6.8.4.1 Point Sources

Based on reports from similar watersheds, the amount and type of trash transported is a function of the surrounding land use. The city of Long Beach recorded trash quantity collected at the mouth of the Los Angeles River; the results suggest total trash amount is linearly correlated with precipitation (Figure 6-14,

R^2=0.90, Signal Hill, 2006). A similar study found that the amount of gross pollutants entering the stormwater system is rainfall dependent but does not necessarily depend on the source (Walker and Wong, 1999). The amount of trash entering the stormwater system depends on the energy available to re-mobilize and transport deposited gross pollutants on street surfaces, rather than the amount of available gross pollutants deposited on street surfaces. Where gross pollutants exist, a clear relationship is established between the gross pollutant load in the stormwater system and the magnitude of the storm event. The limiting mechanism affecting the transport of gross pollutants, in the majority of cases, appears to be re-mobilization and transport processes (i.e., stormwater rates and velocities).

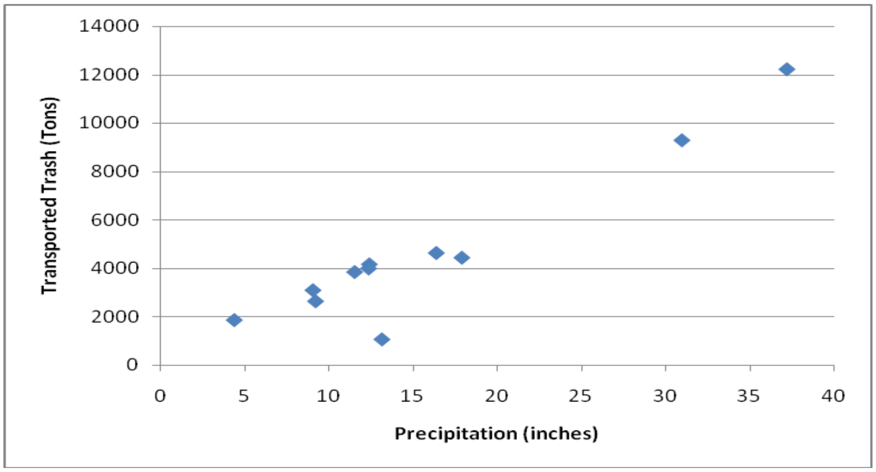

Figure 6-14. Storm Debris Collection Summary for Long Beach (Signal Hill, 2006)

In order to estimate trash generation rates, data from a comparable watershed was analyzed.

The city of Calabasas completed a study on a Continuous Deflective Separation (CDS) unit installed to catch runoff from Calabasas Park Hills to Las Virgenes. The CDS unit is a hydrodynamic separator that uses vortex settling to remove sediment, trap debris and trash, and separate floatables such as oil and grease. It is assumed that this CDS unit prevented all trash from passing through. The calculated area drained by this CDS Unit is approximately 12.8 square miles. Regional Board staff estimated the waterbody's urbanized area to be 0.10 square miles. The results of this clean-out, which represents approximately half of the 1998-1999 rainy season, were 2,000 gallons of sludgy water and a 64-gallon bag two-third full of plastic food wrappers. Part of the trash accumulated in this CDS unit for over half of the rainy season is assumed to have decomposed due to the absence of paper products. Since the CDS unit was cleaned out after slightly more than nine months of use, it was assumed that this 0.10 square mile urbanized area produced a volume of 64 gallons of trash. Therefore, 640 gallons of trash were generated per square mile per year. This estimate is used to determine trash loads.

During the 1998/1999 and 1999/2000 rain seasons, a Litter Management Pilot Study (LMPS) was conducted by Caltrans to evaluate the effectiveness of several litter management practices in reducing litter discharged from Caltrans storm water conveyance systems. The LMPS employed four field study sites, each of which was measured with the amount of trash produced when separate BMPs were applied. The average total load for each site normalized by the total area of control catchments was 6,677 gallons/mi^2/year. Other trash generation rates and studies exist but the LMPS study is the most applicable to Echo Park Lake because of similar land use, population density, and average daily traffic conditions. Therefore, this analysis will use 6,677 gal/mi^2/yr as the baseline estimate of trash for Caltrans roads.

Table 6-38 shows the current estimated volume of trash deposited within each of the responsible jurisdictions, in gallons per year, assuming a trash generation rate of 6,677 gallons of uncompressed trash/mi^2/year for Caltrans and a trash generation rate of 640 gallons of uncompressed trash per square mile per year for other jurisdictions. For responsible jurisdictions that are only partially located in the watershed, the square mileage indicated is for the portion in the watershed only. The current loads need to be reduced 100 percent to meet the TMDL target of zero trash.

Table 6-38. Echo Park Lake Estimated Point Source Trash Loads

Responsible Jurisdictions	Point Source Area (mi^2)	Current Point Source Trash Load (gal/year)
CA DOT (Caltrans)	0.022	150
City of Los Angeles	1.2	750

Note:
For Caltrans: Current Point Source Trash Load (gal/yr) = Point Source Area (mi2) * 6,677 (gal/ mi2/yr).
For all other jurisdictions: Current Point Source Trash Load (gal/yr) = Point Source Area (mi2) * 640 (gal/ mi2/yr)

6.8.4.2 Nonpoint Sources

Nonpoint source pollution is a source of trash in Echo Park Lake. Trash deposited in the lake from nonpoint sources is a function of transport via wind, wildlife, and overland flow and direct dumping.

Few studies have evaluated the relationship between wind strength and movement of trash from land surfaces to a waterbody. Lighter trash with a sufficient surface area to be blown in the wind, such as plastic bags, beverage containers, and paper or plastic food containers, are easily lifted and carried to waterbodies. Also, overland flow carries trash from the shoreline to waterbodies. Transportation of pollutants from one location to another is determined by the energy of both wind and overland stormwater flow.

Existing trash surrounding the lake is the fundamental cause of nonpoint source trash loading. Land use directly surrounding Echo Park Lake includes recreational areas. Visitors may intentionally or accidentally discard trash to grass or trails in the park, which initiate the journey of trash to waterbodies via wind or overland water flow. Varying uses of the park are responsible for different degrees of trash impairment. For example, areas with picnic tables generate more trash than parking lots. Visitation rates are also likely linked to the amount of trash from nonpoint sources.

Table 6-39 summarizes the nonpoint source area and current estimate of nonpoint source trash loads for responsible jurisdictions (see Figure 6-6 for an illustration of the park area surrounding the lake), assuming a trash generation rate of 640 gallons of uncompressed trash per square mile per year. The current loads need to be reduced 100 percent to meet the TMDL target of zero trash.

Table 6-39. Echo Park Lake Estimated Nonpoint Source Trash Loads

Responsible Jurisdictions	Nonpoint Source Area (Mile^2)	Current Nonpoint Source Trash Loads (Gal/year)
City of Los Angeles	0.024	16

Note: Current Nonpoint Source Trash Load (gal/yr) = Nonpoint Source Area (mi^2) * 640 (gal/ mi^2/yr)

6.8.5 Linkage Analysis

These TMDLs are based on numeric targets derived from narrative water quality objectives in the Los Angeles Basin Plan (LARWQCB, 1994) for floating materials and solid, suspended, or settleable materials. The narrative objectives state that waters shall not contain these materials in concentrations that cause nuisance or adversely affect beneficial uses. Since any amount of trash impairs beneficial uses, the loading capacity of Echo Park Lake is set to zero allowable trash.

6.8.6 TMDL Summary

Both point sources and nonpoint sources are identified as sources of trash in Echo Park Lake. For point sources, water quality standards are attained by assigning waste load allocations (WLAs) to Permittees of the Los Angeles County Municipal Separate Storm Sewer System (MS4) Permit and Caltrans (hereinafter referred to as responsible jurisdictions); these WLAs will be implemented through permit requirements. For nonpoint sources, water quality standards are attained by assigning load allocations (LAs) to municipalities and agencies having jurisdictions over Echo Park Lake and its subwatershed. These LAs may be implemented through regulatory mechanisms that implement the State Board's 2004 Nonpoint Source Policy such as conditional waivers, waste discharge requirements, or prohibitions.

The TMDL of zero trash requires that current loads are reduced by 100 percent. Final WLAs and LAs are zero trash (Table 6-40).

Table 6-40. Echo Park Lake Trash WLAs and LAs

Echo Park Lake	Allocation
Trash WLA	0
Trash LA	0

6.8.6.1 Wasteload Allocations

The geographical boundary contributing to point sources is defined by watershed areas which contain conveyances discharging to the waterbodies of concern. Conveyances include, but are not limited to, natural and channelized tributaries, and stormwater drains and conveyances. Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs).

Wasteload allocations are set to 0 (zero) allowable trash.

The permits affected are:

- County of Los Angeles (includes all cities in Los Angeles County except Long Beach): Board Order 01-182 (as amended by Board Orders R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No 97-03-DWQ, CAS000001

6.8.6.2 Load Allocations

Nonpoint source areas refer to locations where trash may be carried by overland flow, wildlife, or wind to waterbodies. Due to the transportation mechanism by wind, wildlife, and overland flow to relocate trash from land to waterbodies, the nonpoint source area may be smaller than the watershed. In addition, trash loadings frequently occur immediately around or directly into the lake making the load allocation a significant source of trash. According to the study by the city of Calabasas, the trash generation rate is 640 gallons per square mile per year from nonpoint sources areas (including, but not limited to, schools, commercial areas, residential areas, public services, road, and open space and parks areas). Current trash rates were calculated in the nonpoint source section.

Load allocations (LAs) for nonpoint sources are zero trash. Zero is defined as no allowable trash found in and on the lake, and along the shoreline. According to the Porter-Cologne Act, load allocations may be addressed by the conditional Waivers of WDRs, or WDRs. Responsible jurisdictions should monitor the trash quantity deposited in the vicinities of the waterbodies of concern as well as on the waterbody to comply with the load allocation.

The area adjacent to Echo Park Lake or defined as nonpoint sources includes parking lots, recreational areas, picnic areas, and walking paths. Assuming that trash within a reasonable distance from Echo Park Lake has a high potential to reach the waterbody, the nonpoint source jurisdiction is the city of Los Angeles. All load allocations are set to zero allowable trash.

6.8.6.3 Margin of Safety

A margin of safety (MOS) accounts for uncertainties in the TMDL analysis. The MOS can be expressed as an explicit mass load, or included implicitly in the WLAs and LAs that are allocated. Because this TMDL sets WLAs and LAs as zero trash, the TMDL includes an implicit MOS. Therefore, an explicit MOS is not necessary.

6.8.6.4 Critical Conditions/Seasonality

Critical conditions for Echo Park Lake are based on three conditions that correlate with loading conditions:

- Major storms
- Wind advisories issued by the National Weather Service
- High visitation – On weekends and holidays from May 15 to October 15.

Critical conditions do not affect wasteload or load allocations because zero trash is a conservative target. However, implementation efforts should be heightened during critical conditions in order to ensure that no trash enters the waterbody.

6.8.6.5 Future Growth

If any sources, currently assigned load allocations, are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality based effluent limitations pursuant to 40 CFR 122.44(d)(1).

6.9 Implementation Recommendations

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA’s recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006).* Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake;

reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize includes bacteria, nutrients, trash, metals and sediment. Reducing sediment loads would reduce OC pesticides and PCBs delivery to the lake in many instances. More information about this prioritization tool is available at: labmpmethod.org.

If necessary, these TMDLs may be revised as the result of new information (See Section 6.10 Monitoring Recommendations).

6.9.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 6-7, Table 6-16, Table 6-26, Table 6-36, and Table 6-40 for nutrients, PCBs, chlordane, dieldrin, and trash, respectively.

6.9.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to MS4 and Caltrans Stormwater permits as well as supplemental water additions. Wasteload allocations are expressed in Table 6-6, **Error! Reference source not found.**, Table 6-14, Table 6-24,
Table 6-34, and Table 6-40 for individual and grouped nutrients, PCBs, chlordane, dieldrin, and trash, respectively. The concentration and mass-based wasteload allocations will be incorporated into the Caltrans and Los Angeles County MS4 permits.

6.9.3 Source Control Alternatives

Echo Park Lake has nutrient-related, chlordane, dieldrin, PCB, and trash impairments. There are some management strategies that would address multiple impairments (i.e., sediment removal BMPs in the watershed) while other pollutants require a more specific management plan. The City of Los Angeles Department of Recreation and Parks submitted a recommendation to develop the Echo Park Lake Rehabilitation plan to the Proposition O program funds in 2006 (CDM, 2006), developed the concept plan and presented it to the Prop O Citizens Oversight Committee for bond funding approval. BMP and restoration efforts associated with this plan are now underway and will impact several of the Echo Park Lake impairments and initial modeling predicts that TMDL targets will be met upon its full implementation. An explicit goal of this project is to provide multiple environmental benefits by also enhancing open water, wetland, and nesting island habitat for native migratory waterfowl, turtles and gamefish.

The objective of the Plan is to improve water quality in both Echo Park Lake and the Los Angeles River watershed. Funds were allocated to general tasks including: site investigation and preliminary studies, engineering design tasks, permitting costs, construction of structural improvements to the lake and storm drain system, implementation of water quality BMPs, habitat restoration, educational efforts regarding water quality improvements, and post-construction monitoring. Due to the wide range of components, the

Plan is divided into four phases: pre-design, design, construction, and post-construction. Major lake improvements are summarized below; however, additional improvements are discussed in the Plan.

In-lake improvements, as part of the construction phase, will begin with draining the lake and removing contaminated soils. Fishes will also be removed. Once contaminated soils are properly disposed of offsite, an impermeable liner will be placed on the lake's bottom to eliminate infiltration, thus conserving the potable water used to supplement water levels in the lake. Structural BMPs to the lake's infrastructure will include the installation of trash capture and pollution control devices at the city's storm drain inlets. Sedimentation basins at all storm drains will be designed as stilling basins to enhance sedimentation and additional biological filters will trap pollutants, trash, and debris before stormwater flows into the lake. In-lake habitat and vegetation improvements will include lotus bed reconditioning as well as enhancement of the wetland and the lake's edge. Finally, the Plan details specific BMPs to be implemented throughout the surrounding park area, including grass swales, infiltration strips, porous pavement, "smart" irrigation systems, and educational signage.

Proposition O improvements to Echo Park Lake will assist with achieving local and regional water quality goals, including load reductions specific to the impairments addressed within these TMDLs. While there are some management strategies that would address multiple impairments (i.e., sediment removal BMPs in the watershed), their differences warrant separate implementation and monitoring discussions.

6.9.3.1 Nutrient-Related Impairments

The Echo Park Lake Rehabilitation Plan identified a number of BMPs that may help prevent degradation of this waterbody due to nutrient loading associated with future land use changes. Several of the recommended BMPs would function as sediment removal devices, which may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. The sediment removal BMPs proposed in the plan include:

- Hydrodynamic sediment and trash removal units within the city's concrete stormdrain structure or at the forebay of the lake
- Sediment removal device at the county stormdrain outfall
- Sediment basins at stormdrain outfall locations

The plan also proposes BMPs that provide that provide filtration, infiltration, and vegetative uptake and these removal processes may reduce nutrient loads. These BMPs include:

- Lotus bed reconditioning
- Submerging of existing floating wetland islands
- Lake edge vegetation
- Grassy swales/infiltration strips
- Porous pavement
- New "smart" irrigation system

The rehabilitation plan also proposes educational signage and kiosks regarding the above improvements. In addition to these efforts, education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive resident bird populations. The NNE BATHTUB model indicated Additional Parkland Loading is present in Echo Park Lake. This lake is heavily frequented by bird feeders and the additional bird feces produced by bird feeding contributes to this load; loads linked to trash and associated food scraps would also be reduced.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

6.9.3.2 Organochlorine Pesticides and PCB Impairments

The manufacture and use of chlordane, dieldrin, and PCBs are currently banned in the U.S. except for certain limited uses of PCBs authorized by USEPA. Therefore, no additional allowances for future growth are needed in the TMDLs. Source control BMPs and pollutant removal are the most suitable courses of action to reduce OC pesticides and PCBs in Echo Park Lake. The TMDL calculations performed for each pollutant (described above in their individual sections) indicated internal lake storage as the greatest contributing source and driving factor affecting fish tissue concentrations. Additionally, the watershed loads for chlordane and PCBs are less than one percent of the total loading that would be required to maintain the current sediment concentrations in the lake under steady-state conditions. Therefore, the most effective remedial actions and/or implementation efforts will focus on addressing the internal lake storage, such as capping or removal of contaminated lake sediments. As described above in Section 6.9.3, the Echo Park Lake Rehabilitation Plan proposes the draining of the lake, removal of sediments, and placement of an impermeable layer to address any residual contaminated soil.

A thorough remedial design study should be conducted prior to implementing removal of lake sediments and impermeable layer placement for Echo Park Lake. When properly conducted, removal of contaminated lake sediments, or dredging, can be an effective remediation option. The object of sediment dredging is to eliminate the pollutants that have accumulated in sediments at the lake bottom. Dredging is optimal in waterbodies with known spatial distribution of contamination because sediment removal can focus on problem areas. However, no spatial pattern of pollutant contamination was apparent in Echo Park Lake. Removal of the contaminated sediments reduces the pollutants available to the in-lake cycling by discontinuing exposure to benthic organisms, water column loading, and consequent bioaccumulation in higher trophic level fish. Potential negative effects of dredging include increased turbidity and lowered dissolved oxygen concentrations in the short term, and disturbance to the benthic community and reactivation of buried sediment and any associated pollutants. These negative impacts could be avoided through a plan that combines thorough removal of sediments and placement of an impermeable layer or cap.

In some cases, sediment capping may be appropriate to sequester contaminated sediments below an uncontaminated layer of sediment, clay, gravel, or media material. Capping is effective in restricting the mobility of OC pesticides and PCBs; however, it is most useful in deep lakes and capping alone may not be a viable solution at Echo Park Lake. Capping of in-place sediments without removal should be restricted to areas with sediments that can support the weight of a capped layer, and to areas where hydrologic conditions of the waterbody will not disturb the cap. The combination of sediment removal and capping of any residuals could be an effective solution if properly designed.

The in-lake options for remediation are costly, but would be the only way to achieve full use support in a short timeframe. It is, however, also true that the OC pesticides and PCBs in question are no longer manufactured and will tend to decline in concentration due to dilution by clean sediment and natural attenuation. Natural attenuation includes the chemical, biological, and physical processes that degrade compounds, or remove them from lake sediments in contact with the food chain, and reduce the concentrations and bioavailability of contaminants. These processes occur naturally within the environment and do not require additional remediation efforts; however, the half-lives of OC pesticides and PCBs in the environment are long, and natural attenuation often requires decades before observing significant improvement.

Loading from the watershed can also be expected to decline over time due to natural attenuation. While reductions are called for in watershed loads, these loads are a small fraction of the historic loads already stored in the lakes. Limited sampling has not identified any hotspots of elevated loading under current conditions. It may, however, be necessary to further investigate potential sources of OC pesticide and PCBs loading in the watershed, such as active and abandoned industrial sites, waste disposal areas, former chemical storage areas, and other potential hotspots, if sediment concentration is found to be elevated after the planned dredging project.

6.9.3.3 Trash Impairment

WLA may be complied with via full capture systems, partial capture systems, nonstructural BMPs, or any other lawful method which meet the target of zero trash. USEPA recommends the installation of full capture systems throughout the watershed. The Linear Radial, Inclined Screen, Baffle Box, and Catch Basin Insert are examples of full capture systems that fulfill the criteria of capturing all trash greater than 5 mm during flows less than the 1-year 1-hour storm. The Linear Radial utilizes a casing with louvers to serve as screens or mesh screen. Flows are routed through the louvers and into a vault. The Inclined Screen uses a wedge-wire screen with the slotting perpendicular or parallel to the direction of flow. This device is configured with an influent trough to allow solids to settle. The Baffle Box applies a two-chamber concept: the first chamber utilizes an underflow weir to trap floatable solids, and the second chamber uses a bar rack to capture material. The catch basin has an opening cover screen which is a coarse mesh screen at street level that is paired with a catch basin insert, a 5 mm screen inside the catch basin which filters out smaller trash. USEPA recommends implementation plans be consistent with the Los Angeles River trash TMDL. A monitoring plan should be developed in order to understand the effectiveness of the implementation efforts.

Similar devices to those described above were proposed in the Echo Park Lake Rehabilitation Plan. The plan proposes the installation of hydrodynamic units (either Continuous Deflective Separation (CDS) or Vortechnics units) which are estimated to capture 100 percent of floatables as well as provide sediment, nutrient, and other pollutant removal. These devices would be installed in the city’s concrete stormdrain structure or at the forebay of the lake, adjacent to the inlet structure. The Prop O recirculation system will also assist in removal of small pieces of trash.

LA may be complied with through the implementation of nonstructural BMPs or any other lawful methods which meet the target of zero trash. A minimum frequency of trash collection and assessment should be established at an interval that prevents trash from accumulating in deleterious amounts in between collections. Trash should be prevented by providing effective public education about littering impacts. Signs dissuading littering and wildlife feeding along roadways and around the lake are recommended. A city ban, tax, or incentive program reducing single-use plastic bags, Styrofoam containers, and other commonly discarded items which cannot decompose is recommended (Los Angeles County Department of Public Works, 2007).

Echo Park’s grounds and facilities are maintained by the city of Los Angeles. Trash is currently collected and removed from the park every other day during typical conditions and daily during windy or rainy weather. USEPA recommends continuation and expansion of the current trash pickups by the city of Los Angeles, including the collection of small trash items, such as cigarette butts.

The city of Los Angeles is also responsible for collection of trash in the lake. Currently a boat is used to remove large trash items from the lake. USEPA recommends a more frequent in-lake trash removal schedule to prevent the accumulation of small trash pieces.

The prevention and removal of trash in Echo Park Lake will lead to enhanced aesthetics, improved water quality, and the protection of habitat.

6.10 Monitoring Recommendations

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations result in compliance with the chlorophyll *a*, fish tissue, and trash targets a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be: 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies, fish tissue organochlorine compounds concentrations and trash levels..

6.10.1 Nutrient Related Impairments

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. Wasteload allocations are assigned to stormwater inputs and supplemental water additions. These sources should be measured near the point where they enter the lakes twice a year for at minimum: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient-response analysis for Echo Park Lake indicates that existing levels of nitrogen and phosphorus loading are resulting in attainment of the summer average chlorophyll *a* target concentration of 20 µg/L. As an antidegradation measure, nitrogen and phosphorus TMDLs are allocated based on existing loading. As an example of concentrations that responsible jurisdictions may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 6-5), the target concentrations of total phosphorus and total nitrogen that may be 0.58 mg-P/L and 4.5 mg-N/L for the Caltrans areas, and 0.71 mg-P/L and 4.5 mg-N/L for the city of Los Angeles areas. Targeted concentrations in the supplemental water additions may be 0.12 mg-P/L and 1.13 mg-N/L assuming volumes remain at existing levels. Assuming average precipitation depths, the targeted concentration of nitrogen in precipitation may be 0.204 mg-N/L. The flows associated with the additional parkland sources are unknown, so LA concentrations cannot be estimated. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

6.10.2 Organochlorine Pesticides and PCB Impairments

To assess compliance with the organochlorine compounds TMDLs, monitoring should include monitoring of fish tissue at least every three years as well as once yearly sediment and water column sampling. For the OC pesticides and PCBs TMDLs a demonstration that fish tissue targets have been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350 mm in length. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total suspended sediments, total PCBs, total chlordane and dieldrin; as well as the following in-lake sediment parameters: total organic carbon, total PCBs, total chlordane, and dieldrin. Environmentally relevant detection limits should be used (i.e., detection limits lower than applicable target), if available at a commercial laboratory. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Wasteload allocations are assigned to stormwater inputs and supplemental water additions. These sources should be measured near the point where they enter the lakes once a year during a wet weather event. Sampling should be designed to collect sufficient volumes of suspended solids to allow for the analysis of at minimum: total organic carbon, total suspended solids, total PCBs, total chlordane, and dieldrin. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken.

WLAs and LAs for each pollutant are assigned to the sediment-associated load from the watershed as well as the lake bottom sediments. The concentration-based WLAs and LAs are 2.10 µg/kg dry weight for total chlordane, 1.77 µg/kg dry weight for total PCBs, and 0.80 µg/kg dry weight for dieldrin. The associated reductions from the watershed load needed to meet the WLAs are 74.7 percent for total chlordane, and 92.7 percent for total PCBs. A quantitative percent reduction cannot be calculated for dieldrin because all sediment samples are below detection limits (which are greater than the TMDL target concentration); however, the needed reduction appears to be on the order of 39 percent.

6.10.3 Trash Impairments

Responsible jurisdictions should monitor the trash quantity deposited in the vicinity of Echo Park Lake as well as on the waterbody to comply with the load allocation and to understand the effectiveness of various implementation efforts. Quarterly monitoring using the Rapid Trash Assessment Method is recommended. The trash TMDL target is zero trash; a 100 percent reduction is required.

7 Lake Calabasas TMDLs

Lake Calabasas (#CAL4052100019990203084034) is listed as impaired by ammonia, DDT (originally on the consent decree, but not on the current 303(d) list), eutrophication, odor, organic enrichment/low dissolved oxygen, and pH (SWRCB, 2010). This section of the TMDL report describes the impairments and the TMDLs developed to address them. Nutrient load reductions are required to achieve the chlorophyll *a* target; these reductions are also expected to alleviate pH, odor, DO and ammonia problems.

7.1 Environmental Setting

Lake Calabasas is a private lake located in the Los Angeles River Basin (HUC 18070105) in the city of Calabasas (Figure 7-1). The Urban Lakes Study (UC Riverside, 1994) reported that the lake was constructed in 1968. The area occupied by the lake was excavated to bedrock, a layer of soil was added, and then a plastic liner was put down and covered with soil along with cement in some areas. The lake is surrounded by dense residential development (Figure 7-2) and owned by the Calabasas Park Homeowners Association. This 17.8-acre lake (surface area based on Southern California Association of Governments (SCAG) 2005 land use data) does not discharge to surface waters but rather loses water via evaporation (UC Riverside, 1994). During storm events water discharges to the storm drain system. With a volume of 71.2 acre-feet, the average depth is approximately 4 feet (depth provided by the city of Calabasas; volume is calculated from this depth and the land use-based surface area). Recreation includes paddle boating and limited fishing (catch and release fishing is mandated by the Calabasas Park Homeowner's Association). Bird feeding may be another recreational activity at Lake Calabasas; however, it has not been observed during recent fieldwork. Residents are not allowed to swim in the lake. Figure 7-3 shows a view of Lake Calabasas facing the southwest. There are approximately 25 aerators in the lake (Figure 7-4). Lake managers use algaecides (including Cutrine Plus and copper sulfate) to control algal growth in the lake on an as-needed basis. Additional characteristics of the watershed are summarized below.

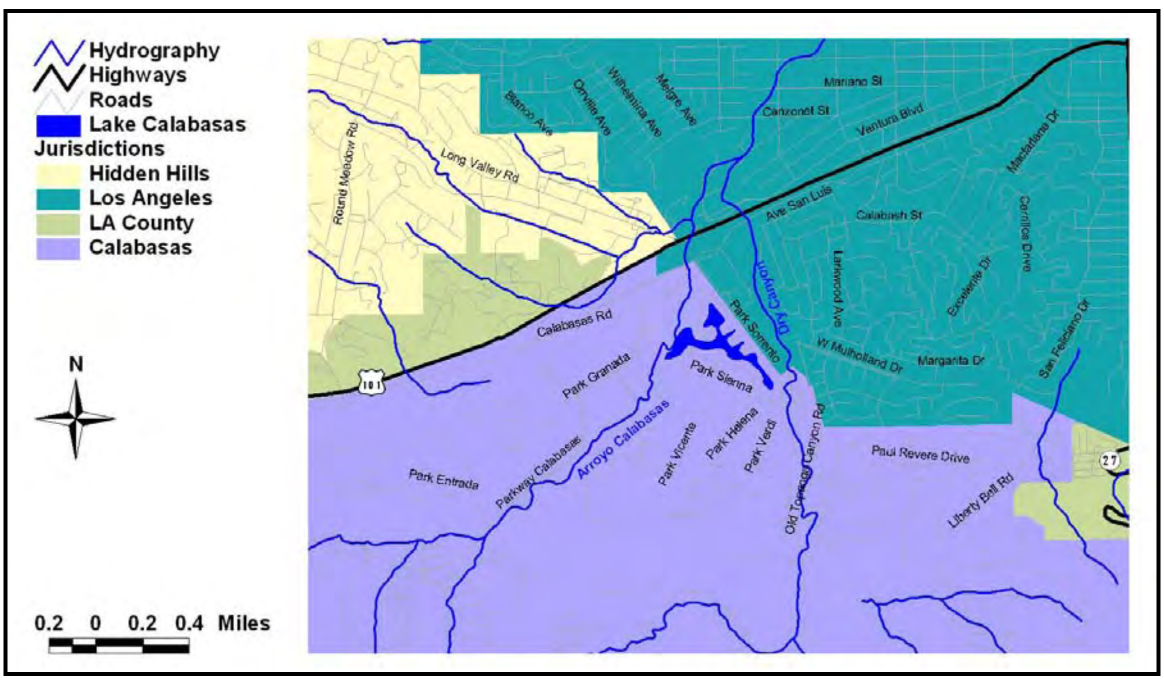

Figure 7-1. Location of Lake Calabasas

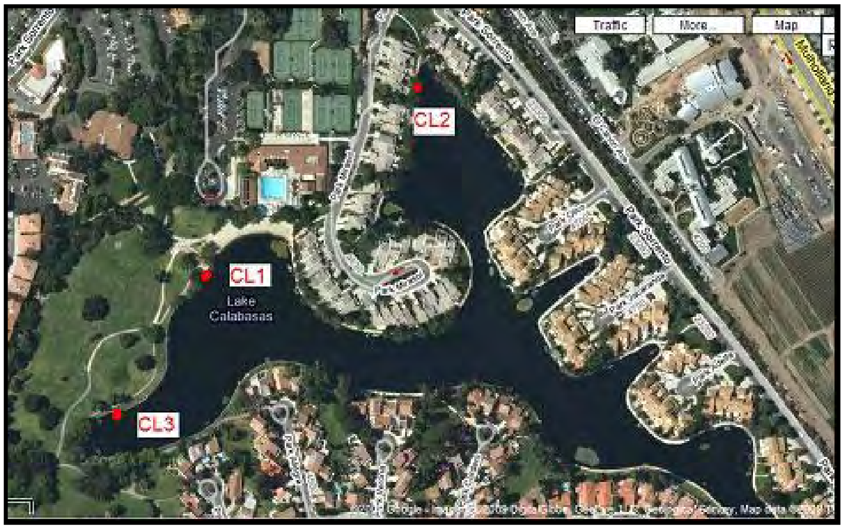

Figure 7-2. Satellite Imagery of Lake Calabasas

Figure 7-3. Lake Calabasas (facing southwest)

Note: multiple aerators are in the lake (several are visible in this picture)

Figure 7-4. Lake Calabasas Aerators

7.1.1 Elevation, Storm Drain Networks, and Subwatershed Boundaries

The Lake Calabasas watershed is 86.5 acres and ranges in elevation from 287 meters to 398 meters. Due to the small scale of this watershed, the boundary was manually delineated based on aerial photography, digital elevation data, and the county of Los Angeles storm drain coverage (Figure 7-5). Because many small storm drains discharge into the lake, all allocations for the TMDLs will be wasteload allocations except load allocations for atmospheric deposition. Figure 7-6 shows one of the storm drains capturing flow from the surrounding watershed. As shown in Figure 7-5, multiple storm drains contribute directly to the lake.

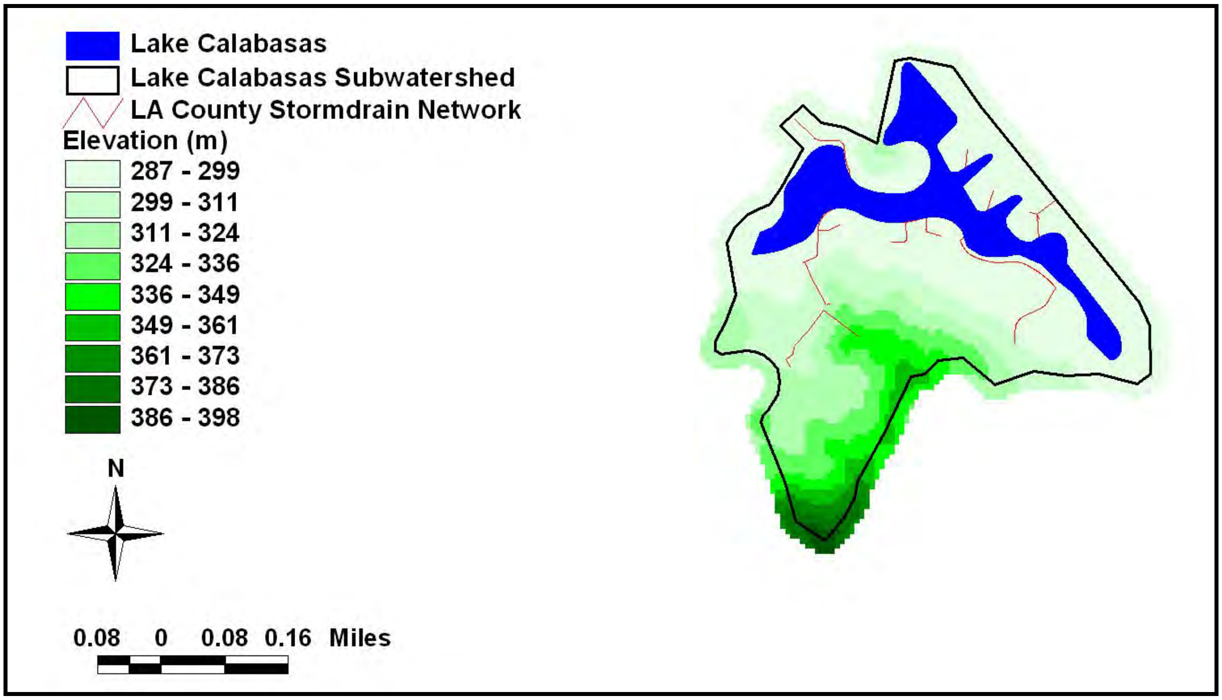

Figure 7-5. Elevation, Storm Drain Network, and the TMDL Subwatershed Boundary for Lake Calabasas

Note: many small storm drains capture flow from surrounding areas into the lake.

Figure 7-6. Lake Calabasas Storm Drain

7.1.2 MS4 Permittees

Figure 7-7 shows the MS4 stormwater permittee in the Lake Calabasas watershed. The entire subwatershed is comprised of the city of Calabasas. The storm drain coverage was provided by the county of Los Angeles.

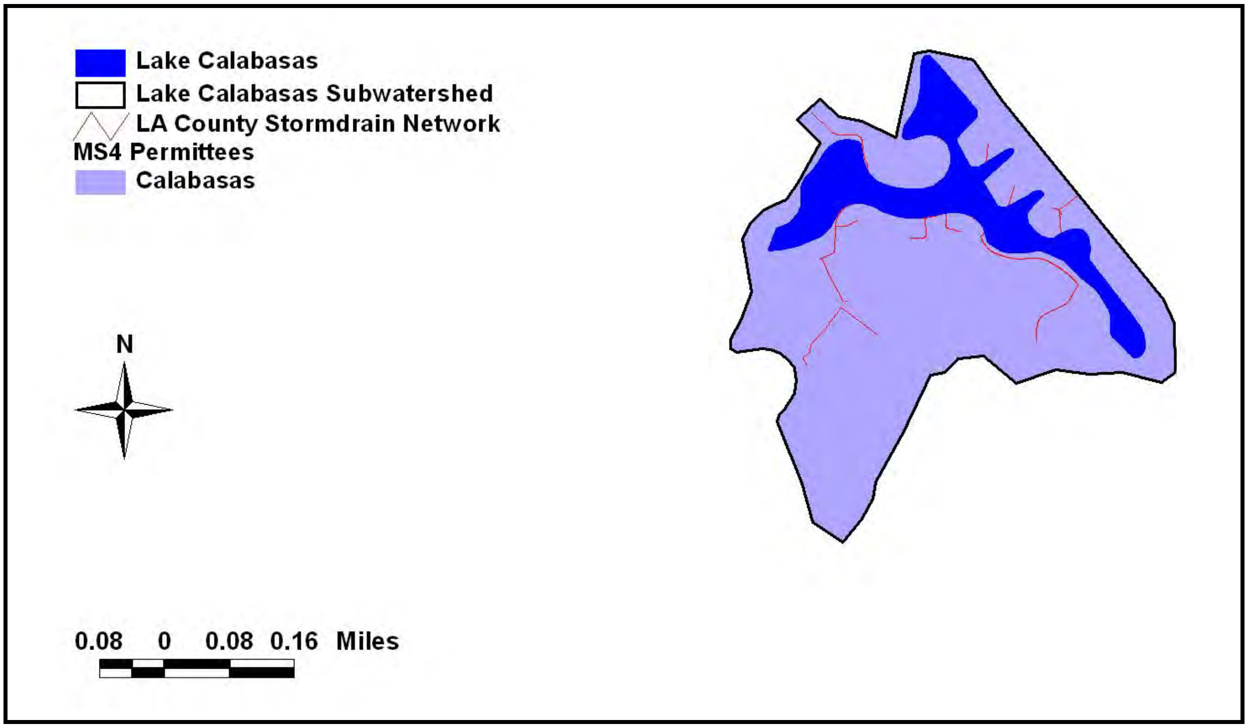

Figure 7-7. MS4 Permittee and the Storm Drain Network in the Lake Calabasas Subwatershed

7.1.3 Non-MS4 NPDES Dischargers

As of the writing of these TMDLs, there are no additional (non-MS4) NPDES permitted discharges in the Lake Calabasas watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

7.1.4 Land Uses and Soil Types

The analysis for this watershed includes estimates of existing watershed loading obtained from the Los Angeles River Basin LSPC Model, discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the Los Angeles River Basin LSPC model are shown in Figure 7-8. The watershed is comprised of residential development and open space. Table 7-1 summarizes the land use areas draining to Lake Calabasas.

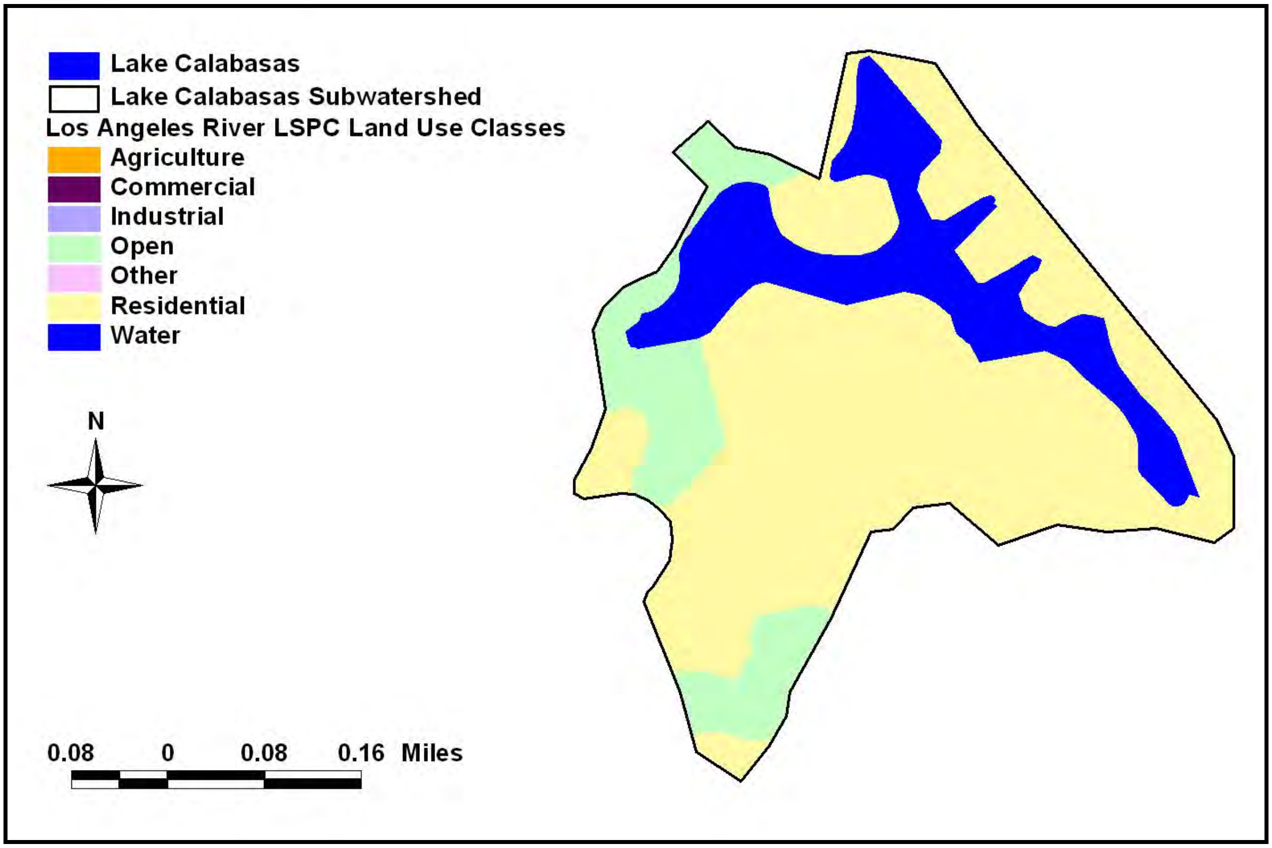

Figure 7-8. LSPC Land Use Classes for the Lake Calabasas Subwatershed

Table 7-1. Land Use Areas (ac) Draining to Lake Calabasas

Land Use	City of Calabasas
Agriculture	0
Commercial	0
Industrial	0
Open	14.2
Other Urban	0.0
Residential	72.3
Total	**86.5**

There are no Resource Conservation and Recovery Act (RCRA) contaminated industrial facilities located near the Lake Calabasas watershed. Figure 7-9 shows the predominant soils identified by STATSGO (Appendix D, Wet Weather Loading) in the Lake Calabasas subwatershed. The soil type identified as MUKEY 660489 is Urban Land-Lithic Xerorthents-Hambright-Castaic, a hydrologic group D soil, which has high runoff potential, very low infiltration rates, and consists chiefly of clay soils. Soil MUKEY 660473 is Urban Land-Sorrento-Hanford, a hydrologic group B soil, which has moderate infiltration rates and moderately coarse textures. The representative soil group for each LSPC modeling subbasin was based on the dominant soil type present in the subbasin. For the modeling subbasin that contains the Lake Calabasas watershed, the predominant soil type was type D. Additionally, the watershed around Lake Calabasas rests on alluvium and the Monterey Formation. The Monterey Formation is a petroleum source rock which can produce high concentrations of nutrients, organic carbon, trace metals, selenium and high sulfate salts (USGS, 2002).

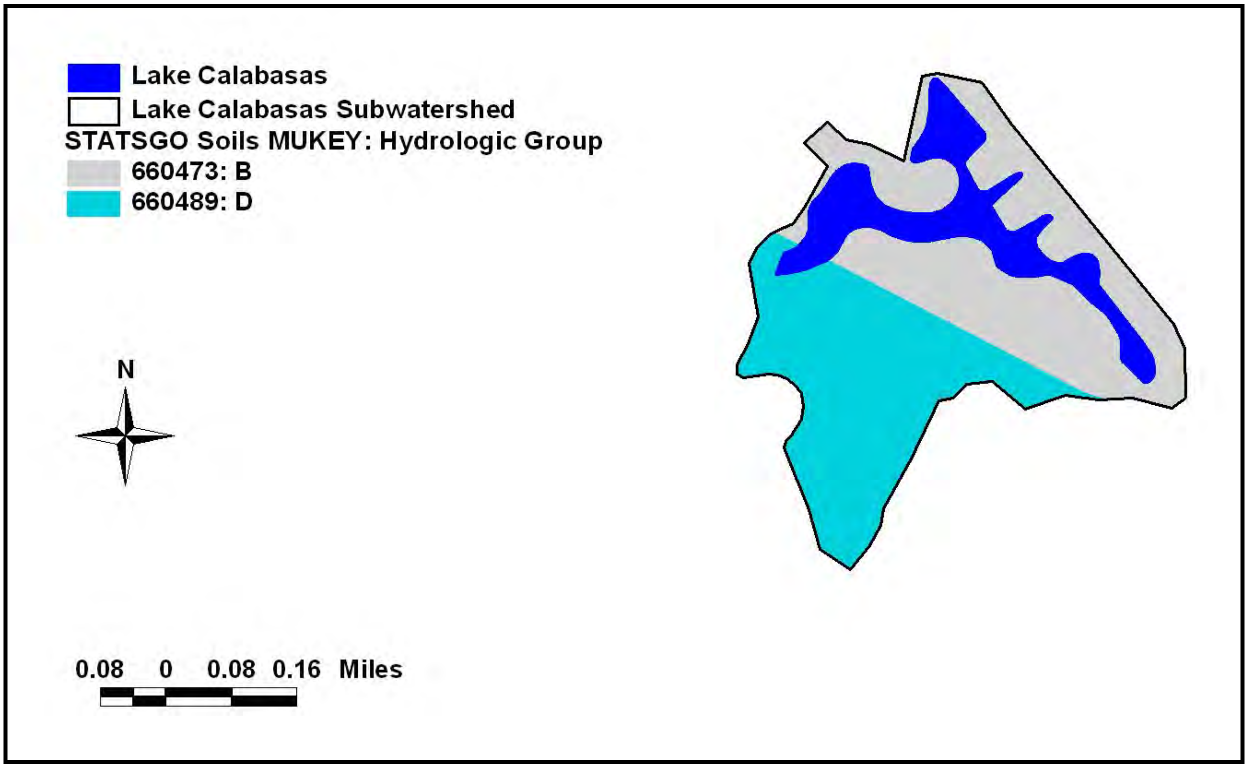

Figure 7-9. STATSGO Soil Types Present in the Lake Calabasas Subwatershed

7.1.5 Additional Inputs

According to the 1994 Urban Lakes Study (UC Riverside, 1994), the primary sources of water to Lake Calabasas are potable water from the Las Virgenes Municipal Water District and stormwater from the surrounding housing development. These water sources were confirmed during recent fieldwork performed by USEPA.

7.2 NUTRIENT-RELATED IMPAIRMENTS

A number of the assessed impairments for Lake Calabasas are associated with nutrients and eutrophication. Nutrient-related impairments for Lake Calabasas include ammonia, eutrophication, odor, organic enrichment/low dissolved oxygen, and pH (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algal photosynthesis removes carbon dioxide from the water, which can lead to elevated pH in poorly buffered systems. Respiration during nighttime hours may cause decreased dissolved oxygen (DO) concentrations. Algal blooms may also contribute to odor problems.

7.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Lake Calabasas was not identified specifically in the Basin Plan; therefore, the beneficial uses associated with the downstream segment (Arroyo Calabasas) apply: REC1, REC2, WARM, WILD, and MUN (personal communication, Regional Board, February 24, 2010). Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, and WARM uses by stimulating algal growth that may form mats that impede recreational and drinking water uses, alter pH and dissolved oxygen (DO) levels and alter biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD and MUN uses could become impaired.

7.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Lake Calabasas. The following targets apply to the ammonia, eutrophication, odor, organic enrichment/low dissolved oxygen, and pH impairments (see Section 2 for additional details and Table 7-2 for a summary):

- The Basin Plan expresses ammonia targets as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life. In order to assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for Lake Calabasas in these TMDLs, a median temperature of 21.8 ºC and a 95th percentile pH of 9.4 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 0.78 mg-N/L, the four-day average is 0.46 mg-N/L, and the 30-day average (chronic) target is 0.19 mg-N/L (Note: the median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target varies with the values determined during sample collection.).
- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient concentrations (e.g., nitrogen and phosphorous) in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Lake Calabasas; however, as described in Tetra Tech (2006), summer (May to September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and is also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target must be met at one-half the Secchi depth during the summer (May – September) and annual averaging periods.
- The Basin Plan states that "waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses."
- The Basin Plan states "at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges." Shallow, well-mixed lakes, such as Lake Calabasas, must meet the DO target in the water column from the surface to 0.3 meters above the bottom of the lake.

- The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." Shallow, well-mixed lakes, such as Lake Calabasas, must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake.

Nitrogen and phosphorus target concentrations are based on simulation of allowable loads with the NNE BATHTUB model (see Section 7.2.5). Based on the calibrated model for Lake Calabasas, the target nutrient concentrations within the lake are

- 0.66 mg-N/L summer average (May – September) and annual average
- 0.066 mg-P/L summer average (May – September) and annual average

Table 7-2. Nutrient-Related Numeric Targets for Lake Calabasas

Parameter	Numeric Target	Notes
Ammonia[1]	0.78 mg-N/L acute (one-hour) 0.46 mg-N/L four-day average 0.19 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.66 mg-N/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model
Total Phosphorous	0.066 mg-P/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model

[1] The median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan..

7.2.3 Summary of Monitoring Data

This section briefly summarizes the nutrient-related monitoring data for Lake Calabasas. Appendix G (Monitoring Data) contains more detailed information regarding water quality sampling in the lake.

Lake Calabasas was monitored from the southwestern lobe of the lake as part of the Urban Lakes Study (UC Riverside, 1994). Total Kjeldahl nitrogen (ammonia plus organic nitrogen; TKN) ranged from 1.0 mg-N/L to 1.8 mg-N/L with two samples less than the detection limit (0.01 mg-N/L). Ammonium concentrations were usually less than or equivalent to the detection limit (0.01 mg-N/L) although four samples collected in February and March 1993 ranged from 0.3 mg-N/L to 0.5 mg-N/L (less than the acute target assuming the analysis methodology converted all ammonia to ammonium). All of the nitrite and nitrate samples were less than the detection limit (0.01 mg-N/L) except one nitrate sample of 0.1 mg/L. Five of 28 phosphate samples measured 0.1 mg-P/L; the others were less than the detection limit (0.01 mg-P/L). Total phosphorus concentrations ranged from 0.1 mg-P/L to 0.2 mg-P/L with seven samples less than detection (0.01 mg-P/L). pH in the lake ranged from 8.3 to 9.3 throughout the water column, and 78 percent of samples exceeded the allowable range. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 μg/L to 172 μg/L with an average of 39 μg/L, which is greater than the target summer average of 20 μg/L.

The 1996 Water Quality Assessment Report (LARWQCB, 1996) states that DO was partially supporting the aquatic life use and that 92 measurements of DO were collected which ranged from 0.2 mg/L to 15.7 mg/L. pH was partially supporting the aquatic life use and not supporting the secondary drinking water standards. pH was measured 85 times, and values ranged from 7.4 to 9.3. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonia samples were collected ranging from non-detect to 0.45 mg-N/L with an average of 0.06 mg-N/L. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Odor was listed as not supporting the contact and non-contact recreation uses. Eutrophication was not supporting the aquatic life use.

The city of Calabasas has been monitoring water quality in Lake Calabasas since 2004. Samples were collected from the surface waters. Nitrate concentrations have ranged from 0.04 mg-N/L to 1.6 mg-N/L; phosphate concentrations ranged from 0.03 mg-P/L to 0.77 mg-P/L. Secchi depths range from 0.5 m to greater than 2.7 m, and pH ranged from 7.91 to 9.69. Dissolved oxygen has been observed ranging from 4.8 mg/L to 15.82 mg/L with water temperatures ranging from 9.2 ºC to 32.7 ºC. Exceedances of the pH target were observed in approximately 77 percent of the measurements; DO exceedances were observed approximately 3 percent of the time.

The Regional Board sampled Lake Calabasas from two in-lake sites on August 6, 2009. Ammonia concentrations were less than or equal to 0.03 mg-N/L; TKN ranged from 1.17 mg-N/L to 1.23 mg-N/L. Nitrate and nitrite samples were less than the detection limit of 0.01 mg-N/L. Orthophosphate ranged from 0.0129 mg-P/L to 0.0453 mg-P/L and total phosphorus ranged from 0.152 mg-P/L to 0.221 mg-P/L. Chlorophyll *a* ranged from 35 µg/L to 81 µg/L. Secchi depth ranged from 0.66 m to 0.74 m. Profile data were also collected between 9:00 a.m. and 9:50 a.m. The temperature in the lake ranged from 25.6 °C to 26.4 °C. The DO ranged from 6.37 to 9.74 mg/L, and pH ranged from 7.98 to 9.30 over the assessment depth. Exceedances of the pH target occurred in 98 percent of the measurements taken during the profiles conducted on this day (excluding the measurements taken less then 0.3 m above the lake bottom).

Water quality data collected in Lake Calabasas indicate impairment due to elevated nutrient loads. Summer average chlorophyll *a* concentrations exceed the target concentration of 20 μg/L. The DO target has been met during recent sampling events, but historic data indicate that low DO may have been an issue for the lake. Currently, aerators appear to be controlling DO concentrations. No odors were observed during two recent sampling events by USEPA and/or Regional Board. pH measurements have exceeded the maximum allowable value (8.5) during recent and historic monitoring. There were no exceedances of the acute or chronic ammonia criteria during any recent sampling events with associated pH and temperature measurements. The nutrient TMDLs for Lake Calabasas presented in Section 7.2.6 account for summer season critical conditions by assessing loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 μg/L. These reductions in nutrient loading are expected to alleviate pH, odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication.

7.2.4 Source Assessment

The majority of nutrient loading to Lake Calabasas originates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading), including irrigation (5.3 percent of the total irrigation volume is assumed to reach the lake). The watershed is entirely within the city of Calabasas and contributes 97.7 percent of the total phosphorus load and 74.4 percent of the total nitrogen load. Loading due to direct deposition from the atmosphere is discussed in Appendix E (Atmospheric Deposition).

Table 7-3. Summary of Average Annual Flows and Nutrient Loading to Lake Calabasas

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
City of Calabasas	MS4 Stormwater[1]	69.3	129 (97.7)	769 (74.4)
Calabasas Park Homeowners Association	Supplemental Water Additions (Potable Water)	57.9	3.28 (0.03)	252 (24.4)
City of Calabasas	Parkland Irrigation	0.151	0.0085 (0.00)	0.655 (0.00)
	Atmospheric Deposition (to the lake surface) [2]	26.0	NA	12.4 (0.01)
Total		**153**	**132**	**1,034**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

7.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on Lake Calabasas, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus. In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Lake Calabasas, the following inputs apply: surface area of 17.8 acres, average depth of 4 ft, and volume of 71.2 ac-ft. Based on the phosphorus turnover ratio for this lake (Walker, 1987), the annual averaging period is appropriate (i.e., annual loads are input to the model rather than summer season loads).

The NNE BATHTUB Tool was calibrated to average summer season water quality data observed over twice the typical Secchi depth (2*1.1 m = 2.2 m). To predict the average observed total nitrogen concentration over this depth (1.47 mg-N/L), the calibration factor on the net nitrogen sedimentation rate was set to 1.5. The calibration factor on the net phosphorus sedimentation rate was set to the maximum suggested (2) (Walker, 1987) and the resulting concentration is 0.11 mg-P/L, slightly higher than the average observed 0.099 mg-P/L. Although this calibrated sedimentation rate reflects the net effects of phosphorus settling and resuspension, the high calibration factor indicates that settling is the more dominant mechanism in this system, and internal phosphorus loading is likely insignificant relative to the other sources of loading. The reductions in external phosphorus loading in the lake required by this TMDL should lead to further suppression of internal loading. To simulate the average observed chlorophyll *a* concentration, the calibration factor on concentration was set to 0.84 for a predicted concentration of 48.7 µg/L.

7.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Following calibration of the NNE BATHTUB Tool (Section 5.2.5), the allowable loading combinations of nitrogen and phosphorus were calculated using Visual Basic's GoalSeek function (Appendix A, Nutrient TMDL Development). The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 0.66 mg-N/L summer average (May – September) and annual average
- 0.066 mg-P/L summer average (May – September) and annual average

The loading capacities for total nitrogen and total phosphorus are 328 lb-N/yr and 55.1 lb-P/yr, respectively. These loading capacities can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load (divided among WLAs and LAs) is 28.6 percent of the existing load of 1,034 lb-N/yr, or 295 lb-N/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. WLAs and LAs are developed assuming equal percent load reductions in all sources. The resulting TMDL equation for total nitrogen is then:

328 lb-N/yr = 292 lb-N/yr + 3.54 lb-N/yr + 32.8 lb-N/yr

For total phosphorus, the allocatable load (divided among WLAs and LAs) is 37.7 percent of the existing load of 132 lb-P/yr, or 49.7 lb-P/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. The resulting TMDL equation for total phosphorous is then:

55.1 lb-P/yr = 49.7 lb-P/yr + 0 lb-P/yr + 5.51 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections. As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on simulation of allowable loads with the NNE BATHTUB model (see Section 7.2.5). These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

7.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs and alternative WLAs for total phosphorous and total nitrogen. The alternative WLAs will be effective and supersede the WLAs listed in Table 7-4 if the conditions described in Section 7.2.6.1.2 are met.

Under any of the wasteload allocation schemes responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on the Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

7.2.6.1.1 Wasteload Allocations

The Lake Calabasas watershed drains to a series of storm drains prior to discharging to the lake. Therefore, all loads associated with the surrounding drainage area are assigned WLAs (Note: the loading associated with irrigation is included in the City of Calabasas' WLA). The supplemental water source used to maintain lake levels discharges at a single point and is also assigned a WLA. The relevant permit number associated with the stormwater input is

- County of Los Angeles (including the city of Calabasas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001

Table 7-4 summarizes the existing nutrients loads and WLAs for these sources. Total phosphorus WLAs represent a 62.4 percent reduction in existing loading, and total nitrogen WLAs represent a 71.4 percent reduction in existing loading (Table 7-4). Each WLA must be met at the point of discharge.

Table 7-4. Wasteload Allocations of Phosphorus and Nitrogen Loading to Lake Calabasas

Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Wasteload Allocation Total Phosphorus[1] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Wasteload Allocation Total Nitrogen[1] (lb/yr)
City of Calabasas	MS4 Stormwater[2]	129	48.5	770	220
Calabasas Park Homeowners Association	Supplemental Water Additions	3.28	1.23	252	72.0
Total		**132**	**49.7**	**1,022**	**292**

[1] The wasteload allocation must be met at the point of discharge.

[2] This input includes effluent from storm drain systems during both wet and dry weather.

7.2.6.1.2 Alternative "Approved Lake Management Plan Wasteload Allocations"

Concentration-based WLAs not exceeding the concentrations listed in Table 7-5 are effective and supersede corresponding WLAs for a responsible jurisdiction in Table 7-4 if:

1. The responsible jurisdiction requests that concentration-based wasteload allocations not to exceed the concentrations established in Table 7-5 apply to it;
2. The responsible jurisdiction provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 7-2. Responsible jurisdictions may work together to develop, submit and implement the Lake Management Plan. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The responsible jurisdiction may use monitoring data and modeling to show that the water quality criteria, targets and requested WLAs will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based wasteload allocations for total nitrogen and total phosphorus. These wasteload allocations are not to exceed the concentrations in Table 7-5 as a summer average (May-September) and annual average; and,
4. USEPA does not object to the Regional Board's determination within sixty days of receiving notice of it.

The concentration-based WLAs must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 7-5. Alternative Wasteload Allocations of Phosphorus and Nitrogen in Lake Calabasas if an Approved Lake Management Plan Exists

Responsible Jurisdiction	Input	Maximum Allowable Wasteload Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Wasteload Allocation Total Nitrogen[1] (mg-N/L)
City of Calabasas	MS4 Stormwater[2]	0.1	1.0
Calabasas Park Homeowners Association	Supplemental Water Additions	0.1	1.0

[1] Each concentration-based wasteload allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

[2] This input includes effluent from storm drain systems during both wet and dry weather.

7.2.6.2 Load Allocations

Atmospheric deposition of nitrogen to the lake surface is a nonpoint source and is assigned a load allocation (LA). Table 7-6 lists the existing and allowable load (28.6 percent of the existing load) from this source. Atmospheric deposition does not contribute significant loads of phosphorus (Appendix E, Atmospheric Deposition). LAs are provided for each responsible jurisdiction and input. These loading values (in pounds per year) represent the TMDL load allocations (Table 7-6).

Table 7-6. Load Allocations of Nitrogen Loading to Lake Calabasas

Input	Existing Total Phosphorus Load (lb-P/yr)	Load Allocation Total Phosphorus (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Load Allocation Total Nitrogen (lb/yr)
Atmospheric Deposition (to the lake surface)*	0	0	12.4	3.54
Total	**0**	**0**	**12.4**	**3.54**

* Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

7.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. To account for the uncertainties concerning the relationship between nutrient loading and the resultant in-lake chlorophyll *a* an explicit MOS is included in these TMDLs. This explicit MOS is set at 10 percent of the loading capacity for total phosphorus and total nitrogen.

7.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are expected to alleviate any pH, odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication. These TMDLs therefore protect for critical conditions.

7.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of nutrient loading to Lake Calabasas occurs during wet weather events that deliver pollutant loads from the surrounding watershed, the daily maximum allowable loads of nitrogen and phosphorus are calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) to the Lake multiplied by the allowable concentrations consistent with achieving the long-term loading targets. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the Lake Calabasas watershed. USGS Station 11105500, Malibu Creek at Crater Camp near Calabasas, CA, was selected as a surrogate for flow determination. This gage is the closest USGS StreamStats gage. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for Malibu Creek (355 cfs) (Wolock, 2003). To estimate the peak flow to Lake Calabasas, the 99th percentile flow for Malibu Creek was scaled down by the ratio of drainage areas (86.5 acres/67,200 acres; Lake Calabasas watershed area/Malibu Creek watershed area at the gage). The resulting peak flow estimate for Lake Calabasas is 0.457 cfs.

The allowable concentrations for phosphorus and nitrogen were calculated from the annual allowable load (49.7 lb-P/yr and 295 lb-N/yr, respectively; sum of WLA and LA values) divided by the total annual volume delivered to the lake (127 ac-ft). Multiplying the average allowable concentrations (0.257 mg-P/L for phosphorous and 1.17 mg-N/L for nitrogen) by the 99th percentile peak daily flow (0.457 cfs) yields the daily maximum load. The daily maximum allowable loads of phosphorus and nitrogen for Lake Calabasas are 0.634 lb-P/d and 2.88 lb-N/day, respectively. These loads represent the maximum allowable daily load, which for Lake Calabasas, is entirely due to wet weather stormwater from city of Calabasas areas (supplemental water additions and irrigation are not needed during large storm events). For comparison, the existing phosphorus load (132 lb-P/yr) would yield an event mean concentration of 0.382 mg-P/L and a daily load of 0.942 lb-P/d. The existing nitrogen load (1,034 lb-N/yr) would yield an event mean concentration of 2.99 mg-N/L and a daily load of 7.38 lb-N/d. As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

7.2.6.6 Future Growth

The Lake Calabasas watershed is fully developed. No load allocation has been set aside for future growth, and it is unlikely that any dischargers will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

7.3 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that could reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize includes bacteria, nutrients, trash, metals and sediment. More information about this prioritization tool is available at: labmpmethod.org

If necessary, these TMDLs may be revised as the result of new information (See Section 7.4 Monitoring Recommendations).

7.3.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy, and the Conditional Waiver for Discharges from Irrigated Lands, adopted by the Los Angeles Regional Water Quality Control Board on November 3, 2005. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 7-6.

7.3.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to MS4 Stormwater permits as well as supplemental water additions (Table 7-4 for Standard and Table 7-5 for Alternative Allocations). The MS4 stormwater mass-based wasteload allocations will be incorporated into the Los Angeles County MS4 permit. Wasteload allocations for supplemental water additions will be implemented by the Regional Board.

7.3.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

To address nutrient-related impairments, source reduction and pollutant removal BMPs designed to reduce sediment loading should be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather stormwater also contributes nutrient loading to Lake Calabasas. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

The rules and regulations set forth by the Calabasas Park Homeowners Association regarding waterfowl, fertilization, pesticide application, and pets aim to reduce pollutant loading to the lake. Fertilizers and pesticides may be used on adjacent lake properties and properties that eventually drain to the lake. However, fertilizers and pesticides are prohibited from reaching the lake by any means per the rules and regulations. Education of homeowners and lake maintenance staff regarding the proper placement, timing, and rates of fertilizer and pesticide products will result in reduced pollutant loading. Citizens should be advised to follow product guidelines regarding product amounts and to spread products when the chance of heavy precipitation in the following days is low. Pet owners are required to properly dispose of pet wastes. Visitors to the lake (members, tenants, and guests) are prohibited from feeding birds or other animals. Following these rules will reduce nutrient loading associated with fecal material or fertilizers that may wash directly into the lake or into storm drains that eventually discharge to the lake.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

7.4 Monitoring Recommendations

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations will indeed result in compliance with the chlorophyll *a* target, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies.

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. At Lake Calabasas wasteload allocations are assigned to supplemental water additions. This source should be monitoring once a year during the summer months (critical conditions) for at minimum; ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids. Wasteload allocations are also assigned to stormwater inputs from the City of Calabasas. This source should be measured near the points where it enters the lakes twice a year for at minimum: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient TMDLs for Lake Calabasas conclude that a 62.4 percent reduction in total phosphorus loading and a 71.4 percent reduction in total nitrogen loading are needed to maintain a summer average chlorophyll *a* concentration of 20 µg/L. As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 7-3), the targeted concentrations may be 0.257 mg-P/L and 1.17 mg-N/L for the city of Calabasas. For the supplemental water additions, the targeted concentrations may be 0.0078 mg-P/L and 0.46 mg-N/L. Assuming average precipitation depths, the targeted concentration of nitrogen in precipitation may be 0.0569 mg-N/L. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

8 El Dorado Park Lakes TMDLs

The El Dorado Park lakes (#CAL4051501020000228153407) are located in the San Gabriel River Basin (HUC 18070106) in the city of Long Beach. The El Dorado Park lakes are listed as impaired by algae, ammonia, copper, eutrophication, lead, mercury (fish tissue), and pH (SWRCB, 2010). There are not sufficient data to calculate TMDLs for each lake individually, so TMDLs have been developed for each lake system; i.e., northern and southern lake systems. This section of the TMDL report describes the impairments and the TMDLs developed to address nutrients (Section 8.2) and mercury (Section 8.3). In the northern lake system, nutrient load reductions are required to achieve the chlorophyll *a* target and restore beneficial uses. Nutrient TMDLs are identified for the southern lake system based on existing conditions since nitrogen and phosphorus levels are achieving the chlorophyll *a* target level. The mercury TMDL identified for the southern lake system is also based on existing conditions since mercury levels are likely achieving the fish tissue target level. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that copper and lead are currently achieving numeric targets at El Dorado Park lakes; therefore, TMDLs are not included for these pollutants. Analyses are presented below for lead (Section 8.4) and for copper (Section 8.5).

8.1 Environmental Setting

The El Dorado Park lakes are a chain of six small lakes located within El Dorado Regional Park in the county of Los Angeles (Figure 8-1). The park was opened to the public in 1969. The northern four lakes (Coyote, Alamo, Large, and Horseshoe) are hydraulically connected and separate from the system comprised by the two southern lakes (Nature Center North and Nature Center South), which are hydraulically connected to each other. The 2006 303(d) GIS coverage shows only four of the six lakes in the system. There is an additional lake in each system, one at the downstream end of the northern chain (Horseshoe Lake) and one at the upstream end of the southern chain (Nature Center North Lake). The 2006 303(d) GIS coverage also shows an additional lake to the left of the San Gabriel River, which is located in the El Dorado Park Golf Course. The State Water Board concluded this lake was erroneously included in the GIS coverage and is removing it for the following reasons: 1) it is not hydraulically connected with the El Dorado Park lakes, 2) it is in another drainage area, and 3) it has not been sampled for water quality (personal communication, Nancy Kapellas, SWRCB to Thomas Siebels, RWQCB, February 4, 2009). This updated layer will be available from the SWRCB after finalization of the 2010 303(d) list.

The park borders the San Gabriel River for approximately two miles (Figure 8-2) and Coyote Creek for three-quarters of a mile. The lakes were created on what was formerly San Gabriel River floodplain but are not hydrologically connected to the river at this time. The northern four lakes have a cumulative surface area of 30.1 acres, and the southern two lakes have a combined surface area of 5.2 acres (surface areas based on Southern California Association of Governments [SCAG] 2005 land use). Figure 8-3 shows Coyote Lake, the northernmost lake in the park. When constructed, the depth in Nature Center South Lake was approximately 28 feet (personal communication, Ed Gahafer [park staff], USEPA field notes 2-26-09); however, the maximum depth measured by the USEPA Region 9 laboratory staff, on February 26th, 2009 was less than ten feet (USEPA field notes 2-26-09). Restrooms on the park grounds are connected to the city sewer system. The lakes are periodically stocked (CDFG, 2009) and recreational fishing is allowed in the northern four lakes (the CDFG "Fishing in the City" program periodically holds events at the lakes). Paddle boating and radio controlled model boating occurs in Alamo Lake (Figure 8-4), but boating is prohibited in all other lakes. Visitors are not allowed to swim in the lakes. Bird feeding is another recreational activity at the lakes and some feeding has been observed during recent fieldwork. The Nature Center, located in the southern part of the park, conducts environmental education and receives more than 150,000 visitors a year. Lake managers use algaecides including

(Cutrine Plus and Reward) in some of the lakes on an as-needed basis. Additional characteristics of the watershed are summarized below.

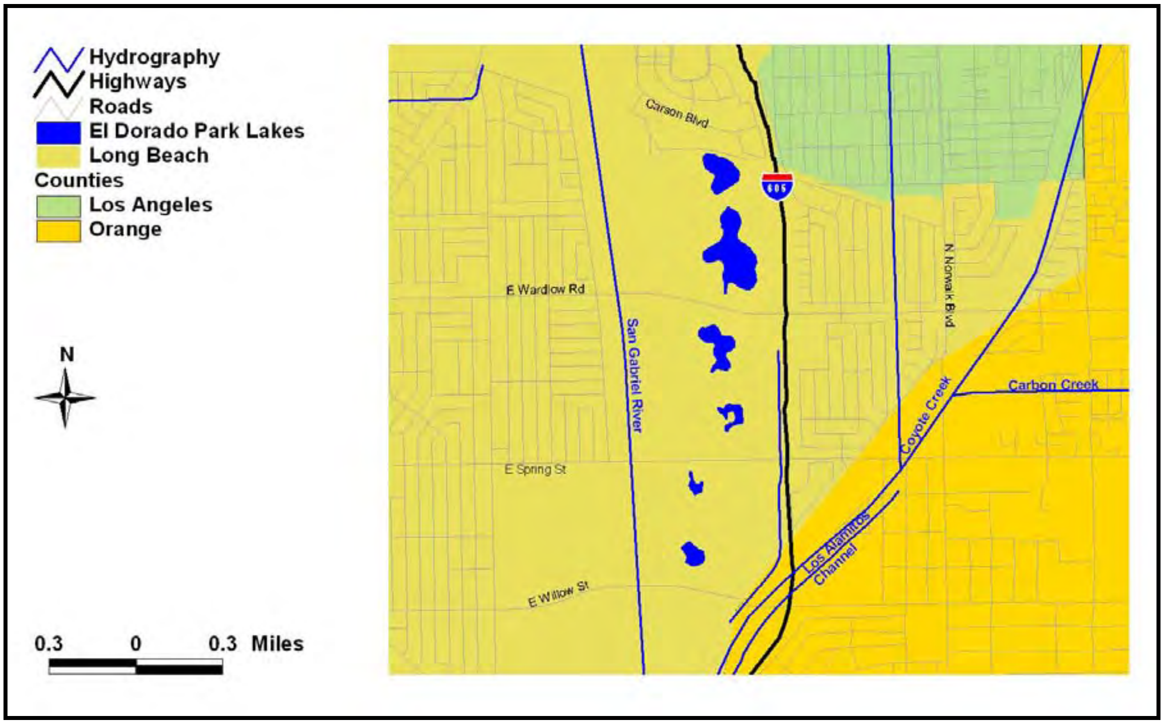

Figure 8-1. Location of El Dorado Park Lakes

Figure 8-2. San Gabriel River Adjacent to the El Dorado Regional Park

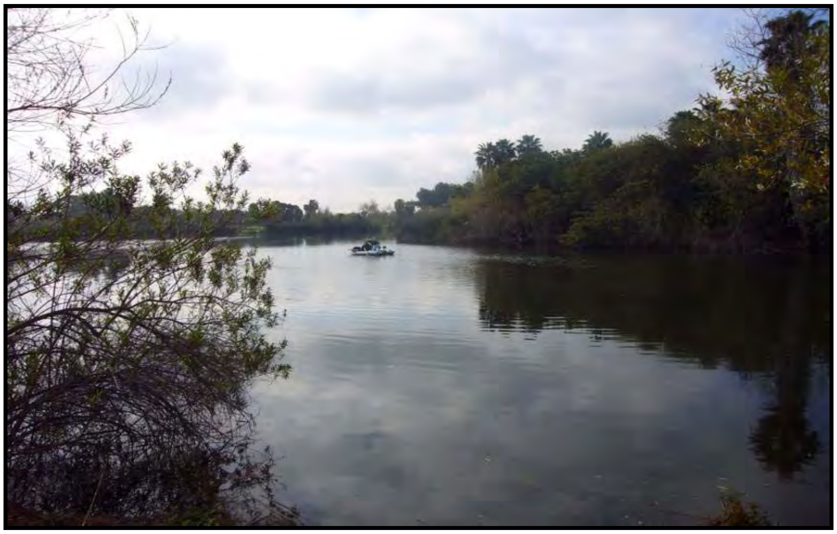

Figure 8-3. North Side of Coyote Lake

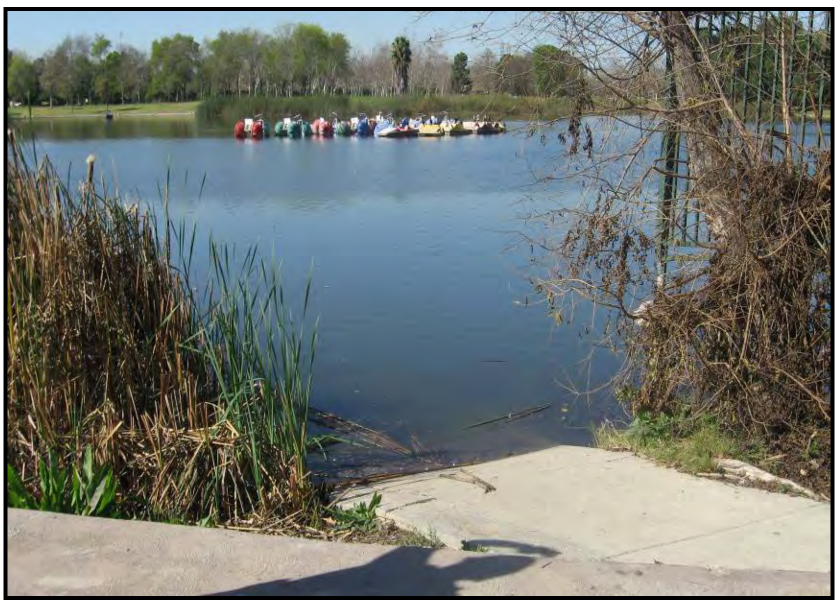

Figure 8-4. Paddle Boating at Alamo Lake

8.1.1 Elevation, Storm Drain Networks, and Subwatershed Boundaries

The El Dorado Park lakes have a 219-acre drainage area and are located in a low-elevation watershed (6.5 meters to 9.9 meters above sea level). The two TMDL subwatershed boundaries for the El Dorado Park lakes were based on watershed boundaries obtained from the county of Los Angeles, digital elevation data, aerial imagery, and the storm drain network provided by the county of Los Angeles (Figure 8-5). The subwatershed draining to the northern four lakes is comprised of 185 acres, and the subwatershed draining to the southern two lakes is comprised of 33.8 acres. Neither subwatershed contains an organized storm drain network nor a permitted point source, so all allocations for the surrounding watershed will be load allocations except wasteload allocations for the supplemental water additions; however, the lakes are actively pumped into the county of Los Angeles storm drain network during heavy rain events.

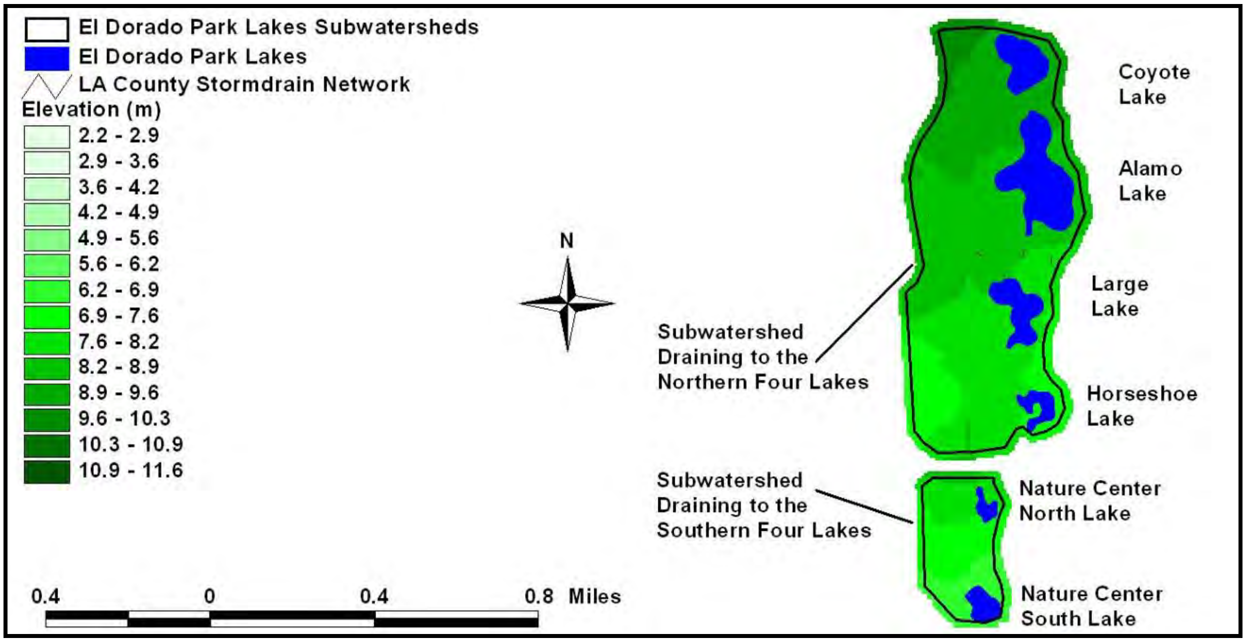

Figure 8-5. Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries for the El Dorado Park Lakes

8.1.2 MS4 Permittee

Figure 8-6 shows the MS4 stormwater permittee that comprises both the northern and southern subwatersheds of the El Dorado Park lakes as well as the county of Los Angeles storm drain network. Although both watersheds are in the city of Long Beach incorporated area, there are no major drains that divert runoff directly to any of the lakes. Loads from the parkland will be assigned load allocations because they do not drain to pipes or culverts prior to discharge to the lake.

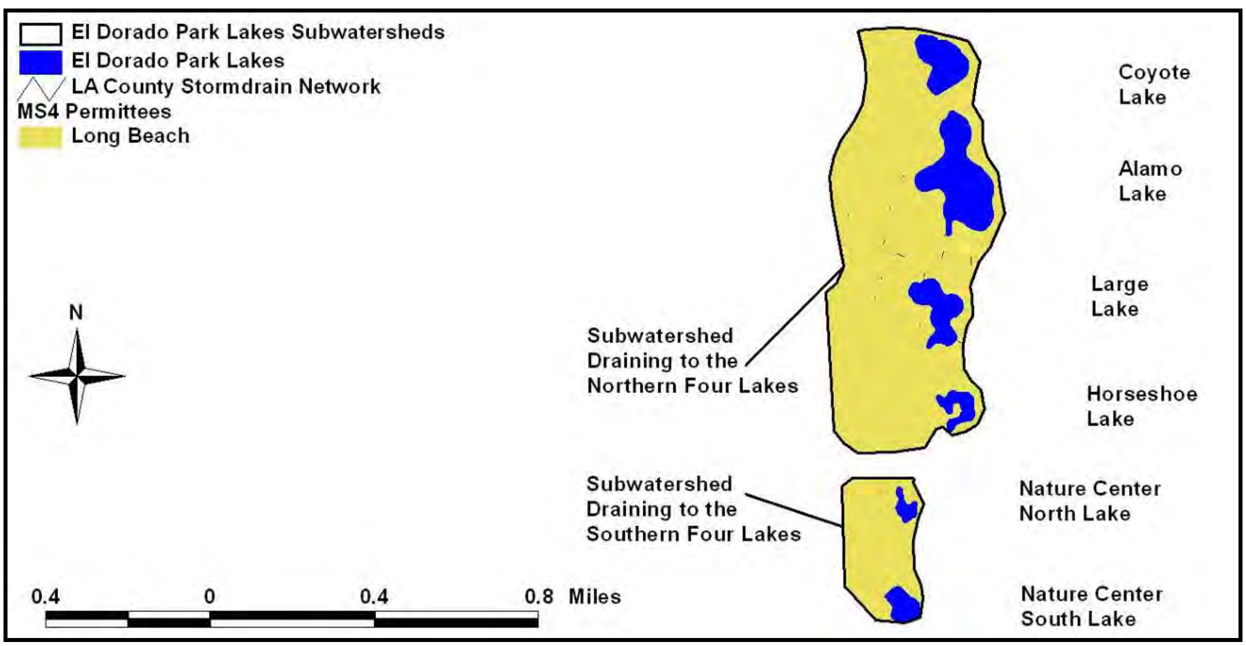

Figure 8-6. MS4 Permittee and the Storm Drain Network in the El Dorado Park Lakes Subwatersheds

8.1.3 Non-MS4 NPDES Dischargers

As of the writing of these TMDLs, there are no additional (non-MS4) NPDES permitted discharges in the El Dorado Park lakes watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

8.1.4 Land Uses and Soil Types

Several of the analyses for the El Dorado Park lakes watershed include source loading estimates obtained from the San Gabriel River Basin LSPC Model, discussed in Appendix D (Wet Weather Loading) of this TMDL report. Both subwatersheds are comprised of land classified by the San Gabriel River Basin LSPC model as "other urban or built-up" (based on SCAG 2000 land use data), except for the two polygons classified as water (Figure 8-7). Comparison of the LSPC land use coverage to SCAG 2005 data and recent satellite imagery indicate that the areas draining to the El Dorado Park lakes are parkland.

The LSPC land use data were also inaccurate with regard to lake surface area and omitted two of the six lakes in the park. To improve accuracy in land use areas, the SCAG 2005 database was used to estimate the area of the lakes in each subwatershed. All remaining areas were assumed parkland (185 acres in the northern subwatershed and 33.8 acres in the southern subwatershed).

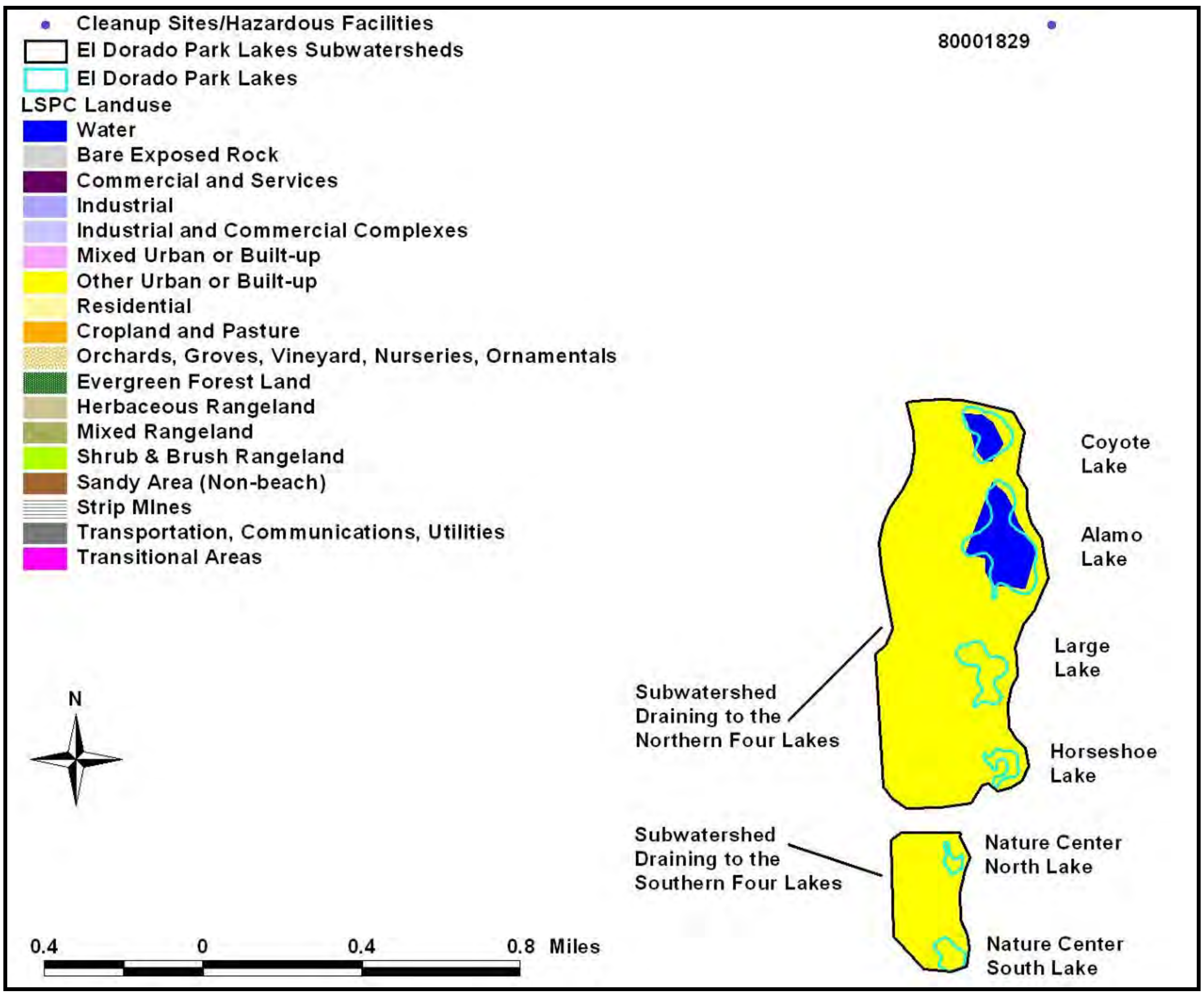

Figure 8-7. LSPC Land Use Classes for the El Dorado Park Lakes Subwatersheds

There is one Resource Conservation and Recovery Act (RCRA) contaminated industrial facility located within one mile of the El Dorado Park lakes. Available information for this facility (a liquid waste refiner) is summarized in Table 8-1. No additional information was readily available regarding potential contaminants of concern for this facility; however, the site does not drain to the El Dorado Park lakes.

Table 8-1. RCRA Cleanup Site near the El Dorado Park Lakes

Envirostor #	Facility Name	Cleanup Status
80001829 (CAT080011059)	Enviropur West Corporation	Assessed; not identified for corrective action

Figure 8-8 shows the predominant soils identified by STATSGO (Appendix D, Wet Weather Loading) in the El Dorado Park lakes subwatersheds. The soil type is identified as Urban land-Sorrento-Hanford (MUKEY 660473), a hydrologic group B soil, which has moderate infiltration rates when wet and consists chiefly of soils that have a moderately coarse texture.

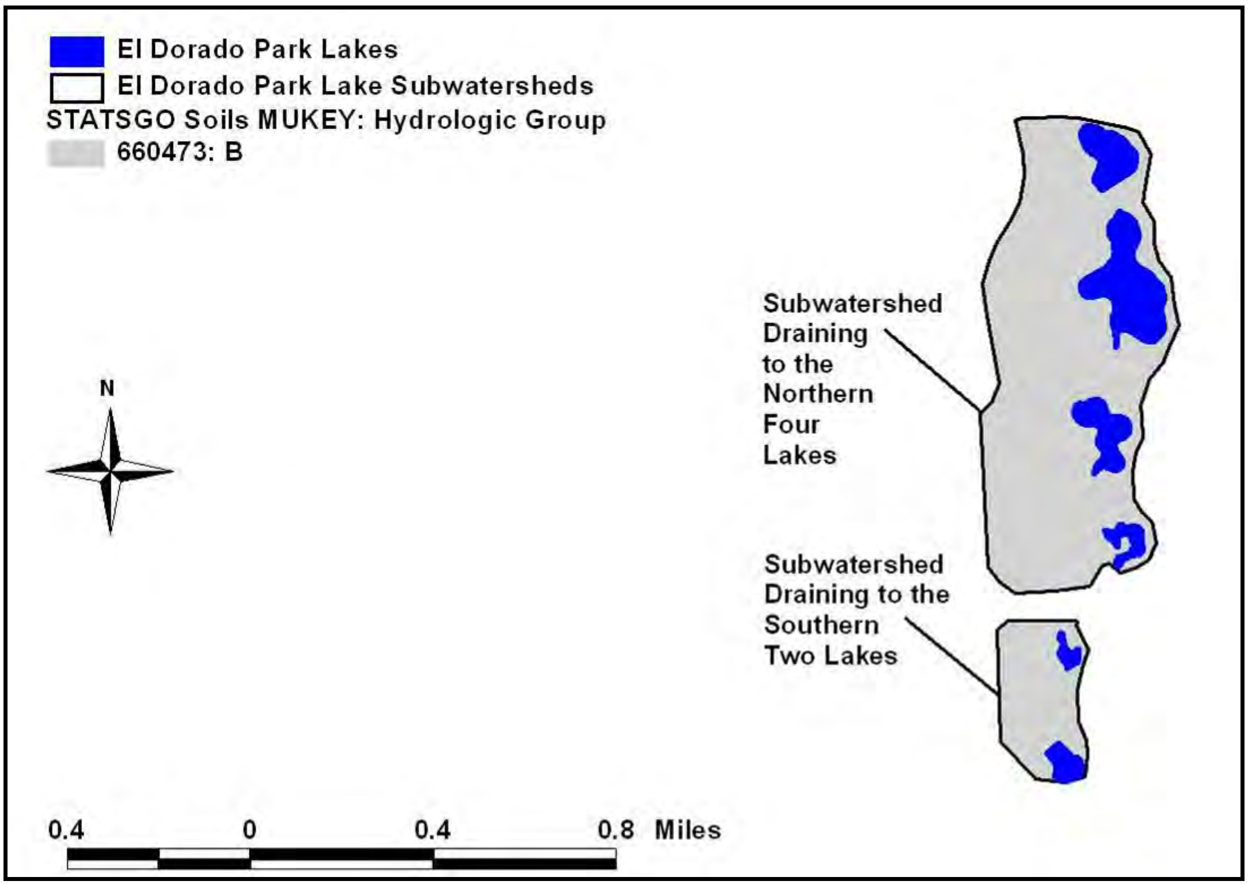

Figure 8-8. STATSGO Soil Types Present in the El Dorado Park Lakes Subwatersheds

8.1.5 Additional Inputs

The El Dorado Park lakes are comprised of two hydraulically separate systems. The northern four lakes receive supplemental water additions from a groundwater well that pumps into Coyote Lake (Figure 8-5) at a rate of approximately 110 ac-ft/yr. The southern lakes in El Dorado Park receive supplemental water from a potable water source. On average, 105 ac-ft are pumped annually into Nature Center North Lake

(Figure 8-5). Parklands surrounding both systems are irrigated with reclaimed water, some of which may reach the lakes. Irrigation water is applied to 221 acres surrounding Coyote and Alamo Lakes (known as Area III) and 179 acres surrounding Large and Horseshoe Lakes (known as Area II). At the Nature Center where the two southern lakes are located, 91.1 acres are irrigated. The applied average annual volumes to these respective areas (based on utility bills) are 244 ac-ft, 280 ac-ft, and 64.7 ac-ft; applied depths range from 8.5 inches to 18.8 inches (3.9 percent of the total irrigation volume is assumed to reach the lake). Loads resulting from these inputs are described in Appendix F (Dry Weather Loading).

8.2 NUTRIENT RELATED IMPAIRMENTS

A number of the assessed impairments for the El Dorado Park lakes are associated with nutrients and eutrophication. Nutrient-related impairments for the El Dorado Park lakes include algae, ammonia, eutrophication, and pH (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algal photosynthesis removes carbon dioxide from the water, which can lead to elevated pH in poorly buffered systems. Algal blooms may also contribute to odor problems.

8.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to the El Dorado Park lakes include REC1, REC2, WARM, WILD, MUN, and WET. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, and WARM uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels and biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD, MUN, and WET uses could become impaired.

8.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to the El Dorado Park lakes. The following targets apply to the algae, ammonia, eutrophication, and pH impairments (see Section 2 for additional details and Table 8-2 for a summary):

- The Basin Plan expresses ammonia targets as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life. In order to assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for the El Dorado Park lakes in these TMDLs, a median temperature of 16.2 ºC and a 95th percentile pH of 8.5 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 3.20 mg-N/L, the four-day average is 2.44 mg-N/L, and the 30-day average (chronic) target is 0.98 mg-N/L (Note: the median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target varies with the values determined during sample collection.).
- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient (e.g., nitrogen and phosphorous) concentrations in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not

contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in the El Dorado Park lakes; however, as described in Tetra Tech (2006), summer (May – September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and is also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target must be met at half of the Secchi depth during the summer (May – September) and annual averaging periods.

- The Basin Plan states "at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges." Shallow, well-mixed lakes, such as the El Dorado Park lakes systems, must meet the DO target in the water column from the surface to 0.3 meters above the bottom of each lake.
- The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." Shallow, well mixed lakes, such as the El Dorado Park lakes, must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake.

Nitrogen and phosphorus target concentrations are based on simulation of allowable loads with the NNE BATHTUB model (see Section 8.2.5). Based on the calibrated model for the northern four El Dorado Park lakes, the target nutrient concentrations within the lakes are

- 0.69 mg-N/L summer average (May – September) and annual average
- 0.069 mg-P/L summer average (May – September) and annual average

For the southern two El Dorado Park lakes, the target nutrient concentrations within the lakes are

- 1.15 mg-N/L summer average (May – September) and annual average
- 0.115 mg-P/L summer average (May – September) and annual average

Table 8-2. Nutrient-Related Numeric Targets for the El Dorado Park Lakes

Parameter	Numeric Target	Notes
Ammonia[1]	3.20mg-N/L acute (one-hour) 2.44 mg-N/L four-day average 0.98 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider

Parameter	Numeric Target	Notes
	natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	Northern Lake System: 0.69 mg-N/L summer average (May – September) and annual average Southern Lake System: 1.15 mg-N/L summer average (May – September) and annual average	Northern Lake System: Based on simulation of allowable loads from the NNE BATHTUB model Southern Lake System: Conservatively based on existing conditions, which are maintaining chlorophyll *a* levels below the target of 20 µg/L
Total Phosphorous	Northern Lake System: 0.069 mg-P/L summer average (May – September) and annual average Southern Lake System: 0.115 mg-P/L summer average (May – September) and annual average	Northern Lake System: Based on simulation of allowable loads from the NNE BATHTUB model Southern Lake System: Based on an in-lake TN to TP ratio of 10, typical of natural systems

[1] The median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan..

8.2.3 Summary of Monitoring Data

Water quality monitoring has been conducted in the El Dorado Park lakes since the early 1990s. This section summarizes the monitoring data relevant to the nutrient impairments. Additional details regarding monitoring are discussed in Appendix G (Monitoring Data).

The El Dorado Park lakes were included in the 1992/1993 sampling effort to support the Urban Lakes Study (UC Riverside, 1994). Data were collected from the north end of Alamo Lake. Total Kjeldahl Nitrogen (TKN) concentrations ranged from 1.2 mg-N/L to 4.2 mg-N/L. Nineteen of 45 samples for ammonium were less than the detection limit (0.01 mg-N/L); ammonium concentrations as high as 1.9 mg-N/L were observed and therefore exceeded the acute ammonia target of 0.98 mg-N/L. Nitrite samples were consistently less than the detection limit (0.01 mg-N/L), as were the majority of nitrate concentrations. Measurable amounts of nitrate were only observed in January and February of 1993 when concentrations ranged from 0.1 mg-N/L to 0.3 mg-N/L. Orthophosphate concentrations ranged from 0.2 mg-P/L to 0.9 mg-P/L, and total phosphorus concentrations generally ranged from 0.3 mg-P/L to 0.5 mg-P/L though two samples near the lake bottom were 0.6 mg-P/L and 1.1 mg-P/L. pH ranged from 8.2 to 9.4. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 µg/L to 133 µg/L with an average of 48 µg/L.

Although the 1996 Water Quality Assessment Database does not contain monitoring data for the El Dorado Park lakes, the summary table in the Report does include a synopsis. pH was listed as partially supporting the aquatic life use and not supporting the contact recreation use: 116 measurements of pH were collected with values ranging from 6.9 to 9.4. Ammonium was not supporting the aquatic life or contact recreation uses; 45 ammonia samples were collected with concentrations ranging from non-detect to 1.92 mg-N/L. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Algae were listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

On May 8th 2008, the northern four lakes were sampled by Marine Biochemists. DO concentrations ranged from 7.36 mg/L to 8.63 mg/L, and pH ranged from 7.37 to 8.76. The concentrations of nitrates were highly variable and ranged from 0.3 mg/L to 3.0 mg/L; phosphates ranged from 0.09 mg/L to 0.58 mg/L. It is not clear from the report if the units on the nitrate samples were "as N" or "as NO_3" or if the units on the phosphate samples were "as P" or "as PO_4."

The El Dorado Park lakes were sampled February 26, 2009 and July 15, 2009 by USEPA and the Regional Board. In the northern four lakes, ammonia, nitrite, and nitrate concentrations were below detection limits (0.03 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively) during both monitoring events. TKN averaged 1.98 mg-N/L in the winter event and 0.92 mg-N/L in the summer event. Orthophosphate averaged 0.022 mg-P/L in the winter event and was less than the detection limit of 0.0075 mg-P/L in the summer. Total phosphorus concentrations averaged 0.129 mg-P/L in the winter event and 0.101 mg-P/L in the summer event. Chlorophyll *a* concentrations averaged 31 μg/L to 34 μg/L during both events in the northern four lakes. Profile measurements were conducted in Coyote and Alamo lakes during these events. pH ranged from 7.17 to 8.47 during both events. During the winter sampling event, DO concentrations near the surface of each lake were greater than 9 mg/L. DO declined with depth and concentrations measured at 0.3 meters above the bottom of both lakes were less than the target concentration of 5 mg/L (4.37 mg/L in Coyote Lake and 3.35 mg/L in Alamo Lake). During the summer event, DO concentrations in Coyote Lake decreased from 8.2 mg/L at the surface to 2.0 mg/L at 0.3 meters above the bottom of the lake. In Alamo Lake, DO concentrations decreased from 9.6 mg/L at the surface to 2.5 mg/L at 0.3 meters above the bottom of the lake.

In the southern two lakes, ammonia, nitrite, and nitrate concentrations were at or below detection limits (0.03 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively) during the winter monitoring event. TKN was 1.1 mg-N/L in both lakes. Orthophosphate and total phosphorus were approximately 0.016 mg-P/L and 0.03 mg-P/L, respectively, in both lakes. Chlorophyll *a* measurements in the winter were 5.3 μg/L and 5.9 μg/L. During the summer event, ammonia ranged from 0.04 mg-N/L to 0.1 mg-N/L and nitrate ranged from 0.09 mg-N/L to 0.12 mg-N/L. TKN was only measured in Nature Center South Lake and had a concentration of 0.98 mg-N/L. Orthophosphate was less than the detection limit of 0.0075 mg-P/L, and total phosphorus was approximately 0.139 mg-P/L in both lakes. Chlorophyll *a* concentrations ranged from 1.3 μg/L to 6.2 μg/L, although application of algaecide in mid-June may have continued to impact concentrations in July. pH ranged from 7.95 to 8.6 during both events. Both of the Nature Center lakes and Horseshoe Lake were treated with algaecides in mid-June (personal communication, Ed Gahafer, July 15, 2009), which may have reduced chlorophyll *a* concentrations during the July sampling event. However, Horseshoe Lake was not included in the nutrient monitoring, so this application does not impact sampling in the northern four lakes. Profile measurements were conducted in Nature Center North and Nature Center South lakes on February 26th 2009. DO concentrations decreased from greater than 8 mg/L at the surface to 3.8 mg/L and 4.1 mg/L, respectively, at 0.3 meters above the bottom of each lake. However, this may have been anomalous because during the July 15th 2009 event, profile measurements were conducted in Nature Center South Lake. DO concentrations decreased from 9.6 mg/L at the surface to 8.2 mg/L at 0.3 meters above the bottom of the lake.

Field data were also collected at shoreline stations at El Dorado Park on December 1, 2009. In the northern four lakes, temperatures ranged from 14.71 ºC to 17.01 ºC, while the pH range was 8.23 to 9.20. Temperatures were 14.94 ºC and 15.34 ºC and pH values were 8.17 and 8.12 at the Nature Center North and South lakes, respectively.

On August 10, 2010 the southern two lakes were sampled for nutrients. Ammonia concentrations ranged from 0.03 mg-N/L to 0.05 mg-N/L. TKN concentrations ranged from 0.67 to 1.03 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.23 mg-N/L to 0.24 mg-N/L. Orthophosphate ranged from 0.022 mg-P/L to 0.027 mg-P/L, and total phosphorus ranged from 0.027 mg-P/L to 0.038 mg-P/L. Chlorophyll *a* ranged from 4.81 μg/L to 6.23 μg/L. During this event, two continuous monitoring probes were deployed in each southern lake over a 24-hour period at depths of about 0.7 to 1.3 meters below the surface. DO concentrations ranged from 8.3 mg/L to 9.5 mg/L in Nature Center North Lake and from 9.5 mg/L to 12.6 mg/L in Nature Center South Lake. pH ranged from 8.5 to 9.0 in both lakes. On August 10, 2010, DO measurements collected at varying depths (from the surface to 0.3 meters above the bottom) in Nature Center North Lake ranged from 8.4 mg/L to 8.5 mg/L. In Nature Center South Lake, depth-varying DO ranged from 11.8 mg/L at the surface to 9.9 mg/L at 0.3 meters above the bottom of the lake.

On September 28, 2010 the southern two lakes were sampled again for nutrients. Ammonia concentrations ranged from <0.03 mg-N/L to 0.05 mg-N/L. TKN concentrations ranged from 0.79 to 0.86 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.36 mg-N/L to 0.41 mg-N/L. Orthophosphate ranged from 0.008 mg-P/L to 0.017 mg-P/L, and all total phosphorus measurements were below the detection limit of 0.0165 mg-P/L. Chlorophyll *a* ranged from 6.01 μg/L to 6.68 μg/L. During this event, two continuous monitoring probes were deployed in each southern lake over a 24-hour period at depths of about 1 to 1.3 meters below the surface. DO concentrations ranged from 7.4 mg/L to 8.2 mg/L in Nature Center North Lake and from 6.6 mg/L to 9.7 mg/L in Nature Center South Lake. pH ranged from about 7.6 to 8.1 in both lakes. On September 28, 2010, depth-variable DO measurements collected from the surface of Nature Center North Lake ranged from 9.2 mg/L to 10.9 mg/L. At 0.4 meters above the bottom, DO was measured as 9.2 mg/L. Depth-profile data were not collected at Nature Center South Lake due to time constraints.

In summary, pH exceedances have been observed in both systems (northern and southern). Ammonia concentrations exceeded the acute target in the northern lake system in the early 1990s. There were no exceedances of the acute or chronic ammonia criteria during any recent sampling events with associated pH and temperature measurements. DO concentrations have consistently been observed at less than the target concentration of 5 mg/L at 0.3 meters above the bottom of the northern four lakes during both 2009 monitoring events (at two lakes each time). Additionally, DO concentrations have been observed at less than the target concentration during one sampling event (winter 2009) in the southern two lakes. Algal concentrations in the northern lake system exceeded the target during historic and recent sampling. Chlorophyll *a* concentrations in the southern lake system have only been monitored recently: neither winter nor summer sampling show exceedances of the chlorophyll *a* target though summer concentrations may have been impacted by prior application of an algaecide. The nutrient TMDLs presented in Section 8.2.6 account for summer season critical conditions by assessing loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 μg/L in the northern four lakes. These reductions in nutrient loading are expected to alleviate pH, odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication.

8.2.3.1 Summary of pH Non-Impairment in the Southern Lake System

The Basin Plan states *"The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge."* In the southern two lakes 97 percent of the flow is potable water discharged to the lakes. Potable water was sampled for pH during the August and September 2010 sampling events, and measurements ranged from 7.98 to 8.22. In addition, the Long Beach Water Department reports the following: El Dorado Park lakes are in an area that primarily receives groundwater during the summer and purchased Metropolitan Water District water during the winter. The "MWD Zone" had a reported average pH of 7.9 (range of 7.4-8.2) and the "Groundwater Zone" had a reported average of 8.1 (7.8-8.2) (Long Beach Water Department, 2008). Based on this information, the potable water discharged to these lakes is not causing elevated pH levels. There are no other waste discharges that could be elevating the pH. As discussed in the linkage analysis (Section 8.2.5), the southern two lakes currently meet the chlorophyll *a* target, so nutrient loading is not elevating pH in those lakes. Based on these multiple lines of evidence, the southern two lakes in El Dorado are attaining beneficial uses and meeting pH water quality standards. USEPA concludes that preparing a

TMDL for pH is unwarranted at this time. USEPA recommends that the southern two lakes in El Dorado Park not be identified as impaired by pH in California's next 303(d) list.

8.2.4 Source Assessment

The source assessment for the El Dorado Park lakes includes loading estimates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading); irrigation (3.9 percent of the total irrigation volume is assumed to reach the lake), groundwater and potable water inputs used for supplemental water additions to the lake (Appendix F, Dry Weather Loading); and atmospheric deposition (Appendix E, Atmospheric Deposition). Table 8-3 summarizes the sources of existing loading to the northern lake system and Table 8-4 summarizes those loadings to the southern lake system. The majority of the phosphorus loading to the northern four lakes is a result of groundwater used for supplemental water additions. The nitrogen loading to the northern four lakes comes primarily from additional parkland loading such as excessive bird populations. The two southern lakes receive the majority of both phosphorus and nitrogen loading from the potable water input used to supplement water levels in the lakes. Section 8.2.5 describes the method used to estimate the additional parkland loading.

Table 8-3. Summary of Average Annual Flows and Nutrient Loading to the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
City of Long Beach	Runoff	1.69	3.08 (2.6)	20.3 (0.95)
City of Long Beach	Supplemental Water Additions (Groundwater)	110	71.5 (59.3)	287 (13.4)
City of Long Beach	Parkland Irrigation	20.6	9.29 (7.7)	320 (14.9)
City of Long Beach	Additional Parkland Loading	unknown	36.6 (30.4)	1,500 (70.0)
	Atmospheric deposition (to the lake surface)*	29.3	NA	16.5 (0.77)
Total		**164**	**120**	**2,144**

*Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

Table 8-4. Summary of Average Annual Flows and Nutrient Loading to the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
City of Long Beach	Runoff	0.309	0.563 (2.9)	3.71 (0.8)
City of Long Beach	Supplemental Water Additions (Potable Water)	105	13.67 (70.5)	269 (59.8)
City of Long Beach	Parkland Irrigation	2.54	1.15 (5.9)	39.6 (8.8)

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
City of Long Beach	Additional Parkland Loading	unknown	4.0 (20.6)	135 (30.0)
	Atmospheric deposition (to the lake surface)*	5.07	NA	2.8 (0.6)
Total		**113**	**19.4**	**450**

*Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

8.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on the El Dorado Park lakes, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated for each hydraulically connected system. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For the northern system, the model was calibrated to represent conditions in Coyote Lake because 1) this lake receives the groundwater input which represents the majority of nutrient loading to this system and 2) simulation of Coyote Lake individually was needed to calibrate the model within recommended guidelines (Walker, 1987). Based on the turnover ratio for this lake (Walker, 1987), the annual averaging period is most appropriate (i.e., annual loads are input to the model rather than summer season loads).

The NNE BATHTUB Tool was calibrated to recent average annual water quality data observed (calibration typically occurs to summer monitoring data but due to the limited monitoring data available for this lake an average of the summer and winter monitoring data were needed to create a more conservative analysis). Both nitrogen and phosphorus concentrations were underpredicted when the calibration factors were adjusted within normal range. To predict the average concentrations of total phosphorus (0.10 mg-P/L) and total nitrogen (1.36 mg-N/L) observed in Coyote Lake, loads from additional parkland sources were increased to 12 lb-P/yr and 300 lb-N/yr, respectively, with calibration factors on the net sedimentation rates set to 1. The amount of the additional parkland loading of phosphorus due to internal recycling was calculated with the method discussed in Appendix A (Nutrient TMDL Development) and is 4.66 lb-P/yr. This portion of the phosphorus load was subtracted out of the additional parkland sources category, and the model was recalibrated with a loading of 7.33 lb-P/yr. The resulting calibration factor on the net phosphorus settling rate is 0.85 which allows the model to account for internal loading implicitly. Though internal loading is not explicitly assigned a load allocation, reductions in external loading of phosphorus will ultimately result in reductions of internal cycling processes. Internal loading of nitrogen was not calculated because 1) internal loading is typically insignificant relative to external loading, and 2) empirical relationships for the estimation of internal nitrogen loading have not been developed. Thus, the additional parkland source loading and calibration factor for nitrogen were not changed. To simulate the average observed chlorophyll *a* concentration in Coyote Lake, the calibration factor on concentration was set to 0.92 for a predicted concentration of 36 µg/L. To estimate loading from additional parkland sources to the entire northern lake system, nutrient loads were scaled up by the ratio of surface areas for the northern lake system relative to Coyote Lake (30.1 acres / 6 acres = 5.0).

For the southern lake system, the cumulative surface area is 5.2 acres, the average depth is 4.6 ft, and the cumulative volume is 24 ac-ft. No historic monitoring data are available for this lake system and based on recent monitoring data, chlorophyll *a* concentrations are relatively low although application of algaecide in the southern two lakes in mid-June likely impacted chlorophyll *a* concentrations during the July 2009 monitoring event. Because insufficient data are available to calibrate the model to chlorophyll *a* concentrations, and no observations of chlorophyll *a* have exceeded the target concentration of 20 µg/L, these TMDLs will require that nutrient loading remain at existing levels as an antidegradation measure. If subsequent data are collected that will allow for calibration of the NNE BATHTUB model, then these TMDLs may be revisited. Note that the NNE BATHTUB Tool was set up to estimate loading from surrounding parkland areas. To predict the average concentrations of total phosphorus (0.061 mg-P/L) and total nitrogen (1.15 mg-N/L) observed in the southern two lakes, loads from the additional parkland sources were increased to 11.5 lb-P/yr and 135 lb-N/yr, respectively, with calibration factors on the net sedimentation rates set to 1. The amount of the phosphorous loading from additional parkland sources due to internal recycling was calculated with the method discussed in Appendix A (Nutrient TMDL Development) and is 7.6 lb-P/yr. This portion of the phosphorus load was subtracted out of the additional parkland source category, and the model was recalibrated with a loading from additional parkland sources of 4.0 lb-P/yr. The resulting calibration factor on the net phosphorus settling rate is 0.13 which allows the model to account for internal loading implicitly. Though internal loading is not explicitly assigned a load allocation, reductions in external loading of phosphorus will ultimately result in reductions of internal cycling processes. Internal loading of nitrogen was not calculated because 1) internal loading is typically insignificant relative to external loading, and 2) empirical relationships for the estimation of internal nitrogen loading have not been developed. Thus, the additional parkland loading and calibration factor for nitrogen were not changed. This configuration of the NNE BATHTUB Tool for the southern two lakes should not be considered a calibrated model as it was only used to develop an estimate of additional parkland loading and the calibration factors on the net phosphorus settling rate of 0.13 is out of the recommend range (0.5 to 2).

8.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Following calibration of the NNE BATHTUB Tool (Section 8.2.5), the allowable loading combinations of nitrogen and phosphorus were calculated using Visual Basic's GoalSeek function (Appendix A, Nutrient TMDL Development). The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus for the northern four lakes are

- 0.69 mg-N/L summer average (May – September) and annual average
- 0.069 mg-P/L summer average (May – September) and annual average

For the northern four lakes, the loading capacities for total nitrogen and total phosphorus are 902 lb-N/yr and 63.7 lb-P/yr, respectively. These loading capacities can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load (divided among WLAs and LAs) is 37.8 percent of the existing load of 2,144 lb-N/yr, or 811 lb-N/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. WLAs and LAs are developed assuming equal percent load reductions in all sources. The resulting TMDL equation for total nitrogen is then:

902 lb-N/yr = 109 lb-N/yr + 703 lb-N/yr + 90.2 lb-N/yr

For total phosphorus, the allocatable load (divided among WLAs and LAs) is 47.6 percent of the existing load of 120 lb-P/yr, or 57.3 lb-P/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. The resulting TMDL equation for total phosphorous is then:

63.7 lb-P/yr = 34.0 lb-P/yr + 23.3 lb-P/yr + 6.37 lb-P/yr

For the southern two lakes, existing levels of nitrogen and phosphorus loading appear to be resulting in attainment of the chlorophyll *a* target. Monitoring data indicate that the average in-lake total nitrogen concentration is 1.15 mg-N/L (Appendix G, Monitoring Data). Because the measured in-lake phosphorous concentrations varied widely between sampling events (<0.0165 mg-P/L to 0.138 mg-P/L), the phosphorus target concentration is based on an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10. This ratio was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 1.15 mg-N/L summer average (May – September) and annual average
- 0.115 mg-P/L summer average (May – September) and annual average

To prevent degradation of the southern two lakes, nutrient TMDLs will be allocated based on existing loading. These TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation. Note that the MOS is zero.

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load is equal to the existing load and is divided among WLAs and LAs, assuming equal percent load reductions from all sources.

450 lb-N/yr = 269 lb-N/yr + 181 lb-N/yr + 0 lb-N/yr

For total phosphorus, the allocatable load is equal to the existing load and allocated to WLAs and LAs.

19.4 lb-P/yr = 13.7 lb-P/yr + 5.7 lb-P/yr + 0 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined for the two lake systems based on simulation of allowable loads with the NNE BATHTUB model (see Section 8.2.5). These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

8.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs for total phosphorus and total nitrogen for the northern and southern lake systems as well as alternative WLAs for total phosphorous and total nitrogen for the northern lake system. The alternative WLAs will be effective and supersede the WLAs listed in Table 8-5 for the northern lake system if the conditions described in Section 8.2.6.1.2 are met.

Under either wasteload allocation scheme responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on the Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

8.2.6.1.1 Wasteload Allocations

There are no MS4 discharges to the El Dorado Park lakes and no other (non-MS4) permitted dischargers in the watershed. The supplemental water sources to maintain lake levels are the only sources of nutrient loading to the El Dorado Park lakes that are assigned WLAs. The WLA for this source to the northern four lakes represents a 62.2 percent reduction in total nitrogen loading and a 52.4 percent reduction in total phosphorus loading (Table 8-5) and must be met as a one year average. In contrast, the WLAs for the supplemental water additions to the southern two lakes are equivalent to existing levels of loading (Table 8-4) and must be met as a three year average. These loading values (in pounds per year) represent the TMDLs wasteload allocations (Table 8-5 and Table 8-6). Each WLA must be met at the point of discharge.

Table 8-5. Wasteload Allocations for Nutrient Loading to the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus[1,2] (lb-P/yr)	Total Nitrogen[1,2] (lb-N/yr)
City of Long Beach	Supplemental Water Additions	110	34.0	109

[1] A one year average will be used to evaluate compliance.

[2] The wasteload allocation must be met at the point of discharge.

Table 8-6. Wasteload Allocations for Nutrient Loading to the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus[1] (lb-P/yr)	Total Nitrogen[1] (lb-N/yr)
City of Long Beach	Supplemental Water Additions	105	13.7	269

[1] Each wasteload allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

8.2.6.1.2 Alternative "Approved Lake Management Plan Wasteload Allocations" for the Northern Lake System

Concentration-based WLAs not exceeding the concentrations listed in Table 8-7 are effective and supersede the corresponding WLAs for the City of Long Beach in Table 8-5 if:

1. The City of Long Beach requests that concentration-based wasteload allocations not to exceed the concentrations established in Table 8-7 apply to it;
2. The City of Long Beach provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 8-2. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to

reduce nutrient availability from sediments. The City of Long Beach may use monitoring data and modeling to show that the water quality criteria, targets and requested WLAs will be met;

3. The Regional Board Executive Officer approves the request and applies concentration-based wasteload allocations for total nitrogen and total phosphorus. These wasteload allocations are not to exceed the concentrations in Table 8-7 as a summer average (May-September) and annual average, and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based wasteload allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 8-7. Alternative Wasteload Allocations for Phosphorus and Nitrogen in the Northern Lake System of the El Dorado Park Lakes if an Approved Lake Management Plan Exists

Responsible Jurisdiction	Input	Maximum Allowable Wasteload Allocation Total Phosphorus[1,2] (mg-P/L)	Maximum Allowable Wasteload Allocation Total Nitrogen[1,2] (mg-N/L)
City of Long Beach	Supplemental Water Additions	0.1	1.0

[1] A one year average will be used to evaluate compliance.

[2] The concentration-based wasteload allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

8.2.6.2 Load Allocations

These TMDLs establish load allocations (LAs) for total phosphorus and total nitrogen for the northern and southern lake systems as well as alternative LAs for total phosphorous and total nitrogen for the northern lake system. The alternative LAs in the northern lake system will be effective and supersede the LAs listed in Table 8-8 if the conditions described in Section 8.2.6.2.2 are met.

8.2.6.2.1 Load Allocations

There are no storm drains that discharge runoff flows into the El Dorado Park lakes. Therefore, all loads associated with the surrounding drainage area are assigned LAs. Atmospheric deposition and additional parkland loading are also assigned LAs. For the northern four lakes, total phosphorus LAs represent a 52.4 percent reduction in existing loading, and total nitrogen LAs represent a 62.2 percent reduction in existing loading (Table 8-8). LAs are provided for each responsible jurisdiction and input and must be met at the point of discharge. These loading values (in pounds per year) represent the TMDLs load allocations (Table 8-8 and Table 8-9).

Table 8-8. Load Allocations for Nutrient Loading to the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr)[1,2]	Total Nitrogen (lb-N/yr)[1,2]
City of Long Beach	Runoff	1.69	1.47	7.68
City of Long Beach	Parkland Irrigation	20.6	4.42	121

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr)[1,2]	Total Nitrogen (lb-N/yr)[1,2]
City of Long Beach	Additional Parkland Loading	unknown	17.4	568
	Atmospheric deposition (to the lake surface)[3]	29.3	NA	6.24
Total		**54.2**	**23.3**	**703**

[1]A one year average will be used to evaluate compliance.

[2] Each load allocation must be met at the point of discharge.

[3] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

For the southern two lakes, the LAs are set equal to existing loading rates. Assuming flow volumes remain at existing levels (Table 8-4), targeted concentrations of nitrogen and phosphorus in the city of Long Beach runoff to the southern two lakes may be 0.670 mg-P/L and 4.42 mg-N/L. Targeted concentrations in the irrigation returns may be 0.166 mg-P/L and 5.73 mg-N/L (3.9 percent of the total irrigation volume to both lake systems is assumed to reach the lake; Appendix F, Dry Weather Loading). The targeted nitrogen concentrations for precipitation to the surfaces of the southern two lakes may be 0.20 mg-N/L. Targeted concentrations for the additional parkland loading cannot be estimated because the associated flow volumes are unknown.

Table 8-9. Load Allocations for Nutrient Loading to the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr)[1]	Total Nitrogen (lb-N/yr)[1]
City of Long Beach	Runoff	0.309	0.563	3.71
City of Long Beach	Parkland Irrigation	2.54	1.15	39.6
City of Long Beach	Additional Parkland Loading	unknown	4.0	135
	Atmospheric deposition (to the lake surface)[2]	5.07	NA	2.8
Total		**6.75**	**5.7**	**181**

[1] Each load allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

8.2.6.2.2 Alternative "Approved Lake Management Plan Load Allocations" for the Northern Lake System

Concentration-based load allocations for the northern lake system not exceeding the concentrations listed in Table 8-10 are effective and supersede corresponding load allocations for the City of Long Beach in Table 8-8 if:

1. The City of Long Beach requests that concentration-based load allocations not to exceed the concentrations established in Table 8-10 apply to it;

2. The City of Long Beach provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 8-2. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The City of Long Beach may use monitoring data and modeling to show that the water quality criteria, targets and requested load allocations will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based load allocations for total nitrogen and total phosphorus. These load allocations are not to exceed the concentrations in Table 8-10 as a summer average (May-September) and annual average; and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based LA must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 8-10. Alternative Load Allocations of Nutrient Loading to the Northern Lake System of the El Dorado Park Lakes if an Approved Lake Management Plan Exists

Responsible Jurisdiction	Input	Maximum Allowable Load Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Load Allocation Total Nitrogen[1] (mg-N/L)
City of Long Beach	Runoff	0.1	1.0
City of Long Beach	Parkland Irrigation	0.1	1.0
City of Long Beach	Additional Parkland Loading	0.1	1.0

[1] Each concentration-based load allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

8.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. To account for the uncertainties concerning the relationship between nutrient loading and the resultant in-lake chlorophyll *a*, an explicit MOS is included in the northern lake system TMDLs. This explicit MOS is set at 10 percent of the loading capacity for total phosphorus and total nitrogen. The southern lake system is currently achieving the in-lake chlorophyll *a* target, and TMDLs are being established at the existing loads. This conservative anti-degradation measure is the implicit margin of safety for these TMDLs.

8.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. The northern lake system TMDLs are expected to alleviate any pH and ammonia problems associated with excessive nutrient loading and eutrophication. The southern lake system TMDLs are based on existing conditions as an anti-degradation measure since nitrogen and phosphorus levels are currently achieving the chlorophyll *a* target level. These TMDLs therefore protect for critical conditions.

8.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs do present a maximum daily load according to the guidelines provided by USEPA (2007).

For each lake system, the primary contributor of nutrient loading is the supplemental water addition. Daily loads are calculated by multiplying the maximum daily flow rates from each source with the average allowable concentrations consistent with attaining the TMDLs. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

For the northern four lakes, the average allowable concentration of total nitrogen in the supplemental water addition is the allowable load from this source (109 lb-N/yr) divided by the average annual flow from this source (110 ac-ft/yr) or (0.363 mg-N/L) (see Table 8-5). For total phosphorus, the average allowable concentration in the supplemental water addition is the allowable load from this source (34.0 lb-P/yr) divided by the average annual flow (110 ac-ft/yr) or (0.113 mg-P/L) (see Table 8-5). Peak daily flow from the supplemental water addition is estimated as the maximum metered flow rate (30.8 ac-ft/mo) divided by the number of days in the peak flow month (31) or 0.994 ac-ft/d. Total maximum daily loads from this source are 0.981 lb-N/d and 0.307 lb-P/d.

For the southern two lakes, daily maximum loads will likely result from use of supplemental water additions to the lakes: this source contributes the majority of the nitrogen and phosphorus loading to this lake system. The peak daily flow rate from this source is estimated from the maximum monthly metered flow rate (29.4 ac-ft/mo) divided by the number of days in the month (31) or 0.948 ac-ft/d. The average allowable nitrogen concentration is the allowable load from the supplemental water addition (269 lb-N/yr) divided by the average annual flow (105 ac-ft/yr) or (0.94 mg-N/L) (see 8.2.6.1). For total phosphorus, the average allowable concentration is the allowable load from the supplemental water addition (13.7 lb-P/yr) divided by the average annual flow (105 ac-ft/yr) or (0.048 mg-P/L) (see 8.2.6.1). Daily maximum allowable loads from supplemental water additions at the southern two lakes are 2.43 lb-N/d and 0.124 lb-P/d.

As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake systems every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

8.2.6.6 Future Growth

The El Dorado Park lakes watershed is comprised entirely of parkland. It is not likely that the watershed will be developed and it is expected to remain as open space. No load allocation has been set aside for future growth, and it is unlikely that any dischargers will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

8.3 MERCURY IMPAIRMENT

The 1996 LA Region Water Quality Assessment Report lists mercury in fish tissue as an impairment of the El Dorado Park lakes, although no study was cited directly. No mercury fish tissue data were included in the summary table or the accompanying database.

Although the data were not included in the Water Quality Assessment Report, the Toxic Substances Monitoring Program (TSMP) collected fish tissue samples in the late 80s and 90s that exceeded the fish tissue guideline of 0.22 ppm. Recent data collected by the Surface Water Ambient Monitoring Program (SWAMP) indicate that fish tissue levels of mercury remain elevated (see Section 8.3.3). All fish tissue samples were collected from either Coyote Lake or Alamo Lake, both of which are in the system comprised by the northern four lakes. Thus, there is no direct evidence of fish tissue impairment for the southern two lakes.

In 2008, the Southern California Coastal Water Research Project (SCCWRP) published a report titled “Extent of Fishing and Fish Consumption by Fishers in Ventura and Los Angeles County Watersheds.” The purpose of the study was to document the fishing habits and consumption rates of fishers in these counties (SCCWRP, 2008). The El Dorado Park lakes were visited three times, during which 45 fishers were observed. Eighteen fishers were interviewed, and 11 percent of those consume fish caught from these lakes. The El Dorado Park lakes are also part of the California Department of Fish and Game “Fishing in the City” program which encourages people in the Los Angeles area to fish from local waterbodies. Fish are periodically stocked and fishing is only allowed from the northern four lakes.

8.3.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each Region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Applicable water quality criteria are also specified in the California Toxics Rule (USEPA, 2000a). The existing beneficial uses assigned to the El Dorado Park lakes include REC1, REC2, WARM, WILD, MUN, and WET. Descriptions of these uses are listed in Section 2 of this TMDL report. Concentrations of mercury measured in fish tissue collected from the northern four lakes indicate that the REC1, REC2, and WARM, uses are currently impaired. Data are not available to assess compliance with the fish tissue standard in the southern two lakes. At high enough concentrations WILD, MUN and WET uses could become impaired.

8.3.2 Numeric Targets

Numeric targets for mercury in the El Dorado Park lakes apply to both the water column and fish tissue. Water column targets are based on beneficial use. For waters designated MUN (existing, potential, or intermittent), the Basin Plan lists a total mercury maximum contaminant level of 0.002 mg/L, or 2 μg/L. The California Toxics Rule includes total mercury human health criteria for the consumption of "water and organisms" or "organisms only" as 0.050 μg/L and 0.051 μg/L, respectively (USEPA, 2000a). California often implements these values on a 30-day average. Because El Dorado Park lakes do not have an existing MUN designated use, a total mercury water column target of 0.051 μg/L (51 ng/L) for "organisms only" is the appropriate target.

In addition, a water column target for dissolved methylmercury of 0.081 ng/L is applicable for the El Dorado Park lakes. This value was calculated by dividing the fish tissue guideline (0.22 ppm) with a national bioaccumulation factor (for dissolved methylmercury) of 2,700,000 applicable for trophic level 4 fish (and multiplying by a factor of 10^6 to convert from milligrams to nanograms).

The fish contaminant goal (FCG) for methylmercury defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 220 ppb or 0.22 ppm (wet weight). This concentration is protective of human and wildlife consumers of trophic level four fish. The target length for comparison to this target is 350 mm (13.8 inches) in largemouth bass. Refer to Section 2.2 of this report for more information regarding these targets.

8.3.3 Summary of Monitoring Data

Total mercury concentrations in the water columns of the El Dorado Park lakes have been measured at various locations since 1992. In-lake water column mercury concentrations were measured in July and August 1992 in Alamo Lake as part of the Urban Lakes Study (UC Riverside, 1994). All 12 measurements were less than the detection limit of 0.51 μg/L (500 ng/L). As the detection limit of this dataset is 10 times higher than the water quality criterion for mercury (51 ng/L), it is difficult to assess compliance in terms of a water column concentration.

More recent samples collected in February and July 2009 were collected and analyzed with ultra-clean methods and detection limits no greater than 0.15 ng/L. Samples were collected from Coyote, Alamo, and Nature Center South lakes. All total mercury samples collected during these events ranged from 0.41 ng/L to 1.17 ng/L and were more than one order of magnitude less than the total mercury water column target. Concentrations of total methylmercury in the northern four lakes ranged from 0.041 ng/L to 0.072 ng/L with an average concentration of 0.056 ng/L, which is less than the dissolved target concentration (0.081 ng/L). The observed concentration of total methylmercury in the southern two lakes was 0.02 ng/L and was therefore less than the dissolved target concentration.

Mercury concentrations were also measured for each supplemental water source. Total mercury concentrations measured in the groundwater ranged from 131 ng/L to 142 ng/L, and methylmercury concentrations in the groundwater ranged from 0.109 ng/L to 0.215 ng/L. Thus, total and methylmercury concentrations in the groundwater used for supplemental water additions to the northern four lakes exceeded the water column targets of 51 ng/L and 0.081 ng/L, respectively. Total mercury concentrations measured from the potable water input ranged from 1.46 ng/L to 2.84 ng/L; methylmercury concentrations were approximately 0.02 ng/L. Neither total nor methylmercury concentrations in the potable water source exceeded the respective targets.

Mercury concentrations in the fish tissue of largemouth bass have been measured in the northern lakes at El Dorado Park since 1991. Coyote Lake was sampled by the TSMP in the 1990s and analyzed as composites, with six fish in each composite. The California Surface Water Ambient Monitoring Program (SWAMP) sampled individual fish from Alamo Lake during the summers of 2007 and 2010. No fish tissue samples have been collected from the southern two lakes at the nature center and recreational fishing is not allowed in those two lakes.

Figure 8-9 shows the total mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length.

As expected, fish tissue mercury concentrations increase with length. Concentrations exceed 0.22 ppm in all individual or composite samples greater than 370 mm. Fourteen individual and three composite samples had fish tissue concentrations greater than the target, while six individual samples had concentrations less than the target. All of the fish tissue data were reported as total mercury concentrations, of which over 90 percent is expected to be in the methyl form (USEPA, 2001a). These total mercury data were compared to the methylmercury fish contaminant guidelines, resulting in conservative assessments.

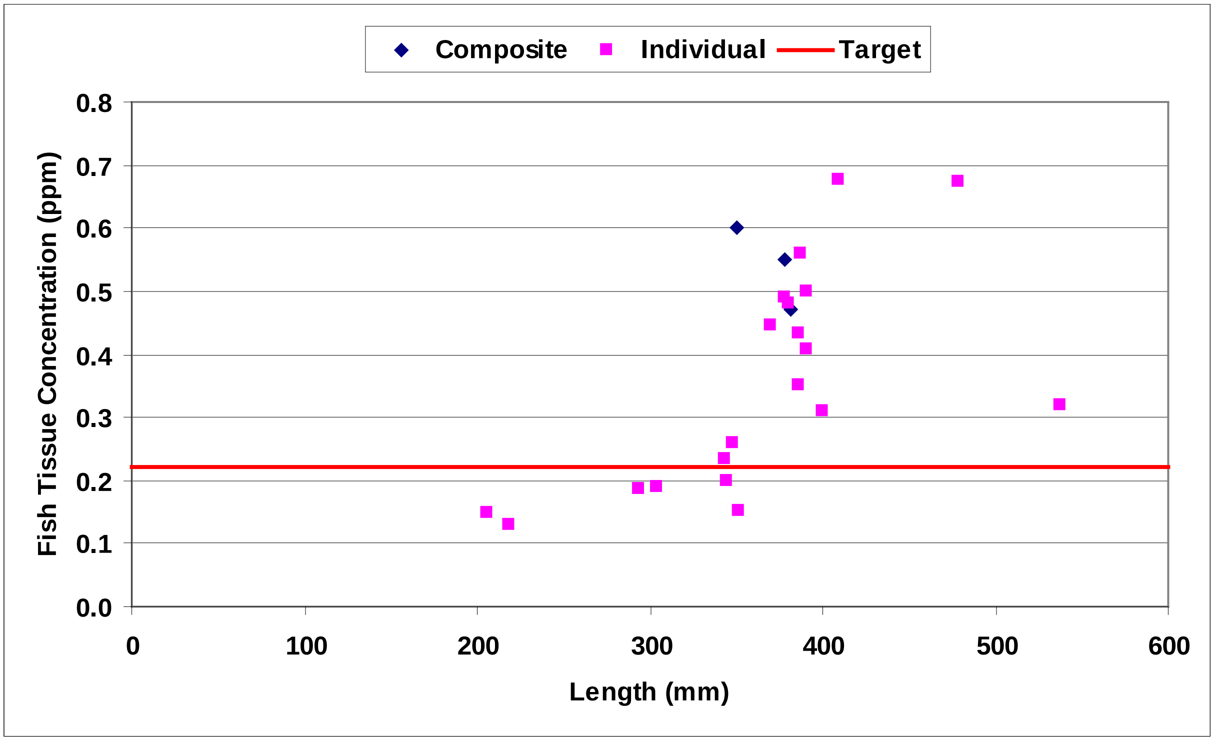

Figure 8-9. Mercury Concentrations in Largemouth Bass Collected from the El Dorado Park Lakes (1991-2010)

8.3.4 Source Assessment

There are several sources of mercury loading to the El Dorado Park lakes. For the northern four lakes, the majority of mercury loading originates from the groundwater that is pumped into Coyote Lake (Figure 8-5) to maintain water levels in the system. Atmospheric deposition is the second largest source of mercury loading to the northern four lakes and the largest contributor to the southern two lakes. The potable water source pumped into Nature Center North Lake (Figure 8-5) is the second largest source of mercury to the southern system. Loads resulting from precipitation and irrigation runoff from the adjacent parklands (3.9 percent of the total irrigation volume is assumed to reach the lake; Appendix F, Dry Weather Loading) contribute insignificant amounts of mercury relative to the other sources.

Table 8-11 and Table 8-12 summarize the total mercury loads to the northern four lakes and southern two lakes, respectively. Estimation of loading from runoff, direct inputs, and irrigation of parkland are discussed in more detail in Appendices D and F (Section 8 of both appendices). The atmospheric deposition component of the mercury load is discussed in Appendix E (Atmospheric Deposition). In both lake systems, the city of Long Beach runoff is assigned a load allocation (associated with 185 acres in the northern lake system and 33.8 acres in the southern lake system). Irrigation and atmospheric deposition will also receive load allocations in both systems; however, the supplemental water additions will receive wasteload allocations. The northern four lakes receive approximately 20 times more mercury annually than the southern two lakes.

Table 8-11. Summary of Existing Total Mercury Loading to the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Total Annual Hg Load (g/yr)	Percent of Load
City of Long Beach	Runoff	0.0109	0.04
City of Long Beach	Supplemental Water Additions (Groundwater)	18.5	73.8
City of Long Beach	Parkland Irrigation	0.0371	0.15
	Atmospheric deposition (to the lake surface)*	6.49	26.0
Total		**25.0**	**100**

*Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

Table 8-12. Summary of Existing Total Mercury Loading to the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Total Annual Hg Load (g/yr)	Percent of Load
City of Long Beach	Runoff	0.00199	0.13
City of Long Beach	Supplemental Water Additions (Potable Water)	0.368	24.6
City of Long Beach	Parkland Irrigation	0.00458	0.31
	Atmospheric deposition (to the lake surface)*	1.12	74.9
Total		**1.49**	**100**

*Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

8.3.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. Specifically, models of watershed loading of mercury are combined with an estimated rate of bioaccumulation in the lake. This enables a translation between the numeric target (expressed as a fish tissue concentration of mercury) and mercury loading rates. The loading capacity is then determined via the linkage analysis as the mercury loading rate that is consistent with meeting the target fish tissue concentration.

Neither data nor resources are available to create and calibrate detailed lake response models for mercury cycling in the El Dorado Park lakes. The TMDL target is based on achieving acceptable concentrations in fish. In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of methylmercury in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of methylmercury from lake sediments. In lakes with high sedimentation rates, fish tissue concentrations are therefore likely to respond approximately linearly to reductions in the watershed methylmercury and total mercury load. For the El Dorado Park lakes, watershed loading is an insignificant amount of the total load compared to the loads from supplemental water addition and air deposition. However, it is expected that fish tissue concentrations will also respond linearly to reductions of direct inputs and atmospheric deposition, which contribute the majority of the loading to each lake system in El Dorado Park.

Nationally, authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between methylmercury in water and methylmercury in fish tissue normalized to length. However, this relationship is well-approximated by a linear relationship for the ranges of fish tissue concentration of concern for these impaired lakes. For the lakes where fish tissue data are available (the northern four lakes), the groundwater supplemental water additions contribute over 70 percent of the total mercury load and 97.5 percent of the methylmercury load (see Section 8 in Appendices D and F; Wet Weather Loading and Dry Weather Loading). Until such time as a lake response model for mercury is constructed, and sufficient calibration data are collected, an assumption of an approximately linear response of fish tissue concentrations to changes in external loads is sufficient for the development of a TMDL. For a more detailed discussion of the linkage analysis between mercury loading and fish body burden, see Section 3.2.3 of this report.

8.3.6 TMDL Summary

A waterbody's loading capacity represents the maximum pollutant load that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum load consistent with meeting the numeric target of 0.22 ppm for mercury in largemouth bass. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix C (Mercury TMDL Development).

Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass. To do this, a linear regression analysis was performed on tissue concentrations versus length for the northern four El Dorado Park lakes. The resulting regression equation is

$$Hg(fish) = -0.15316 + 0.001461 \cdot Len, R^2 = 0.35$$

where *Hg(fish)* is the total mercury concentration in largemouth bass (ppm) and *Len* is length in mm. The regression analysis is shown in Figure 8-10, along with the one-sided 95 percent upper confidence limits on mean predictions about the regression line (95 percent UCL) and the 95 percent upper prediction intervals on individual predicted concentrations (95 percent UPI). The UPI gives the confidence limit on the individual predictions for a given length while the UCL gives the confidence limit on the average of the predictions for a given length. This regression has a non-zero intercept and should not be considered valid for lengths less than 200 mm.

For mercury, long-term cumulative exposure is the primary concern. Therefore, it is appropriate to use the 95 percent UCL rather than the UPI to provide a Margin of Safety on the appropriate age class. Use of the UCL provides an explicit Margin of Safety because it represents an upper confidence bound on the long-term exposure concentration.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in the system comprised by the northern four El Dorado Park lakes. The TMDL target is established for a 350 mm largemouth bass (see Section 2.2.8). The predicted mercury concentration based on the UCL equation for this length is compared to the target concentration to determine the required reduction in mercury loading, which includes an explicit Margin of Safety as described above.

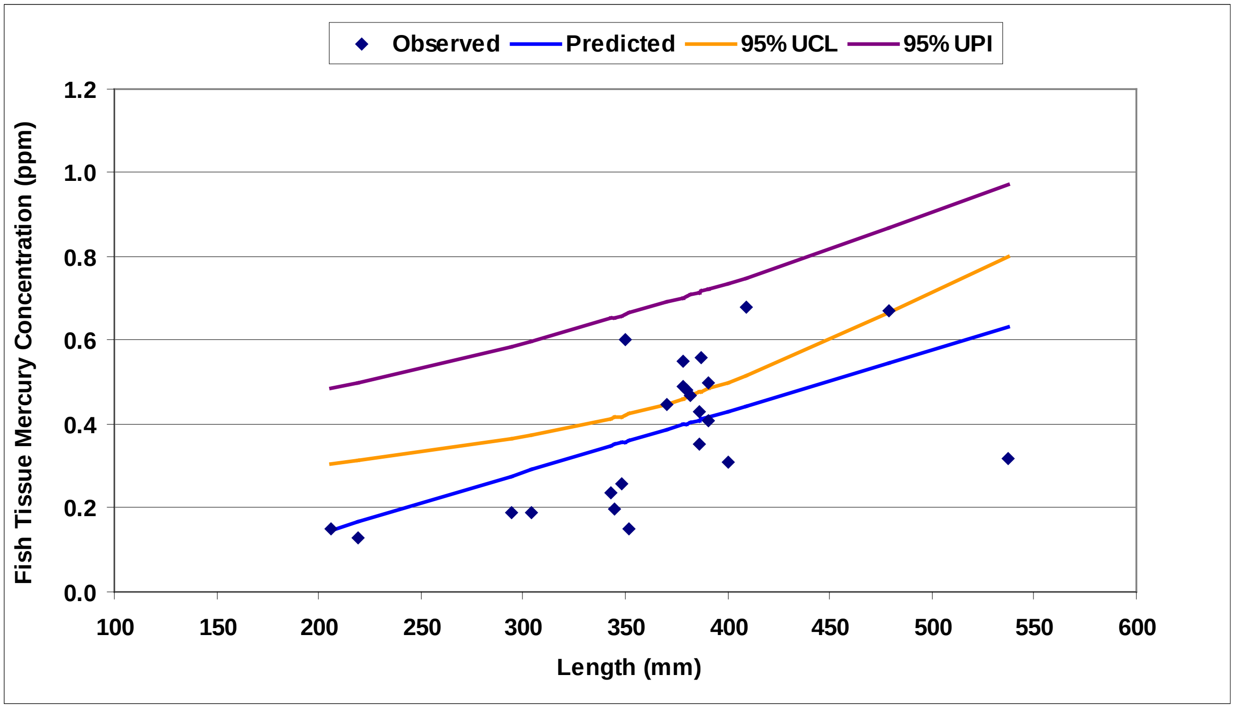

Figure 8-10. Regression Analysis of Mercury in El Dorado Park Lakes Largemouth Bass

For the northern four El Dorado Park lakes, the fraction of the existing load consistent with attaining the target (the loading capacity) is the ratio of the target (0.22 ppm) to the best estimate of current average concentrations in the target fish population. The difference between the direct regression estimate and the 95 percent UCL provides the Margin of Safety. Therefore, the allocatable fraction of the existing load (the loading capacity less the Margin of Safety) is the ratio of the target to the 95 percent UCL. The resulting loading capacities and allocatable loads are expressed as fractions of the existing load as summarized in Table 8-13. This analysis indicates that a 47.8 percent reduction in mercury loading to the northern four lakes will be required to bring fish tissue concentrations in 350 mm largemouth bass down to 0.22 ppm (see Section 2.2.8).

Table 8-13. Estimated Total Mercury Loading Capacity and Allocatable Load for the Northern Lake System of the El Dorado Park Lakes (as Fractions of the Existing Load)

Parameter	Value
Target Concentration (ppm)	0.22
Target Length (mm)	350
Predicted Mercury Concentration at Target Length (ppm)	0.358
95^{th} Percent UCL (ppm)	0.422
Loading Capacity (ratio of target to predicted value)	0.614
Allocatable Load (ratio of target to 95^{th} percent UCL)	0.522
Required Reduction in Existing Load (1 minus allocatable fraction)	0.478
Margin of Safety Fraction (loading capacity fraction minus allocatable fraction)	0.093

The loading capacity can also be expressed as grams per year (g/yr) of total mercury using the existing loads presented in Table 8-11 and the calculated fractions of the existing load (Table 8-13). For the northern four lakes, the loading capacity is 61.4 percent of the existing load of 25.0 g/yr, or 15.4 g/yr.

$$TMDL = \sum WLA + LA + MOS$$

The allocatable load for the northern four lakes (divided among WLAs and LAs) is 52.2 percent of the existing load. Thus the allocatable load is 13.0 g/yr which represents 84.4 percent of the loading capacity. The Margin of Safety is 9.3 percent of the loading capacity.

$$15.4g / yr = \sum 9.62g / yr + 3.41g / yr + 2.32g / yr$$

For the southern two lakes, there are no fish tissue data to indicate whether or not the system is impaired, and the observed total mercury concentrations in the water column are well below the targets (Section 8.3.3). The following comparisons may be made to the northern four lakes:

1) The ratio of the allowable load to the northern four lakes (13.0 g/yr) divided by their cumulative volume (243 ac-ft) is 0.05. The ratio of the existing load to the southern two lakes (1.49 g/yr) divided by their cumulative volume (24 ac-ft) is 0.06. Thus, the volume-weighted existing load to the southern two lakes is approximately equal to the volume-weighted allowable load to the northern four lakes.

2) The northern four lakes require a reduction in mercury loading of 47.8 percent. Over 73 percent of the loading to the northern four lakes is a result of direct groundwater input. Based on data collected and analyzed with ultra-clean methods in February and July 2009, the average total mercury concentration of the groundwater is 136 ng/L (nearly three times the water column target of 50 ng/L). The largest contributor of mercury loading to the southern two lakes is atmospheric deposition. Areal rates of atmospheric mercury deposition to each system are the same. The second largest contributor of mercury to the southern lake system is the potable water input, which has an average concentration of 2.84 ng/L, which is more than one order of magnitude below the water column target (based on data collected in February and July 2009). If the existing loading from atmospheric deposition to the northern four lakes was held constant, but the groundwater concentration was reduced to the same level observed in the potable water input, the total mercury load to the northern four lakes would be 7.4 g/yr, which is a reduction from existing loading of almost 71 percent. Thus, the two major sources of loading to the southern two lakes would not cause impairment of the northern lake system, assuming the volume of water applied to the northern four lakes remains at current levels.

3) The average total methylmercury concentration observed in the groundwater input is 0.162 ng/L, which is two times higher than the dissolved methylmercury water column target (0.081 ng/L) (Note: data are presented for the total fraction, while the water column target is for the dissolved fraction, resulting in a conservative assessment). The potable water input has an observed concentration of 0.02 ng/L (below the 0.081 ng/L methylmercury water column target). As bioaccumulation is directly proportional to methylmercury concentration, the southern two lakes are less likely to exhibit fish tissue concentrations that are as high as those seen in the northern lakes.

4) Fishing is not allowed from the two southern lakes and has not been observed during any of the recent monitoring events.

While none of the above statements offer a direct comparison to the mercury fish tissue guideline, they do indicate that impairment is unlikely. Since the southern lake system has very different mercury loading than the northern lake system, the TMDL for the southern lake system will be different than for the northern lake system. For this TMDL, total mercury loads in the southern two lakes will be held to existing levels as an antidegradation measure until fish tissue data are collected to either confirm or deny the mercury impairment. The MOS for the southern two lakes will be zero.

$$TMDL = \sum WLA + LA + MOS$$

$$1.49g / yr = \sum 0.368g / yr + 1.13g / yr + 0g / yr$$

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

8.3.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). The direct inputs to the northern and southern lake systems are assigned WLAs. Table 8-14 and Table 8-15 summarize the existing total mercury loads and WLAs for these sources. This TMDL establishes WLAs at their point of discharge. For the northern four lakes, the WLA is a 47.8 percent reduction from the existing loads (Table 8-14); for the southern two lakes, the wasteload allocation (Table 8-15) is equal to the existing load (Table 8-12). These loading values (in grams per year) represent the TMDLs wasteload allocations (Table 8-14 and Table 8-15) and each wasteload allocation must be met at the point of discharge. However, point source discharges to the lake must also meet CTR criteria for total mercury so the targeted concentration for the northern lake system must be at a maximum of 51 ng/L. At a maximum concentration of 51 ng/L a greater volume of water may be discharged to the lakes than is currently discharged and still attain the mass-based WLA. In addition to the WLAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L applies. -

Table 8-14. Wasteload Allocations of Total Mercury for the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Existing Annual Hg Load (g/yr)	Wasteload Allocation[1] (g/yr)
City of Long Beach	Supplemental Water Additions	18.5	9.62

[1] Each mass-based wasteload allocations must be met at the point of discharge.

Table 8-15. Wasteload Allocations of Total Mercury for the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Existing Annual Hg Load (g/yr)	Wasteload Allocation[1] (g/yr)
City of Long Beach	Supplemental Water Additions	0.368	0.368

[1] Each mass-based wasteload allocations must be met at the point of discharge.

8.3.6.2 Load Allocations

Load allocations of total mercury are required for the atmospheric deposition and watershed sources. Table 8-16 and Table 8-17 summarize the existing total mercury loads and LAs for the northern and southern lake systems, respectively. The LAs for the northern system are a 47.8 percent reduction from the existing loads. The LAs for the southern two lakes are equal to the existing load (Table 8-12); no reductions are required for the southern lake system. LAs are provided for each responsible jurisdiction and input. These loading values (in grams per year) represent the TMDLs load allocations (Table 8-16 and Table 8-17) and each load allocation must be met at the point of discharge. In addition to the LAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L applies.

Table 8-16. Load Allocations of Total Mercury for the Northern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Existing Annual Hg Load (g/yr)	Load Allocation[1] (g/yr)
City of Long Beach	Runoff	0.0109	0.0057
City of Long Beach	Parkland Irrigation	0.0371	0.0193
	Atmospheric deposition (to the lake surface)[2]	6.49	3.38
Total		**6.54**	**3.41**

[1] Each mass-based load allocations must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

Table 8-17. Load Allocations of Total Mercury for the Southern Lake System of the El Dorado Park Lakes

Responsible Jurisdiction	Input	Existing Annual Hg Load (g/yr)	Load Allocation[1] (g/yr)
City of Long Beach	Runoff	0.00199	0.00199
City of Long Beach	Parkland Irrigation	0.00458	0.00458
	Atmospheric deposition (to the lake surface) [2]	1.12	1.12
Total		**1.13**	**1.13**

[1] Each mass-based load allocations must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

8.3.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. The TMDL for the northern lake system includes both an implicit and explicit MOS. The implicit MOS includes comparing the total mercury concentration reported for fish tissue samples to the methylmercury fish tissue target. Most mercury in fish tissue is in the methyl form, but not all, so this is a conservative assumption. In this TMDL, an explicit MOS is also included by selecting the 95 percent UCL to represent the existing mean fish tissue concentration rather than the regression predicted mean (Figure 8-10). Use of the UCL provides a margin of safety because it represents an upper confidence bound on the long-term exposure concentration. For the northern lake system, the fraction of the existing load set aside for the explicit MOS is 0.093, or 2.32 g/yr, which represents 9.3 percent of the loading capacity. The TMDL for the southern lake system includes an implicit MOS. This lake system is likely achieving the fish tissue target and TMDLs are being established at the existing mercury loads. This conservative anti-degradation measure is the implicit margin of safety for this TMDL.

8.3.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target in the northern lake system and maintaining existing water quality in the southern lake system. Because fish bioaccumulate mercury, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, annual mercury loading is more important for the attainment of standards than instantaneous or daily concentrations, and the TMDL is proposed in terms of annual loads. For the northern four lakes, the primary source of mercury load is the groundwater input, and peak flows do represent a critical condition in terms of peak loading rates. The majority of supplemental flows are added to each system during the dry season (May through October) when precipitation is generally low and evaporation rates are high. For the southern two lakes, the largest source of mercury loading is atmospheric deposition which is not known to have a critical condition.

However, the greatest impact to fish occurs when methylmercury, a more biologically available form of mercury, is at its greatest concentration. Bacterially mediated methylation of mercury varies seasonally and typically results in the greatest methylmercury concentrations in the water column in the late summer. However, the impact of seasonal and other short-term variability in loading is damped out by the biotic response since the target concentrations in tissues of edible sized game fish integrate exposure over a number of years. Additionally, this TMDL includes a methylmercury water column target applicable year round. This TMDL therefore protects for critical conditions.

8.3.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. Although it is long-term cumulative load rather than daily loads of mercury that are driving the bioaccumulation of mercury in fish in the El Dorado Park lakes, these TMDLs does present a maximum daily load according to the guidelines provided by USEPA (2007). These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

For the northern four lakes, the primary contributor of mercury loading is the groundwater input. Peak daily flow from this source is estimated as the maximum metered flow rate (30.8 ac-ft/mo) divided by the number of days in the peak flow month (31) or 0.994 ac-ft/d. The average mercury concentration consistent with achieving the long-term loading target for the northern four lakes is the allowable load from this source (9.62 g/yr; Table 8-14) divided by the total average annual flowrate to the lake system (110 ac-ft, see Appendices D and F) which is 70.9 ng/L. The daily maximum allowable load of mercury to the northern system in the El Dorado Park lakes is the highest measured groundwater flowrate multiplied by the mercury concentration that will be consistent with achieving the long-term loading target, or 0.087 g/d.

$$0.994\ ac\text{-}ft/d \cdot 70.9\ ng/L \cdot 43{,}560\ ft^2/ac \cdot 28.32\ L/ft^3 \cdot 1g\ /\ 1{,}000{,}000{,}000\ ng = 0.087\ g/d$$

For the southern two lakes, the maximum allowable daily mercury load is estimated from the dry and wet atmospheric deposition rates (Appendix E, Atmospheric Deposition) and the cumulative lake surface area for the two lakes (5.2 acres or 0.021 km^2). Dry deposition rates are fairly constant and the average daily load deposited to the southern lake system may be estimated by dividing the annual deposition rate by the average number of days per year:

$$50.0656 g / km^2 / yr \cdot 0.021 km^2 \cdot \frac{1yr}{365.25d} = 0.00288 g / d$$

The daily maximum wet deposition rate is equal to the annual rate times the fraction of precipitation that falls during the wettest month of the year divided by number of days in that month. Weather data for the Long Beach area indicate that February is typically the wettest month, receiving 24.7 percent of annual precipitation. The likely maximum wet deposition rate to the southern lakes at El Dorado Park is:

$$3.176 g / km^2 / yr \cdot 0.021 km^2 \cdot 24.7\% / 100 yr / mo \cdot \frac{1mo}{28.25d} = 0.00058 g / d$$

As no reductions in existing load are required for the southern lake system, the total maximum daily load is the sum of the daily dry and wet loads or 0.00346 g/d.

8.3.6.6 Future Growth

The El Dorado Park lakes watershed is comprised entirely of parkland. It is not likely that the watershed will be developed and it is expected to remain as open space. No load allocation has been set aside for future growth, and it is unlikely that any dischargers will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

8.4 LEAD IMPAIRMENT

The El Dorado Park lakes were listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 38 additional samples (six wet weather) between February 2009 and September 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 38 samples (Appendix G, Monitoring Data). USEPA also collected eight sediment samples between August and September 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, the El Dorado Park lakes meet lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that the El Dorado Park lakes not be identified as impaired by lead in California’s next 303(d) list.

8.5 COPPER IMPAIRMENT

The El Dorado Park lakes were listed as impaired for copper in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 38 additional samples (six wet weather) between February 2009 and September 2010 to evaluate current water quality conditions. There were two dissolved copper exceedances in 38 samples (Appendix G, Monitoring Data). USEPA also collected eight sediment samples between August and September 2010 to further evaluate lake conditions. There were four sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). In order to address the impairment for copper, on January 10, 2012 the Regional Board issued Cleanup and Abatement Order (CAO) No.R4-2012-0003 *Requiring the City of Long Beach to take remedial action to reduce copper loading to El Dorado Park Lakes pursuant to California Water Code Section 13304 in order to implement a Total Maximum Daily Load for copper*. This CAO contained all the elements of a TMDL and was approved by USEPA on March 20, 2012.

8.6 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or nonpoint source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

If necessary, these TMDLs may be revised as the result of new information (See Section 8.7 Monitoring Recommendations). The State Board is in the early stages of developing a Statewide Mercury Policy and Mercury Control Program for Reservoirs. According to CEQA scoping materials, the Policy would define an overall structure for adopting water quality objectives; general implementation requirements; and control plans for mercury impaired water bodies. The final structure of the control program could include a total maximum daily load (TMDL) for mercury in reservoirs along with an implementation plan to achieve the TMDL; or an implementation plan that does not rely on a TMDL. How this upcoming policy and program will affect implementation of this TMDL is unknown at this time.

8.6.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 8-8 and Table 8-16 for the northern lake system and Table 8-9 and Table 8-17 for the southern lake system for nutrients and mercury, respectively.

8.6.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to the supplemental water additions (Table 8-5 and Table 8-14 for the northern lake system and Table 8-6 and Table 8-15 for the southern lake system for nutrients and mercury, respectively). These mass-based waste load allocations will be implemented by the Regional Board.

8.6.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

The El Dorado Park lakes have both nutrient-related and mercury impairments. While there are some management strategies that would address both of these impairments (i.e., sediment removal BMPs), their differences warrant separate implementation and monitoring discussions. One potential source control measure that has previously been proposed by the city of Long Beach would help implement TMDLs for both impairments and is detailed below.

These lakes are currently supplied by potable and groundwater, but the city of Long Beach has proposed adoption of the following grant (City of Long Beach, 2008):

> *The project will convert six lakes in the El Dorado Regional Park and Nature Center from potable water to excess reclaimed water by the installation of nano-filtration plants at the northern-most lake in the Regional Park and in the maintenance yard adjacent to the Nature Center. The nano-filtration will provide clean water to the lakes, and allow the lakes to overflow into the connecting streambeds, thereby providing increased circulation and cleansing of the lake water. The estimated potable water savings would be 190 acre-feet per year.*

The original grant application (Watershed Conservation Authority, 2005) states the following:

> *El Dorado Park lakes Water Usage and Wetlands Restoration integrates water conservation, water quality, habitat restoration and recreational use benefits. Reclaimed water will be used to create a continuous, natural stream flow through the park lakes. The creation of a stream will restore riparian habitat. Wetland habitat will be created within a detention basin that will improve water quality and support a variety of wildlife species. Expansion of the existing Nature Center, introduction of native habitat into the regional park, and expanding environmental education enhancements will offer diverse recreational opportunities in the regional park….*
>
> *El Dorado Park lakes Water Usage and Wetlands Restoration will significantly reduce the pollution in the six lakes in the Park and Nature Center caused by insufficient water circulation and excessive levels of nitrogen in the water. This is especially important in the sensitive Nature Center lakes. It will also improve storm drain outlet flows into the San Gabriel River Estuary in order to meet water quality standards….*
>
> *Effluent from a storm drain from a 100-acre shopping center will be intercepted, filtered for trash, and cleansed in a treatment wetland before discharge into the San Gabriel River or Coyote Creek. The project also improves the water quality of the lakes through the desalination of the*

reclaimed water entering the lakes and replacing an artificially maintained constant water level with a constantly flowing water body.

Implementing changes into the source of water for the lakes will have vast impacts on water quality. The existing groundwater used to fill the northern lake has anomalously high mercury concentrations and switching to a different source of water would likely results in much lower mercury concentrations. Additionally, if filtration of other water sources provides a low nutrient water source and additional flow and circulation to the lakes, reductions in chlorophyll *a* levels should result.

8.6.3.1 Nutrient-Related Impairments

Additionally, to further address nutrient-related impairments, source reduction and pollutant removal BMPs designed to reduce sediment loading could be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to the El Dorado Park lakes. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

If fertilizer application is used in the future at El Dorado Park lakes, education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will be necessary to ensure that there is not excess nutrient loading to the lakes. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations. The NNE BATHTUB model indicated Additional Parkland Loading is present in the northern four lakes. These lakes are those most heavily frequented by bird feeders and the additional bird feces produced by bird feeding contributes to this load.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

8.6.3.2 Mercury Impairment

The primary source of mercury loading to the northern four lakes is the groundwater input. Reducing this loading is imperative to ultimately achieving the fish tissue target in the lakes. Additional source(s) of water may be required to maintain lake levels and/or treatment of the groundwater may be necessary to reduce mercury concentrations to acceptable ranges.

To reduce watershed loading, several management practices can be implemented. Dissolved loading associated with storm event runoff also contributes some mercury loading to the El Dorado Park lakes, however, these were not identified as significant sources of mercury in the El Dorado Park lakes watershed. Specifically, source reduction and pollutant removal BMPs designed to reduce sediment loading can be implemented throughout the watershed as these management practices will also reduce the mercury loading associated with sediments. Some of the sediment reduction BMPs may also result in decreased concentrations of mercury in the runoff water. BMPs that provide filtration or infiltration processes may retain dissolved mercury in the upland areas. Additionally, reducing nutrient loading to the lake and improving aeration would likely reduce methylation rates within the lake overall.

Unfortunately, sediment reduction BMPs will not mitigate mercury loading from the second largest source in the watershed, atmospheric deposition to the lake surface. Mercury available for deposition in the southwest region typically originates from both local and global sources. In the US, mercury emissions from most facilities have been reduced over the past few decades as the best available technology has improved over the years. In 2008, USEPA modeled mercury air emissions nationally as a tool for tracking airborne mercury to assist in watershed planning. The mercury emission estimates were principally based on 2001 data. The highest modeled impact in California was located in the Long Beach area and the largest single source contributor was the Long Beach South East Resource Recovery facility which combusts municipal waste to produce electricity. Since that time USEPA has promulgated regulations to reduce mercury from solid waste incinerators and the emissions from this facility and another solid waste incinerator in the city of Commerce have been significantly reduced. In addition to these regulations for solid waste combustors, USEPA is in the process of finalizing regulations for Portland Cement plants which also contribute to mercury air loading and deposition in the Los Angeles area.

8.7 MONITORING RECOMMENDATIONS

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations will indeed result in compliance with the chlorophyll *a* and mercury targets, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in mercury and algal densities and bloom frequencies.

8.7.1 Nutrient-Related Impairments

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. At El Dorado Park Lakes the only wasteload allocations are for supplemental water additions. These sources should be monitoring once a year during the summer months (the critical condition) for at minimum; ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient TMDLs for the northern four lakes of El Dorado Park lakes conclude that a 52.4 percent reduction in total phosphorus loading and a 62.2 percent reduction in total nitrogen loading are needed to maintain a summer average chlorophyll *a* concentration of 20 µg/L (note that the southern two lakes have TMDLs equal to the existing load, so no reductions are required). As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). For the supplemental water additions, the targeted concentrations may be 0.113 mg-P/L and 0.363 mg-N/L for the northern lake system, and 0.048 mg-P/L and 0.94 mg-N/L for the southern lake system, assuming flow volumes for both sources remain at existing levels (Table 8-5 and Table 8-6). Similarly, targeted concentrations of nitrogen and phosphorus in the city of Long Beach runoff to the northern four lakes may be 0.319 mg-P/L and 1.67 mg-N/L. Targeted concentrations in the parkland irrigation returns may be 0.079 mg-P/L and 2.16 mg-N/L (3.9 percent of the total irrigation volume to both lake systems is assumed to reach the lake; Appendix F, Dry Weather Loading). The targeted nitrogen concentrations for precipitation to the surfaces of the northern four lakes may be 0.082 mg-N/L. Targeted concentrations for the additional parkland loading cannot be estimated because the associated flow volumes are unknown. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

8.7.2 Mercury Impairment

To assess compliance with the mercury TMDLs, monitoring should include monitoring of largemouth bass (325-375mm in length) fish tissue (skin-off fillets) at least every three years as well as twice yearly sediment and water column sampling in each lake. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total mercury, dissolved methylmercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids; as well as the following in-lake sediment parameters: total mercury, methylmercury, total organic carbon, total solids and sulfate. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Additionally, in order to accurately calculate compliance with allocations expressed in yearly loads, monitoring should include flow estimation or monitoring as well as water quality concentration measurements. At El Dorado Park Lakes the only wasteload allocation is to supplemental water additions. This source should be monitored twice a year for at minimum: total mercury, methyl mercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids.

The mercury TMDLs for the El Dorado Park lakes concludes that a reduction in total mercury loading to the northern four lakes of 47.9 percent will result in compliance with the fish tissue target of 0.22 ppm (note that the southern two lakes have TMDLs equal to the existing load, so no reductions are required). As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 8-5 and Table 8-6 for the northern and southern lake systems, respectively), targeted concentrations of total mercury in the supplemental water additions may be 71.0 ng/L for the northern lake system and 2.84 ng/L for the southern lake system. Similarly, the targeted concentration of total mercury in the city of Long Beach runoff to the northern four lakes may be 2.72 ng/L, and the targeted concentration in the irrigation return flows may be 0.768 ng/L. For the southern two lakes, the targeted concentration of total mercury in the runoff from the city of Long Beach may be 5.22 ng/L, and the targeted concentration in the parkland irrigation return flows may be 1.47 ng/L (3.9 percent of the total irrigation volume for both lake systems is assumed to reach the lake; Appendix F, Dry Weather Loading). As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved. An in-lake water column dissolved methylmercury target of 0.081 ng/L also applies.

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9 North, Center, and Legg Lake TMDLs

Legg Lake (#CAL4053100019980917155807) is listed as impaired by ammonia, copper, lead, odor, and pH (SWRCB, 2010). (Note: trash impairment has been addressed by a previous TMDL.) This section of the TMDL report describes the nutrients impairments and the TMDLs developed to address them in North, Center, and Legg lakes (Section 9.2). Nutrient load reductions are required to achieve the chlorophyll *a* target; these reductions are also expected to alleviate ammonia, odor and pH problems. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that copper and lead are currently achieving numeric targets at North, Center, and Legg lakes; therefore, TMDLs are not included for these pollutants. Analyses are presented below for lead (Section 9.3) and copper (Section 9.4).

9.1 Environmental Setting

North, Center, and Legg lakes are located in the Los Angeles River Basin (HUC 18070105) in the Whittier Narrows Recreation Area (WNRA) (Figure 9-1 and Figure 9-2). The WNRA land is 1,283 acres leased to the County of Los Angeles Department of Parks and Recreation in 1957. Legg Lake (also called South Lake) was the first lake constructed in the 1950s (construction involved excavating below the groundwater level). Two additional lakes, Center Lake and North Lake, were constructed in 1967 and are connected to Legg Lake, depending on flow conditions. The northern most lake is North Lake (surface area of 22.9 acres, average depth of 6.8 feet, and volume of 156 ac-ft), which is fed by two storm drains, one of which can either flow into North Lake or bypass North Lake and flow directly to Mission Creek. (It is assumed that this flow primarily enters North Lake.) North Lake itself also discharges to Mission Creek. During low flow periods, Center Lake (surface area of 10.8 acres, average depth of 11.8 feet, and volume of 127 ac-ft) contributes a small amount of flow to North Lake; this lake also discharges to Mission Creek (Figure 9-3). The southernmost lake, Legg Lake (surface area of 42.9 acres, average depth of 6.8 feet, and volume of 297 ac-ft) is continuously connected to Center Lake by a channel (Valentina Cabrera-Stagno, USEPA Region IX, personal communication, July 21, 2009). Overflow from the lake system drains from Center Lake to Mission Creek. (All surface areas are estimated based on Southern California Association of Governments 2005 land use data. Volume estimates were provided by the County of Los Angeles Department of Parks and Recreation. Average depths were calculated by dividing volume by surface area.)

There are several areas associated with the WNRA and Area D is located near the lakes. Some restrooms in this area are on septic systems (Restroom #5, Restroom #8, and the Adult Crew Sub-Office; personal communication, Joyce Gibson, park superintendent, Los Angeles County Department of Parks and Recreation, December 21, 2009), while the remaining restrooms are connected to the city sewer system. Recreational uses include fishing, and the California Department of Fish and Game periodically stock the lake with trout. Swimming is prohibited in the lakes, although the locations where the groundwater wells that pump supplemental water cascade to the lakes (this applies to North Lake and Center Lake) are accessible for contact recreation (Figure 9-4). Paddle boating is allowed in North Lake and radio-controlled model boating is allowed in Legg Lake. Bird feeding may be another recreational activity, although it is currently prohibited based on park rules. Park staff, however, have indicated that bird feeding is still a very common activity for lake visitors. While it has not been observed during recent fieldwork, bird feeding is mentioned in the draft Legg Lake Management Plan, which also includes results of a one-day bird population survey that identified over 600 resident birds (County of Los Angeles, 2008). Lake managers use algaecides to control algal growth in the lakes on an as-needed basis. Additional characteristics of the watershed are summarized below.

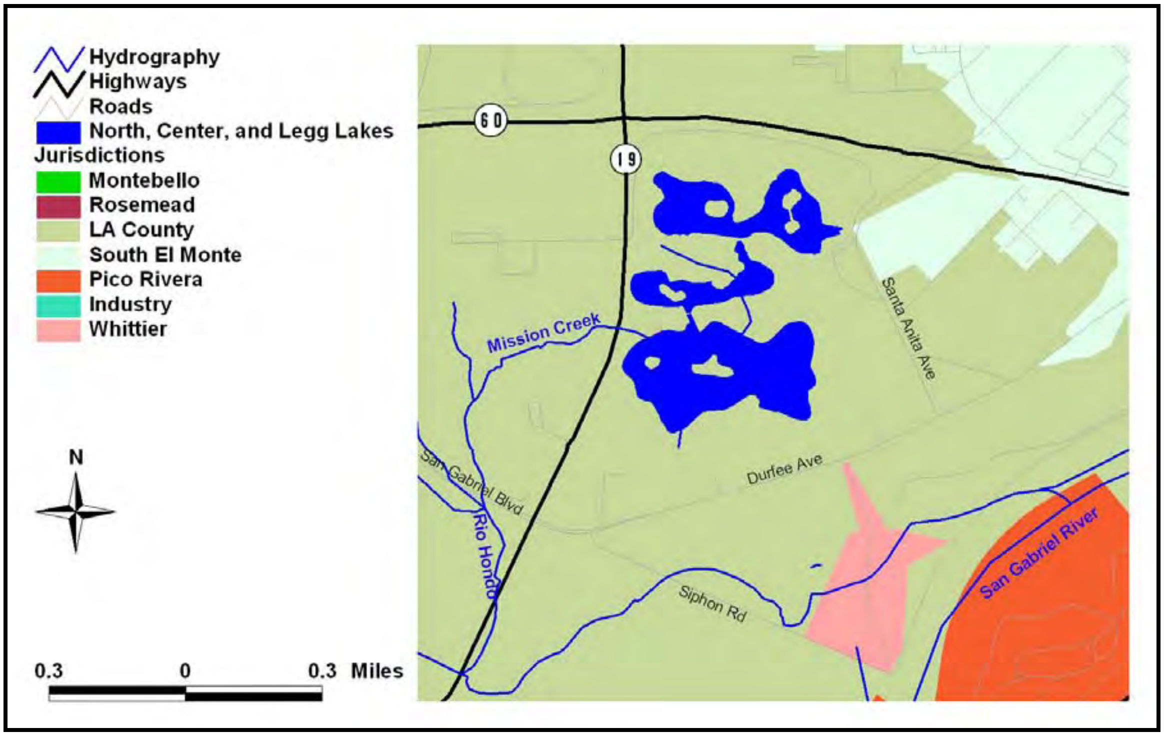

Figure 9-1. Location of North, Center, and Legg Lakes

Figure 9-2. View of Legg Lake

Figure 9-3. Center Lake Discharges to Mission Creek

Note: Groundwater is input to the North Lake and Center Lake via manmade rock cascades.

Figure 9-4. Groundwater Input to North Lake

9.1.1 Elevation, Storm Drain Networks, and Subwatershed Boundaries

The North, Center, and Legg lakes watershed (1,172 acres) ranges in elevation from 60 meters to 89 meters. Five subwatersheds comprise the drainage area to these lakes. The northwestern and northeastern subwatersheds each drain to separate storm drains that enter North Lake from the northeast side. These two subwatersheds were based on the county of Los Angeles subwatersheds. Three separate drainage areas have been delineated around the lakes to designate overland flow into each individual lake (Figure 9-5). The storm drain coverage was provided by the county of Los Angeles.

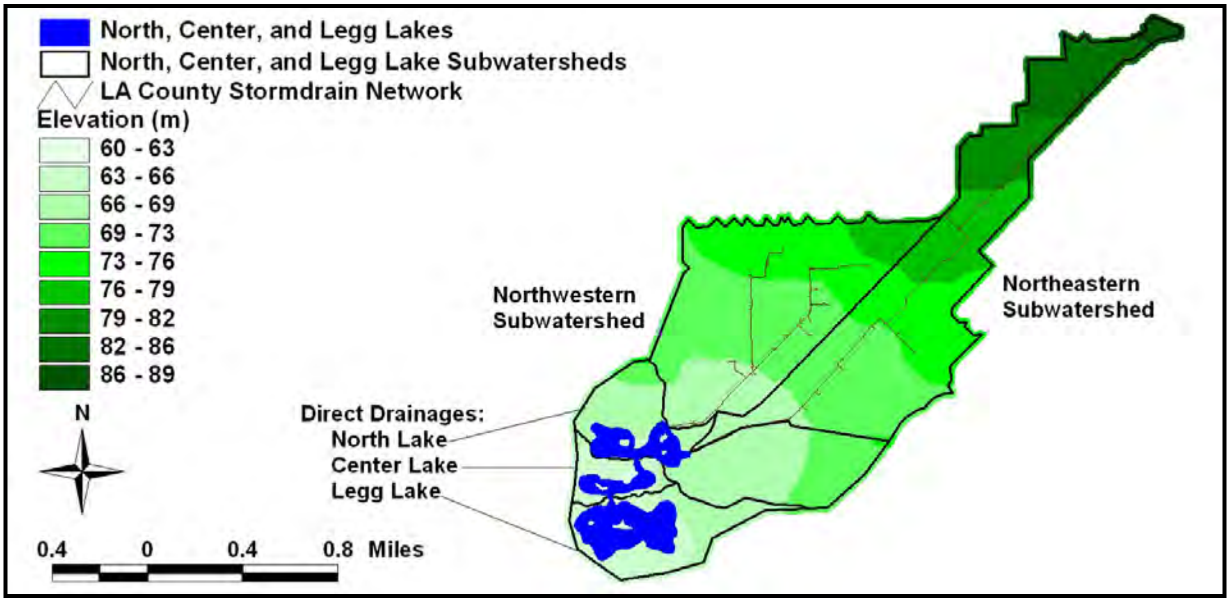

Figure 9-5. Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries for the Legg Lake System

9.1.2 MS4 Permittees

Figure 9-6 shows the MS4 stormwater permittees in the North, Center, and Legg lakes watershed. Loads generated from El Monte, South El Monte, and the county of Los Angeles in either the northwestern or northeastern subwatersheds are assigned wasteload allocations in the TMDLs because they drain to the storm drain network before discharging into the lakes. Figure 9-7 and Figure 9-8 show some of the storm drains to North Lake. Loads generated by South El Monte or the county of Los Angeles areas in the direct drainage subwatersheds are assigned load allocations. Caltrans roads in these subwatersheds are assigned wasteload allocations.

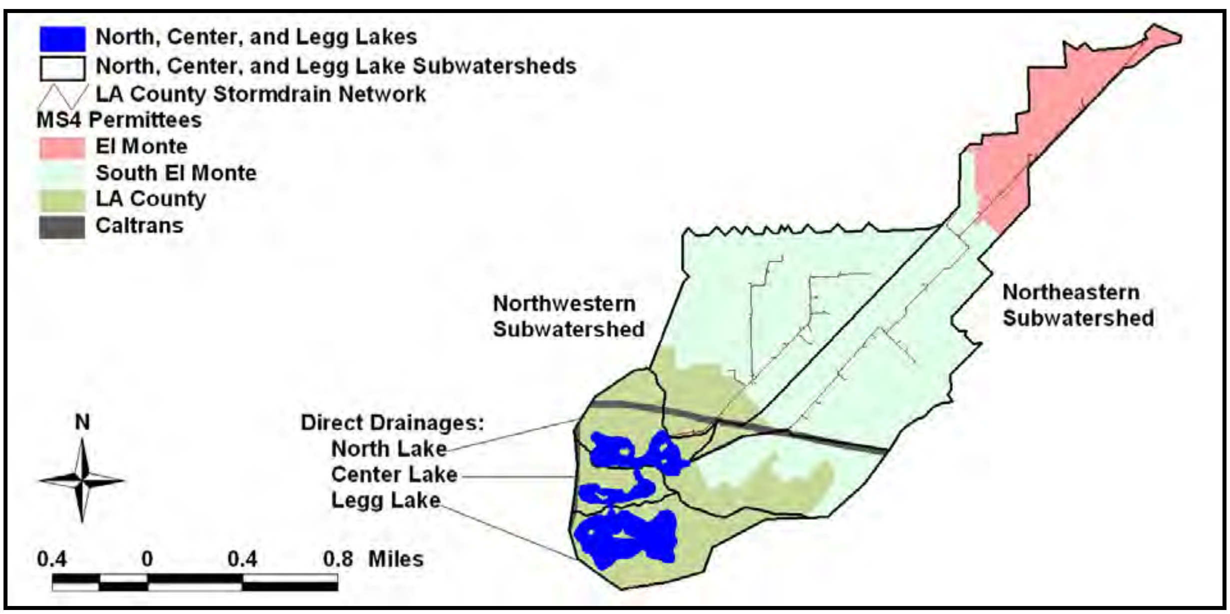

Figure 9-6. MS4 Permittees and the Storm Drain Network in the North, Center, and Legg Subwatersheds

Figure 9-7. Smaller Storm Drain to North Lake (Northwestern Subwatershed)

Note: Grates visible at the bottom discharge lake water into Mission Creek.

Figure 9-8. Largest Storm Drain to North Lake (Northeast Subwatershed)

9.1.3 Non-MS4 NPDES Dischargers

There are several additional NPDES permits (non-MS4) in the Legg Lake watershed (Table 9-1). These include five dischargers covered under a general industrial stormwater permit (see Section 3.1 for a detailed discussion of these permit types). These permits were identified by querying excel files of permits from the Regional Board website (excel files for each watershed are available from this link; www.waterboards.ca.gov/losangeles/water_issues/programs/regional_program/index.shtml#watershed; accessed on October 5, 2009). They are all in South El Monte in the northwestern subwatershed (Figure 9-9) and result in 9.27 disturbed acres. (Note: According to the permit database Vacco Industries has a disturbed area of 327 acres. Based on satellite imagery and parcel data, this area was estimated to be between 3.0 acres and 3.5 acres. Assuming the error in the database is due to a misplaced decimal point, a disturbed area of 3.27 acres was used for this facility.) Specific information is not available regarding these dischargers; therefore, they are assigned existing loads and wasteload allocations based on their area (industrial stormwater).

Table 9-1. Non-MS4 Permits in the North, Center, and Legg Lakes Watershed

Type of NPDES Permit	Number of Permits	Subwatershed	Jurisdiction	Disturbed Area
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000003)	5	Northwestern	South El Monte	9.27 acres

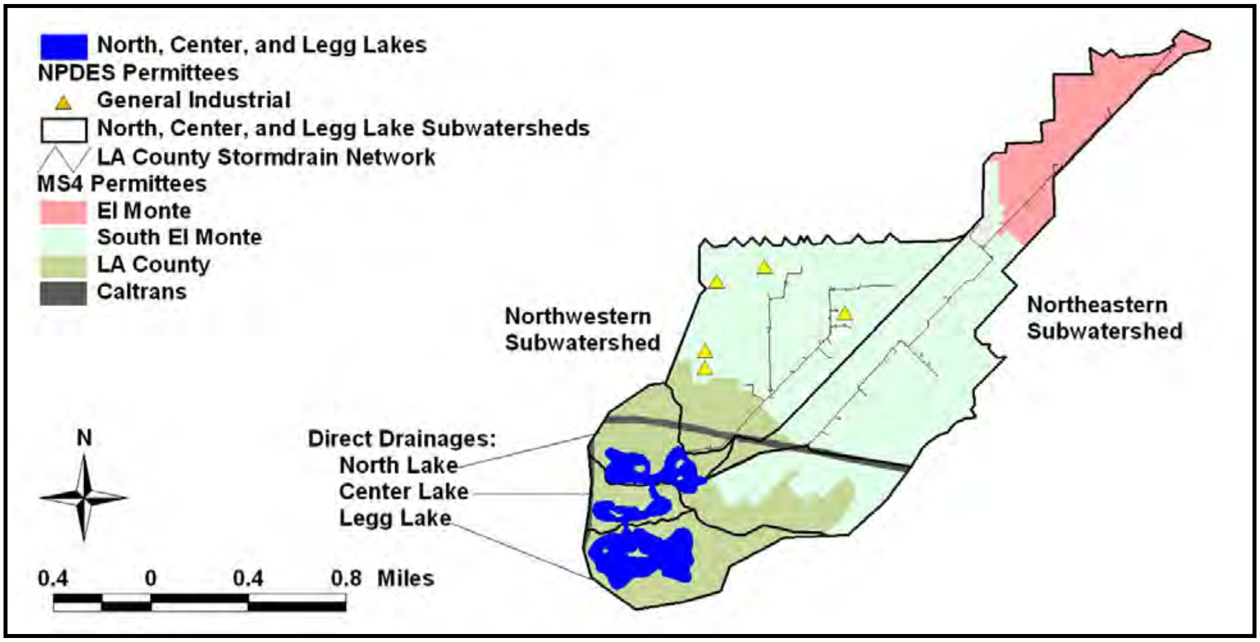

Figure 9-9. Non-MS4 Permits in the Legg Lake Subwatersheds

9.1.4 Land Uses and Soil Types

Several of the analyses for the North, Center, and Legg lakes watershed include source loading estimates obtained from the Los Angeles River Basin LSPC Model discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the Los Angeles River Basin LSPC model for these subwatersheds are shown in Figure 9-10. Tetra Tech reviewed the SCAG 2005 database and current satellite imagery to confirm the acreage of agricultural areas present in the LSPC model. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the following changes were made to maintain consistency with the SCAG 2005 land use database: in the direct drainage subwatershed to Legg Lake, approximately half of the agricultural area was reclassified as it is actually parkland and the agricultural areas assigned in the direct drainage to North Lake and northeastern subwatersheds were changed to vacant land. In addition, the agricultural area present in the northwestern subwatershed is classified by SCAG 2005 as nurseries; however, this area was reclassified to parkland as current satellite imagery shows this area to be Shiveley Park. For the purposes of estimating flows and pollutant loads to this lake system, all agricultural areas are reassigned as open space, with the exception of 1.02 acres located in the direct drainage to Legg Lake subwatershed, which were confirmed to be strawberry fields. The area classified as “other urban” in the LSPC land use categories is a high school according to SCAG 2005. Table 9-2 and Table 9-3 summarize the land use areas for the northern two subwatersheds and the direct drainage subwatersheds, respectively, by jurisdiction. These areas are combined because all of the northern watersheds are associated with WLAs and the direct drainage subwatersheds are all assigned LAs.

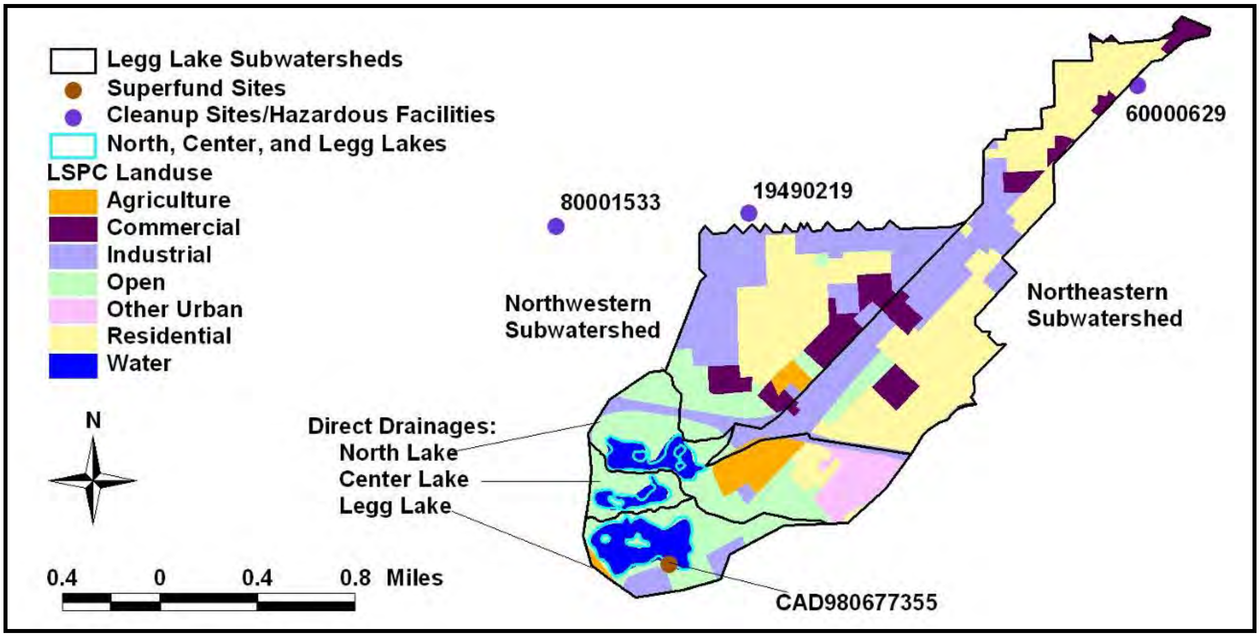

Figure 9-10. LSPC Land Use Classes for the North, Center, and Legg Lake Subwatersheds

Table 9-2. Land Use Areas (ac) Draining from the Northern Subwatersheds to North, Center, and Legg Lakes

Land Use	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	0	0	0	**0**
Commercial	23.5	58.0	11.9	0	**93.5**
Industrial/Roads	6.49	269	13.4	11.5	**300**
Open	0	29.3	44.6	0	**73.9**
Other Urban	0	0	0	0	**0**
Residential	104	267	0.271	0	**371**
Total	**134**	**623**	**70.2**	**11.5**	**838**

Table 9-3. Land Use Areas (ac) Draining from the Direct Drainage Subwatersheds to North, Center, and Legg Lakes

Land Use	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	1.04	0	**1.04**
Commercial	0	0	0	**0**
Industrial/Roads	1.78	24.1	17.6	**43.4**
Open	29.8	202	0	**232**
Other Urban	28.2	12.1	0	**40.3**
Residential	15.8	1.19	0	**17.0**
Total	**75.7**	**240**	**17.6**	**334**

There are three Resource Conservation and Recovery Act (RCRA) cleanup sites close to the Legg Lake watershed (see Table 9-4); these are located within approximately 0.6 miles of the watershed boundary (Figure 9-10). There is one Superfund site in the watershed that treats groundwater contaminated with volatile organic compounds, which discharges to Legg Lake following treatment (Figure 9-11). Most of these sites are not likely to contribute to the existing impairments at Legg Lake, except possibly the El Monte Disposal Service. Lead is listed as a potential contaminant of concern at this site; however, as described below, recent lead samples collected from Legg Lake are below the CTR criteria resulting in a finding of non-impairment. Table 9-4 summarizes the available information regarding these sites.

Note: The above treatment facility discharges under the surface of the water in Legg Lake.

Figure 9-11. Superfund Groundwater Remediation Site

Table 9-4. RCRA Cleanup and Superfund Sites Located within or near the Legg Lake Watershed

Envirostor #	Facility Name	Cleanup Status	Potential Contaminants of Concern
80001533 (CAD008246746)	Boer Graphics / Paragon Press	Inactive	Information not listed in database
19490219	El Monte Disposal Service	Certified	Lead, Benzene, Arsenic, Motor oil
60000629	Hytone Cleaners	Active	Volatile organic compounds
CAD980677355	San Gabriel Valley Area 1 Whittier Narrows Operable Unit	Active	Volatile organic compounds

Figure 9-12 shows the predominant soil identified by STATSGO in the Legg Lake subwatersheds. The soil type is identified as Urban land-Sorrento-Hanford (MUKEY 660473), a hydrologic group B soil, which has moderate infiltration rates and moderately coarse textures.

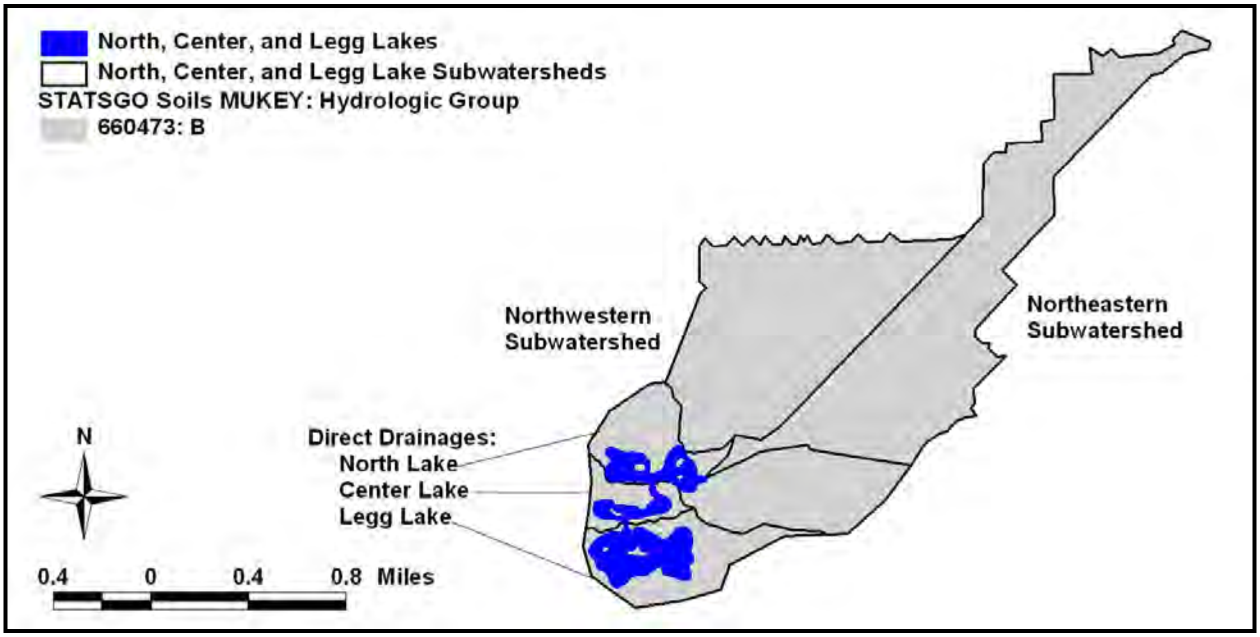

Figure 9-12. STATSGO Soil Types Present in the Legg Lake Subwatersheds

9.1.5 Additional Inputs

North, Center, and Legg lakes receive water from several additional sources; including reclaimed water, potable water, and post-treatment Superfund site discharge. Prior to May 2010 additional groundwater had been used to supplement water levels, but this input was discontinued.

An additional 1,239 ac-ft/yr of water are used to irrigate 568 acres of parkland adjacent to the Legg Lake system (6.3 percent of the total irrigation volume is assumed to reach the lake). Staff at the park indicate that approximately 10 percent of this is potable water and 90 percent is reclaimed wastewater. Irrigation with the reclaimed water source began in 2006. The usage total also includes irrigation at Norman's Nursery, which is outside the watershed of the Legg Lake System. In 2006, Norman's Nursery used approximately 6.7 percent of the reclaimed water applied at Whittier Narrows. Subtracting out the usage at Norman's Nursery leaves approximately 1,040 ac-ft of reclaimed water applied around the Legg Lake system. As previously noted, 10 percent of the irrigation water is potable water, resulting in an additional 124 ac-ft of water applied to the parkland. Some of the potable and reclaimed irrigation water applied to the parklands may reach the lakes.

The San Gabriel Valley Area 1 Whittier Narrows Operable Unit Superfund site (EPA #CAD980677355) treats contaminated groundwater from a 4 mi^2 area located in and around the North, Center, and Legg lakes watershed. There is no NPDES permit associated with this discharge. Contamination by volatile organic compounds (VOCs) was identified in local groundwater wells in the southern portion of the San Gabriel Basin in 1979. Contamination, caused by decades of improper chemical handling and disposal by hundreds of industries, resulted in high concentrations of compounds including tetrachloroethylene (PCE), trichloroethylene (TCE), 1-4 dioxane, perchlorate, and N-nitrosodimethylamine (NDMA) within groundwater wells. Remediation efforts, including containment of groundwater contamination and conveyance to and from the liquid-phase granular activated carbon groundwater treatment plant, began in September of 2000. Initial conveyance of treated groundwater from the treatment plant began in February of 2002 with discharge of this remediated groundwater to Legg Lake commencing in October of 2002. The treatment effectively removes the VOCs and has no impact on the concentrations of nutrients or metals in the treated groundwater. Continued groundwater monitoring has been completed by USEPA, and significant reductions in contaminant concentrations have been documented (USEPA, 2006). Annual average post-treatment flows from this source are approximately 2,534 ac-ft per year as measured by USEPA. The flow is discharged to the Legg and North lakes using a cascading water delivery method that had previously been used for the additional groundwater inputs prior to May 2010

9.2 NUTRIENT RELATED IMPAIRMENTS

A number of the assessed impairments for Legg Lake are associated with nutrients and eutrophication. Nutrient-related impairments for Legg Lake include ammonia, odor, and pH (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algal photosynthesis removes carbon dioxide from the water, which can lead to elevated pH in poorly buffered systems. Respiration during nighttime hours and decay of algae cause decreased dissolved oxygen (DO) concentrations. Algal blooms may also contribute to odor problems.

9.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Legg Lake include REC1, REC2, WARM, WILD, MUN, WET, GWR, and COLD. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are currently impairing the REC1, REC2, WARM, and COLD uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels, alter biology that impair the aquatic life use, and cause odor and aesthetic problems. At high enough concentrations WILD, MUN, and GWR uses could become impaired.

9.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Legg Lake. The following targets apply to the ammonia, odor, and pH impairments (see Section 2 for additional details and Table 9-5 for a summary):

- Most ammonia in fresh water is present in the ionized form of ammonium (NH_4^+). The Basin Plan expresses ammonia targets as a function of pH and temperature because it is un-ionized ammonia (NH_3) that is toxic to fish and other aquatic life. In order to assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for the Legg Lake system in these TMDLs, a median temperature of 16.0 ºC and a 95th percentile pH of 9.6 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 0.42 mg-N/L, the four-day average is 0.56 mg-N/L, and the 30-day average (chronic) target is 0.23 mg-N/L (Note: the median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target varies with the values determined during sample collection.).
- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient concentrations (e.g., nitrogen and phosphorous) in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Legg Lake; however, as

described in Tetra Tech (2006), summer (May – September) mean and annual average chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and are also consistent with supporting warm water aquatic life. The chlorophyll *a* target must be met at half of the Secchi depth during the summer (May – September) and annual averaging periods.

- The Basin Plan states that “waters shall not contain taste or odor-producing substances in concentrations that impart undesirable tastes or odors to fish flesh or other edible aquatic resources, cause nuisance, or adversely affect beneficial uses.”
- The Basin Plan states “at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L, except when natural conditions cause lesser concentrations.” In addition, the Basin Plan states, “the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges” and “the dissolved oxygen content of all surface waters designated as COLD shall not be depressed below 6 mg/L as a result of waste discharges.” The Legg Lake system has a COLD beneficial use; therefore, the COLD DO target applies. Shallow, well-mixed lakes, such as the Legg Lake system, must meet the COLD DO target in the water column from the surface to 0.3 meters above the bottom of the lake.
- The Basin Plan states that “the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge.” Shallow, well-mixed lakes, such as Legg Lake, must meet the pH target in the water column from the surface to 0.3 meters above the bottom of the lake.

Nitrogen and phosphorus target concentrations are based on simulation of allowable loads with the NNE BATHTUB model (see Section 9.2.5). Based on the calibrated model for Legg Lake, the target nutrient concentrations within the lake are

- 0.65 mg-N/L summer average (May – September) and annual average
- 0.065 mg-P/L summer average (May – September) and annual average

Table 9-5. Nutrient-Related Numeric Targets for North, Center, and Legg Lakes

Parameter	Numeric Target	Notes
Ammonia[1]	0.42 mg-N/L acute (one-hour) 0.56 mg-N/L four-day average 0.23 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 6 mg/L single sample minimum except when natural conditions cause lesser concentrations	

Parameter	Numeric Target	Notes
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.65 mg-N/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model
Total Phosphorous	0.065 mg-P/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model

[1] The median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan..

9.2.3 Summary of Monitoring Data

Water quality in Legg Lake proper has been monitored since the early 1990s. Monitoring in North and Center lakes began more recently. This section summarizes the monitoring data relevant to the nutrient impairments. Shoreline sampling is not discussed as these samples are typically not reflective of the lake as a whole. Additional details regarding monitoring are discussed in Appendix G (Monitoring Data).

Legg Lake proper was monitored in 1992 and 1993 for water quality as part of the Urban Lakes Study from the lower section of the lake on the western side. Total Kjeldahl Nitrogen (TKN) generally ranged from 0.6 mg-N/L to 1.0 mg-N/L although three samples were less than the detection limit (0.01 mg-N/L) and one outlier had a concentration of 37 mg-N/L. The majority of the ammonium samples (33 of 43) were less than the detection limit (0.01 mg-N/L); ammonium concentrations as high as 0.4 mg-N/L were observed. All nitrite samples were less than the detection limit (0.01 mg-N/L), and nitrate concentrations did not exceed 0.2 mg-N/L. Both phosphate and total phosphorus were less than the detection limit (0.01 mg-P/L) in all 43 samples. pH ranged from 8.0 to 8.9. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 2 μg/L to 27 μg/L (average of 15 μg/L). The Study reported that algae levels and macrophyte growth were not problematic.

The Regional Board's 1996 Water Quality Assessment Database does not include data for Legg Lake or its watershed. The Assessment Report does include summary information for the impairments. Ammonia was partially supporting the aquatic life use; 43 ammonium samples were collected with concentrations ranging from non-detect to 0.35 mg-N/L. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. pH was listed as partially supporting the aquatic life use and not supporting the secondary drinking water use. Eighty-four measurements of pH ranged from 7.6 to 8.9. Odor was listed as not supporting the contact and non-contact recreation uses. The Legg Lake system was sampled multiple times during May, June, and July 2007 (data provided by the county of Los Angeles). Nineteen of 21 mid-lake samples of ammonia had concentrations ranging from less than the detection limit of 0.01 mg-N/L to 0.36 mg-N/L; two samples had ammonia concentrations of 0.51 mg-N/L and 0.53 mg-N/L (both were collected from Center Lake in May). None of these samples exceeded the acute or chronic ammonia criteria based on the associated pH and temperature measurements. Nitrate concentrations ranged from less than the detection limit of 0.02 mg-N/L to 0.59 mg-N/L. Orthophosphate ranged from less than the detection limits (either 0.01 mg-P/L or

0.02 mg-P/L, depending on the sampling event) to 0.07 mg-P/L. Dissolved oxygen concentrations for these samples ranged from 7.7 mg/L to 12.2 mg/L; pH ranged from 7.1 to 8.2.

North, Center, and Legg lakes were sampled by the USEPA and Regional Board on July 14, 2009. Ammonia, nitrite, nitrate, and orthophosphate samples were less than the detection limits (0.03 mg-N/L, 0.01 mg-N/L, 0.01 mg-N/L, and 0.0075 mg-P/L, respectively) in all three lakes. TKN ranged from 1.4 mg-N/L to 1.7 mg-N/L. Total phosphorus ranged from 0.046 mg-P/L to 0.089 mg-P/L. Chlorophyll *a* in the three lakes ranged from 37.4 μg/L to 93.4 μg/L. pH measurements ranged from 7.7 to 9.1 in the three lakes. DO ranged from 6.7 mg/L to 13.6 mg/L over the first 2 meters of depth from the surface. Measurements taken from 2.5 meters to 2.8 meters (Center Lake only) ranged from 1.7 mg/L to 1.9 mg/L.

USEPA sampled North, Center, and Legg lakes on June 8, August 11, and September 29, 2010 (see Appendix G for monitoring data). Secchi depth ranged from 0.5 m to 1.27 m. In-lake samples of TKN ranged from 0.57 to 1.4 mg-N/L. Ammonia samples ranged from 0.03 to 0.082 mg-N/L. Nitrate-nitrite concentrations were below the detection limit of 0.015 mg-N/L during the June event for all stations and the September events at all Legg 9 and 10; nitrate-nitrite of 0.059 to 0.081 mg-N/L was observed at Legg 8 in September. During the August and September events, nitrate ranged from below the detection limit of 0.05 mg-N/L to 0.29 mg-N/L, and nitrite samples were below detection limits of 0.25 mg-N/L. All 2010 orthophosphate measurements were below the detection limit of 0.5 mg-P/L; total phosphorus concentrations ranged from 0.02 mg-P/L to 0.06 mg-P/L. Chlorophyll *a* concentrations ranged from 11 μg/L to 44 μg/L. The August chlorophyll *a* data represent estimated values as the samples were held past the holding times. The September sample was split and half was processed within the standard holding time while half was held longer than the holding time and processed at the same relative time as the August sample had been processed. The ratio of the split sample was applied to the August sample to generate an estimated chlorophyll *a* value, had that sample been processed promptly. According to depth-profile measurements, pH ranged from 7.3 to 13.2 in the three lakes. DO ranged from 3.4 mg/L to 11.3 mg/L over the first 2 meters of depth from the surface. Measurements taken from 2.2 meters to 2.7 meters ranged from 1.2 mg/L to 6.6 mg/L (Center Lake).

In summary, exceedances of the allowable range of pH have been measured during historic and recent monitoring events. DO concentrations are typically above 6 mg/L throughout the water column although measurements near the bottom of Center Lake during one sampling event have been observed at less than 2 mg/L. No odors were observed during the recent sampling events by USEPA and/or the Regional Board. Chlorophyll *a* concentrations seem to have increased dramatically relative to conditions observed in the early 1990s. Shoreline sampling conducted in February 2009 by the Regional Board had chlorophyll *a* concentrations ranging from 26.7 μg/L to 115 μg/L. Although these samples were not used for calibration of the NNE BATHTUB model (Section 9.2.5), they do provide further indication of elevated algae levels under current conditions. The nutrient TMDLs for North, Center, and Legg lakes presented in Section 9.2.6 account for summer season critical conditions by assessing loading rates consistent with meeting the summer chlorophyll *a* target of 20 μg/L. These reductions in nutrient loading are expected to alleviate pH, odor, and DO problems associated with excessive nutrient loading and eutrophication.

9.2.3.1 Summary of Ammonia Non-Impairment

Legg Lake was listed as impaired for ammonia in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 50 additional samples between May 2007 and September 2010 to evaluate current water quality conditions. There was one ammonia exceedance in 50 samples (Appendix G, Monitoring Data). Therefore, Legg Lake meets ammonia water quality standards and USEPA concludes that preparing a TMDL for ammonia is unwarranted at this time. USEPA recommends that Legg Lake not be identified as impaired for ammonia in California's next 303(d) listing.

9.2.4 Source Assessment

The source assessment for the Legg Lake system includes load estimates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading) including irrigation (6.3 percent of the total irrigation volume is assumed to reach the lake), groundwater used for supplemental water additions to maintain lake levels (Appendix F, Dry Weather Loading), discharge of treated groundwater from the Superfund site (Appendix F, Dry Weather Loading), and atmospheric deposition (Appendix E, Atmospheric Deposition). Table 9-6 summarizes the existing loads from sources in the Legg Lake watershed. The largest contributor of total nitrogen loading is the Superfund discharge (51.7 percent). The city of South El Monte contributes the majority of the total phosphorus load (56.6 percent).

Table 9-6. Summary of Average Annual Flows and Nutrient Loading to the Legg Lake System

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Direct to Center Lake	Caltrans	State Highway Stormwater[1]	2.92	4.6 (0.2)	36.1 (0.2)
Direct to Center Lake	County of Los Angeles	Runoff	1.69	0.5 (<0.1)	14.7 (0.1)
Direct to Legg Lake	Caltrans	State Highway Stormwater[1]	0.75	1.2 (0.1)	9.3 (<0.1)
Direct to Legg Lake	County of Los Angeles	Runoff	19.4	26.0 (1.4)	228.2 (1.0)
Direct to North Lake	Caltrans	State Highway Stormwater[1]	12.1	19.1 (1.0)	149.5 (0.6)
Direct to North Lake	County of Los Angeles	Runoff	20.3	26.6 (1.4)	226.0 (0.9)
Direct to North Lake	South El Monte	Runoff	31.0	55.1 (2.9)	369.3 (1.5)
Northwestern	Caltrans	State Highway Stormwater[1]	5.91	9.4 (0.5)	68.3 (0.3)
Northwestern	County of Los Angeles	MS4 Stormwater[1]	33.5	53.6 (2.8)	346.8 (1.5)
Northwestern	South El Monte[2]	MS4 Stormwater[1]	308	526.3 (27.6)	3,500.2 (14.7)
Northwestern	General Industrial Stormwater Permittees (in the city of South El Monte)	General Industrial Stormwater[1]	3.63	5.8 (0.3)	42.0 (0.2)
Northeastern	Caltrans	State Highway Stormwater[1]	6.87	10.9 (0.6)	79.4 (0.3)
Northeastern	El Monte	MS4 Stormwater[1]	122	226.6 (11.9)	1,377.0 (5.8)
Northeastern	County of Los Angeles	MS4 Stormwater[1]	8.18	12.8 (0.7)	91.4 (0.4)
Northeastern	South El Monte	MS4 Stormwater[1]	287	498.7 (26.1)	3,253.5 (13.6)

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Direct to Legg Lake	Whittier Narrows Operable Unit Groundwater Treatment Plant	Treated Groundwater from Superfund Site	2,534	172.3 (9.0)	12,355.2 (51.7)
All Direct Drainage Subwatersheds	County of Los Angeles	Parkland Irrigation	72.9	258.3 (13.5)	1,685.2 (7.1)
Lake Surface		Atmospheric Deposition[3]	105	NA	56.3 (0.2)
Total			**3,471**	**1,908**	**23,888**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of South El Monte in the northwestern subwatershed is 317 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of South El Monte. The disturbed area associated with general construction and general industrial stormwater permittees (9.27 acres) was subtracted out of the appropriate city area and allocated to these permits.

[3] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

9.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on the Legg Lake system, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires morphometric data for the simulation of chlorophyll *a* during the summer. For the Legg Lake system, the three linked segments were simulated as one aggregate waterbody because 1) there are not enough water quality data to calibrate each segment separately and 2) simulation of cumulative loading and morphometry was needed to calibrate the model within recommended guidelines (Walker, 1987). For the system as a whole, the surface area is 76.6 acres, the average depth is 7.6 ft, and the cumulative volume is 580 ac-ft. Based on the phosphorus turnover ratio for this lake (Walker, 1987), the summer averaging period is appropriate (i.e., loads delivered from May through September are input to the model rather than annual loads).

The NNE BATHTUB Tool was set up to match the three 2010 summer sampling events. The August sampling event yielded only an estimated chlorophyll *a* value, however, it was used in generating a seasonal average for the model. Historic data from the 1990s are available, however they do not represent current conditions for the lake (reclaimed water used for irrigation, discharge of treated groundwater from a Superfund site, and higher observed chlorophyll *a* concentrations). July 2009 data do not reflect the change in flow from the Superfund site and discontinuation of the additional groundwater input. All samples collected during the 2010 sampling were collected at one-half of the Secchi depth. To predict the average observed total phosphorus concentration over this depth (0.041 mg-P/L), the calibration factors on the net phosphorus sedimentation rate would need to be set higher than the recommended value of 2. The phosphorus calibration factor was set at 2, which resulted in a predicted concentration of 0.06 mg-P/L, which is within the observed range for the lakes and provides a conservative estimate of the required total phosphorus load reduction. To predict the average observed total nitrogen concentration over one-half of the Secchi depth (1.08 mg-N/L), the calibration factor on the net nitrogen sedimentation was set to 2.46, which is within the recommended range for nitrogen.

To simulate the average observed chlorophyll *a* concentration, the calibration factor on concentration was set to 0.97 for a predicted concentration of 26.7 µg/L. If subsequent data are collected that will allow for calibration of the NNE BATHTUB model, then these TMDLs may be revisited. For now, this preliminary model is being used to determine the load reductions needed to attain the chlorophyll *a* target concentration, based on the best available information.

9.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Following calibration of the NNE BATHTUB Tool (Section 9.2.5), the allowable loading combinations of nitrogen and phosphorus were calculated using Visual Basic's GoalSeek function (Appendix A, Nutrient TMDL Development). The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus for the Legg Lake system are

- 0.65 mg-N/L summer average (May – September) and annual average
- 0.065 mg-P/L summer average (May – September) and annual average

For the Legg Lake system, the loading capacity for total nitrogen is 11,379 lb-N/yr. The loading capacity for phosphorus was set to the existing load of 1,908 lb-P/yr since the existing average observed concentration is meeting the target. These loading capacities can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load (divided among WLAs and LAs) is 42.9 percent of the existing load of 23,888 lb-N/yr, or 10,241 lb-N/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. WLAs and LAs are developed assuming equal percent load reductions in all sources. The resulting TMDL equation for total nitrogen is then:

11,379 lb-N/yr = 9,135 lb-N/yr + 1,106 lb-N/yr + 1,138 lb-N/yr

For total phosphorus, the allocatable load is equal to the existing load and is divided among WLAs and LAs. The resulting TMDL equation for total phosphorous is then:

1,908 lb-P/yr = 1,541 lb-P/yr + 367 lb-P/yr + 0 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Total phosphorus allocations are set to existing loads. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined for the lake system based on simulation of allowable loads with the NNE BATHTUB model (see Section 9.2.5). These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

9.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs and alternative WLAs for total phosphorous and total nitrogen. The alternative WLAs will be effective and supersede the WLAs in Table 9-7 if the conditions described in Section **Error! Reference source not found.** or in Section 9.2.6.1.2 are met.

Under any of the wasteload allocation schemes responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on the Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

9.2.6.1.1 Wasteload Allocations

The northwestern and northeastern subwatersheds drain to a series of storm drains prior to discharging to the Legg Lake system. Therefore, all loads associated with these drainage areas are assigned WLAs. The loads attributed to the Caltrans areas in the direct drainage subwatersheds also receive WLAs along with facilities that operate under a general industrial stormwater permit. WLAs are also assigned to the Whittier Narrows Operable Unit Groundwater Treatment Plant. Relevant permit numbers are

- County of Los Angeles (including the cities of El Monte and South El Monte): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Each WLA must be met at the point of discharge. Total phosphorus WLAs represent a 0 percent reduction in existing loading, and total nitrogen WLAs represent an 57.1 percent reduction in existing loading (Table 9-7). As noted in Table 9-7 below, the concentration-based WLAs will be used to evaluate compliance with the allocations for the current discharges authorized by the general industrial stormwater permit and the construction stormwater permit and any future discharges in the watershed authorized by the general industrial and construction stormwater permits.

Table 9-7. Wasteload Allocations for Nutrient Loading to the Legg Lake System

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus[4] (lb-P/yr)	Total Nitrogen[4] (lb-N/yr)
Direct to Center Lake	Caltrans	State Highway Stormwater[1]	2.92	4.6	15.5
Direct to Legg Lake	Caltrans	State Highway Stormwater[1]	0.75	1.2	4.0
Direct to North Lake	Caltrans	State Highway Stormwater[1]	12.1	19.1	64.1
Northwestern	Caltrans	State Highway Stormwater[1]	5.91	9.4	29.3
Northwestern	County of Los Angeles	MS4 Stormwater[1]	33.5	53.6	148.7
Northwestern	South El Monte[2]	MS4 Stormwater[1]	308	526.3	1,500.6
Northwestern	General Industrial Stormwater Permittees (in the city of South El Monte)	General Industrial Stormwater[1]	3.63	5.8 (0.64 mg-P/L)[3]	18.0 (1.8 mg-N/L)[3]
Northeastern	Caltrans	State Highway Stormwater[1]	6.87	10.9	34.0
Northeastern	El Monte	MS4 Stormwater[1]	122	226.6	590.3

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus[4] (lb-P/yr)	Total Nitrogen[4] (lb-N/yr)
Northeastern	County of Los Angeles	MS4 Stormwater[1]	8.18	12.8	39.2
Northeastern	South El Monte	MS4 Stormwater[1]	287	498.7	1,394.8
Direct to Legg Lake	Whittier Narrows Operable Unit Groundwater Treatment Plant	Treated Groundwater from Superfund Site	2,534	172.3	5,296.8
Total			**3,325**	**1,541**	**9,135**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of South El Monte in the northwestern subwatershed is 317 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of South El Monte. The disturbed area associated with general construction and general industrial stormwater permittees (9.27 acres) was subtracted out of the appropriate city area and allocated to these permits. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations (see footnote #3).

[3] For these responsible jurisdictions, the concentration-based WLA will be used to evaluate compliance.

[4] Each wasteload allocation must be met at the point of discharge.

9.2.6.1.2 Alternative "Approved Lake Management Plan Wasteload Allocations"

Concentration-based WLAs not exceeding the concentrations listed in Table 9-8 are effective and supersede corresponding WLAs for a responsible jurisdiction in Table 9-7 if:

1. The responsible jurisdiction requests that concentration-based wasteload allocations not to exceed the concentrations established in Table 9-8 apply to it;
2. The responsible jurisdiction provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 9-5. Responsible jurisdictions may work together to develop, submit and implement the Lake Management Plan. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The responsible jurisdiction may use monitoring data and modeling to show that the water quality criteria, targets and requested WLAs will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based wasteload allocations for total nitrogen and total phosphorus. These wasteload allocations are not to exceed the concentrations in Table 9-8 as a summer average (May-September) and annual average, and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

The concentration-based WLAs must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 9-8. Alternative Wasteload Allocations of Phosphorus and Nitrogen in the Legg Lake System if an Approved Lake Management Plan Exists

Subwatershed	Responsible Jurisdiction	Input	Maximum Allowable Wasteload Allocation Total Phosphorus[4] (mg-P/L)	Maximum Allowable Wasteload Allocation Total Nitrogen[4] (mg-N/L)
Direct to Center Lake	Caltrans	State Highway Stormwater[1]	0.1	1.0
Direct to Legg Lake	Caltrans	State Highway Stormwater[1]	0.1	1.0
Direct to North Lake	Caltrans	State Highway Stormwater[1]	0.1	1.0
Northwestern	Caltrans	State Highway Stormwater[1]	0.1	1.0
Northwestern	County of Los Angeles	MS4 Stormwater[1]	0.1	1.0
Northwestern	South El Monte[2]	MS4 Stormwater[1]	0.1	1.0
Northwestern	General Industrial Stormwater Permittees (in the city of South El Monte)[3]	General Industrial Stormwater[1]	0.1	1.0
Northeastern	Caltrans	State Highway Stormwater[1]	0.1	1.0
Northeastern	El Monte	MS4 Stormwater[1]	0.1	1.0
Northeastern	County of Los Angeles	MS4 Stormwater[1]	0.1	1.0
Northeastern	South El Monte	MS4 Stormwater[1]	0.1	1.0
Direct to Legg Lake	Whittier Narrows Operable Unit Groundwater Treatment Plant and County of Los Angeles[5]	Treated Groundwater from Superfund Site and Supplemental Water Additions	0.1	1.0

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of South El Monte. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations (see footnote #3).

[3] For these responsible jurisdictions, the concentration-based WLA will be used to evaluate compliance.

[4] The concentration-based wasteload allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

[5] Currently the treatment plant supplements lake water levels entirely but in the past there has been a combination of County and treatment plant water used for this purpose. This allocation is given to the County of Los Angeles and the treatment plant jointly since in the future the County may resume supplemental water additions to the lakes.

9.2.6.2 Load Allocations

These TMDLs establish load allocations (LAs) and alternative LAs for total phosphorous and total nitrogen. The alternative LAs will be effective and supersede the LAs listed in Table 9-9 if the conditions described in Section 9.2.6.2.2 are met.

9.2.6.2.1 Load Allocations

Loads associated with the non-Caltrans areas in the direct drainage subwatersheds are assigned load allocations (LAs). Total phosphorus LAs represent a 0 percent reduction in existing loading, and total nitrogen LAs represent an 57.1 percent reduction in existing loading. LAs are provided for each responsible jurisdiction and input. These loading values (in pounds per year) represent the TMDLs load allocations (Table 9-9).

Table 9-9. Load Allocations for Nutrient Loading to the Legg Lake System

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft/yr)	Total Phosphorus[1] (lb-P/yr)	Total Nitrogen[1] (lb-N/yr)
Direct to Center Lake	County of Los Angeles	Runoff	1.69	0.5	6.3
Direct to Legg Lake	County of Los Angeles	Runoff	19.4	26.0	97.8
Direct to North Lake	County of Los Angeles	Runoff	20.3	26.6	96.9
Direct to North Lake	South El Monte	Runoff	31.0	55.1	158.3
All Direct Drainage Subwatersheds	County of Los Angeles	Parkland Irrigation	72.9	258.3	722.5
Lake Surface		Atmospheric deposition[2]	105	0.00	24.1
Total			**250**	**367**	**1,106**

[1] Each load allocation must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

9.2.6.2.2 Alternative "Approved Lake Management Plan Load Allocations"

Concentration-based load allocations not exceeding the concentrations listed in Table 9-10 are effective and supersede corresponding load allocations for the responsible jurisdictions in Table 9-9 if:

1. The responsible jurisdictions request that concentration-based load allocations not to exceed the concentrations established in Table 9-10 apply to it;
2. The responsible jurisdictions provide to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; and the chlorophyll *a* targets listed in Table 9-5. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The responsible jurisdictions may use monitoring data and modeling to show that the water quality criteria, targets and requested load allocations will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based load allocations for total nitrogen and total phosphorus. These load allocations are not to exceed the concentrations in Table 9-10 as a summer average (May-September) and annual average; and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based LA must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

Table 9-10. Alternative Load Allocations of Nutrient Loading to the Legg Lake System if an Approved Lake Management Plan Exists

Subwatershed	Responsible Jurisdiction	Input	Maximum Allowable Load Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Load Allocation Total Nitrogen[1] (mg-N/L)
Direct to Center Lake	County of Los Angeles	Runoff	0.1	1.0
Direct to Legg Lake	County of Los Angeles	Runoff	0.1	1.0
Direct to North Lake	County of Los Angeles	Runoff	0.1	1.0
Direct to North Lake	South El Monte	Runoff	0.1	1.0
All Direct Drainage Subwatersheds	County of Los Angeles	Parkland Irrigation	0.1	1.0

[1] Each concentration-based load allocation must be met in the lake. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met, then the total phosphorus and total nitrogen allocations are considered attained.

9.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. To account for the uncertainties concerning the relationship between nutrient loading and the resultant in-lake chlorophyll *a* an explicit MOS is included in these TMDLs. This explicit MOS is set at 10 percent of the loading capacity for total phosphorus and total nitrogen.

9.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are expected to alleviate any pH and odor problems associated with excessive nutrient loading and eutrophication. These TMDLs therefore protect for critical conditions.

9.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of phosphorus loading to the Legg Lake system occurs during wet weather events that deliver pollutant loads from the surrounding watershed, the daily maximum allowable load of phosphorus is calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) to the system multiplied by the allowable concentrations consistent with achieving the long-term loading targets. The majority of the nitrogen load results from the discharge of treated Superfund water. Little variability in daily discharge flowrate is expected, so the maximum daily nitrogen load from this source is calculated by dividing the annual load by 365 days per year. The second highest source of nitrogen loading is wet weather runoff. Because the treated groundwater from the Superfund site likely continues at the same discharge rate during dry and wet weather, daily loads from both the Superfund discharge and wet weather events will be accounted for in the estimation of nitrogen and phosphorus daily maximum loads. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the watershed. USGS Station 11102000, Mission Creek near Montebello, CA, was selected as a surrogate for flow determination. This gage is downstream of where the Legg Lake system discharges to Mission Creek. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for Mission Creek (30.2 cfs) (Wolock, 2003). To estimate the peak flow to the Legg Lake system, the 99th percentile flow for Mission Creek was scaled down by the ratio of drainage areas (1,172 acres/2,662 acres; Legg Lake watershed area/Mission Creek watershed area at the gage). The resulting peak daily flow estimate for the Legg Lake system is 13.3 cfs.

The average allowable concentrations of phosphorus and nitrogen were calculated from the allowable loads (1,908 lb-P/yr and 10,241 lb-N/yr, respectively; sum of WLAs and LAs) divided by the total volume reaching the lake (3,471 ac-ft). Multiplying the average allowable concentrations (0.20 mg-P/L for phosphorus and 1.09 mg-N/L for nitrogen) by the 99th percentile peak daily flow (13.3 cfs) yields the daily maximum load associated with wet weather runoff. The wet weather runoff daily maximum allowable loads of phosphorus and nitrogen for the Legg Lake system are 73.74 lb-P/d and 395.8 lb-N/d, respectively. These loads are associated with the MS4 stormwater permittees. The maximum daily loads for the treated groundwater from the Superfund site were calculated by dividing the annual allowable loads (Table 9-7) by 365 days, resulting in 0.47 lb-P/d and 14.5 lb-N/d. Combined, these two sources yield total maximum daily loads for phosphorous and nitrogen of 74.2 lb-P/d and 410 lb-N/d, respectively. As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

9.2.6.6 Future Growth

Areas in the northwestern and northeastern subwatersheds are nearly fully developed and most of the undeveloped land in the direct drainage subwatersheds has been set aside as parkland. If additional development occurs in this watershed, best management practices (BMPs) will be required such that loading rates are consistent with the allocations established by these TMDLs. Therefore, no load allocation has been set aside for future growth. It is unlikely that any additional dischargers of significant nutrient loading will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

9.3 LEAD IMPAIRMENT

Legg Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 45 additional samples (18 wet weather) between February 2009 and September 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 45 samples (Appendix G, Monitoring Data). USEPA also collected three sediment samples during August 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Legg Lake meets lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Legg Lake not be identified as impaired by lead in California's next 303(d) list.

9.4 COPPER IMPAIRMENT

Legg Lake was listed as impaired for copper in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 45 additional samples (18 wet weather) between February 2009 and September 2010 to evaluate current water quality conditions. There were zero dissolved copper exceedances in 45 samples (Appendix G, Monitoring Data). USEPA also collected three sediment samples during August 2010 to further evaluate lake conditions. There were zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Legg Lake meets copper water quality standards, and USEPA concludes that preparing a TMDL for copper is unwarranted at this time. USEPA recommends that Legg Lake not be identified as impaired by copper in California's next 303(d) list.

9.5 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize includes bacteria, nutrients, trash, metals and sediment. More information about this prioritization tool is available at: www.labmpmethod.org.

If necessary, these TMDLs may be revised as the result of new information (See Section 9.6 Monitoring Recommendations).

9.5.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy, and the Conditional Waiver for Discharges from Irrigated Lands, adopted by the Los Angeles Regional Water Quality Control Board on November 3, 2005. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 9-9.

9.5.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to MS4, General Industrial, and Caltrans Stormwater permits as well as supplemental water additions and the Whittier Narrows Operable Unit Groundwater Treatment Plant (Table 9-7 for Standard and Table 9-10 for Alternative Allocations). The mass-based wasteload allocations will be incorporated into the Caltrans and Los Angeles County MS4 permits. The concentration-based wasteload allocations will be incorporated into the General Industrial Stormwater permit. Wasteload allocations for Whittier Narrows Operable Unit Groundwater Treatment Plant and supplemental water additions will be implemented by the Regional Board.

9.5.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

The draft Legg Lake Management Plan identifies ongoing lake management activities that may impact existing impairments. These activities include the addition of beneficial bacteria to control excessive ammonia from waterfowl feces and to reduce aquatic weeds through the digestion of excessive nutrients in the sediment, signs prohibiting the feeding of waterfowl, and trash and debris removal (County of Los Angeles, 2008). The review of ammonia data did not indicate ammonia to be a problem; however, the reduction of excess bird populations due to bird feeding will reduce nutrient loading to the lake. Additionally, the plan recommends installing duck food dispensing machines and enforcing waterfowl feeding ordinances. These two practices would likely significantly reduce the additional fecal loading to the lake while allowing for bird feeding at the lake. The Legg Lake Management Plan also recommends the installation of bottom laid aeration and dredging to increase circulation and aeration. These activities would likely improve water quality by increasing circulation as well as reducing internal loading from lake sediments. Harvesting of weeds will also remove nutrients from the lake system but can cause repeated disturbance to the aquatic biota. Any ongoing nutrient control efforts should be continued and supplemented with other BMPs or management activities to fully address the existing impairments.

For example, source reduction and pollutant removal BMPs designed to reduce sediment loading could be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to Legg Lake. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

Education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

9.6 MONITORING RECOMMENDATIONS

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations will indeed result in compliance with the chlorophyll *a* target, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies.

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. At Legg Lake wasteload allocations are assigned to supplemental water additions and the Whittier Narrows Operable Unit Groundwater Treatment Plant. These sources should be monitored once a year during the summer months (critical conditions) for at minimum, ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

Wasteload allocations are assigned to stormwater inputs from various subwatersheds. These sources should be measured near the point where they enter the lakes twice a year for at minimum: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids. The nutrient TMDLs for Legg Lake conclude that a 0 percent reduction in total phosphorus loading and a 57.1 percent reduction in total nitrogen loading are needed to maintain a summer average chlorophyll *a* concentration of 20 µg/L. As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 9-6), target concentrations may be 0.65 mg-P/L and 1.78 mg-N/L at the outlets of the northern subwatersheds, 1.91 mg-N/L and 0.58 mg-P/L for Caltrans areas, and 0.77 mg-N/L and 0.03 mg-P/L from the groundwater discharge from the Whittier Narrows Operable Unit Groundwater Treatment Plant discharge. Similarly, the targeted concentrations of total phosphorus and total nitrogen in runoff from the direct drainage subwatersheds may be 0.55 mg-P/L and 1.83 mg-N/L; targeted concentrations in the irrigation return flows to the lake may be 1.3 mg-P/L and 3.6 mg-N/L (6.3 percent of the total irrigation volume is assumed to reach the lake). As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

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10 Puddingstone Reservoir TMDLs

Puddingstone Reservoir (#CAL4055200019980918113803) is impaired by organic enrichment/low dissolved oxygen, chlordane, DDT, mercury, and PCBs (SWRCB, 2010). In addition a dieldrin impairment has been identified by new data analyses since the 2008-2010 303(d) list data cut off. This section of the TMDL report describes the impairments and the TMDLs developed to address them: nutrients (see Section 10.2), mercury (Section 10.3) and organochlorine (OC) pesticides and PCBs (Section 10.4 through Section 10.7). Nutrient load reductions are required to achieve the chlorophyll *a* target; these reductions are also expected to alleviate DO problems.

10.1 Environmental Setting

Puddingstone Reservoir is located in the San Gabriel River Basin (HUC 18070106) in Bonelli Regional Park (Figure 10-1 and Figure 10-2). The park is located in the county of Los Angeles, immediately surrounded by the cities of San Dimas and Pomona. Located in a flood control basin, the dam was built in 1929 and the area surrounding the reservoir was converted to a park in 1972. Live Oak Wash (Figure 10-3) is the major inflow to the reservoir, which discharges to Walnut Creek. The reservoir has a surface area of 252 acres (based on Southern California Association of Governments [SCAG] 2005 land use), a total volume of 6,200 acre-feet (based on Los Angeles County Department of Public Works volume estimates from 2000 and 2001), and an average depth of 24.6 feet (volume divided by surface area). Recreational uses include swimming, jet skiing, boating, and fishing. According to the California Department of Fish and Game (2009), the reservoir is periodically stocked with trout. Bird feeding may be another recreational activity at Puddingstone Reservoir; however, it has not been observed during recent fieldwork. The areas immediately surrounding the lake receive many visitors as they include a water theme park, equestrian facilities, golf course, and a lakeside RV park. Restrooms on the park grounds are connected to the city sewer system. There is no known use of algaecide in this lake. Additional characteristics of the watershed are summarized below.

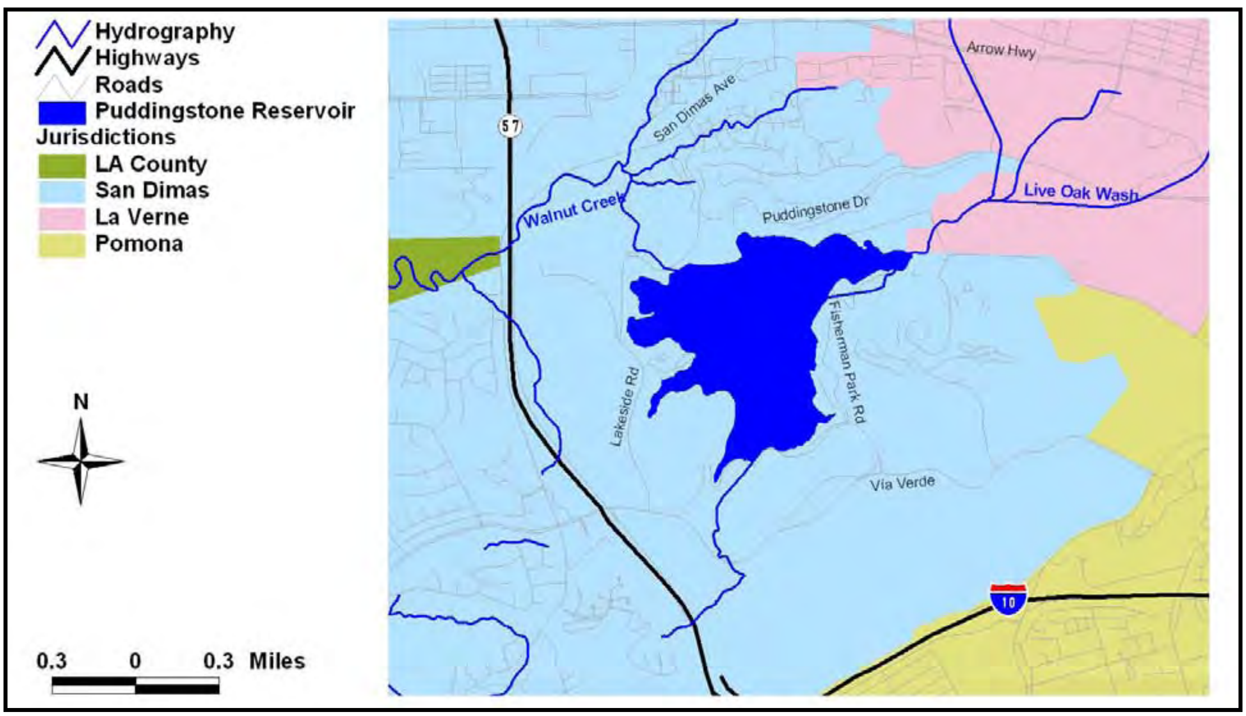

Figure 10-1. **Location of Puddingstone Reservoir**

Figure 10-2. **View of Puddingstone Reservoir**

Figure 10-3. **Live Oak Wash with Puddingstone Channel Joining on the Left**

10.1.1 Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries

Puddingstone Reservoir drains an area of 8,128 acres that ranges in elevation from 267 meters to 1,125 meters (Figure 10-4). The TMDL subwatershed boundaries selected for Puddingstone Reservoir were based on boundaries obtained from the county of Los Angeles. The county of Los Angeles subwatersheds were aggregated to two larger subwatersheds with an internal boundary chosen to separate those areas that drain to a storm drain (the northern subwatershed) and those that enter the reservoir via natural tributaries or overland flow (the southern subwatershed). Loads generated from the northern subwatershed will be assigned wasteload allocations because they drain to the storm drain network, while loads from the southern subwatershed will be assigned load allocations because they do not drain to pipes or culverts prior to discharge to the reservoir (atmospheric deposition throughout the watershed will also receive load allocations). The subwatershed draining the northern part of the watershed is 6,959 acres, and the southern subwatershed is 1,169 acres.

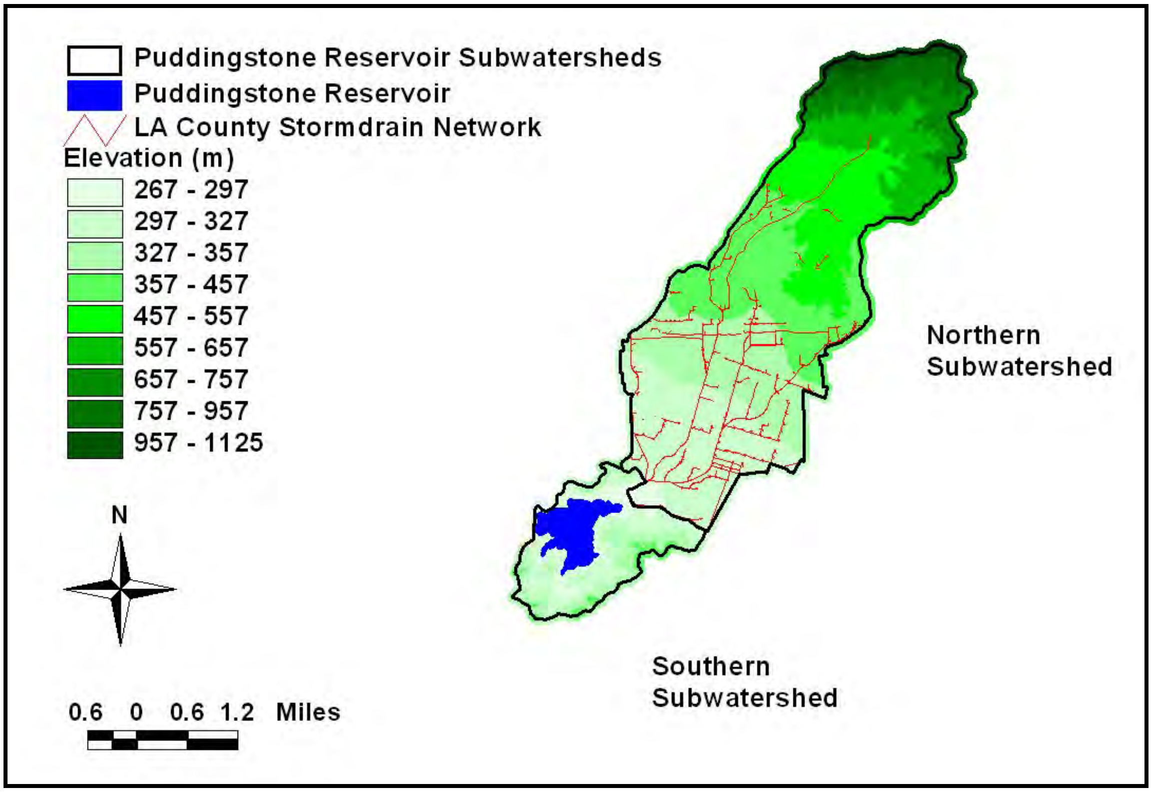

Figure 10-4. **Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries for Puddingstone Reservoir**

10.1.2 MS4 Permittees

Figure 10-5 shows the MS4 stormwater permittees in the Puddingstone Reservoir watershed. The storm drain coverage was provided by the county of Los Angeles. The northern subwatershed is primarily comprised of the county of Los Angeles, Claremont, and La Verne areas, with a small amount of San Dimas, Caltrans, and Angeles National Forest areas. Loads generated from those jurisdictions in the northern subwatershed will be assigned wasteload allocations because they drain to the storm drain network. The southern subwatershed is comprised of San Dimas, La Verne, and Pomona areas. Loads from those jurisdictions originating in the southern subwatershed will be assigned load allocations because they do not drain to pipes or culverts prior to discharge to the reservoir. Figure 10-6 through Figure 10-8 show some of the storm drain and natural drainages to Puddingstone Reservoir. The small amount of Caltrans area in the southern subwatershed will be assigned a wasteload allocation.

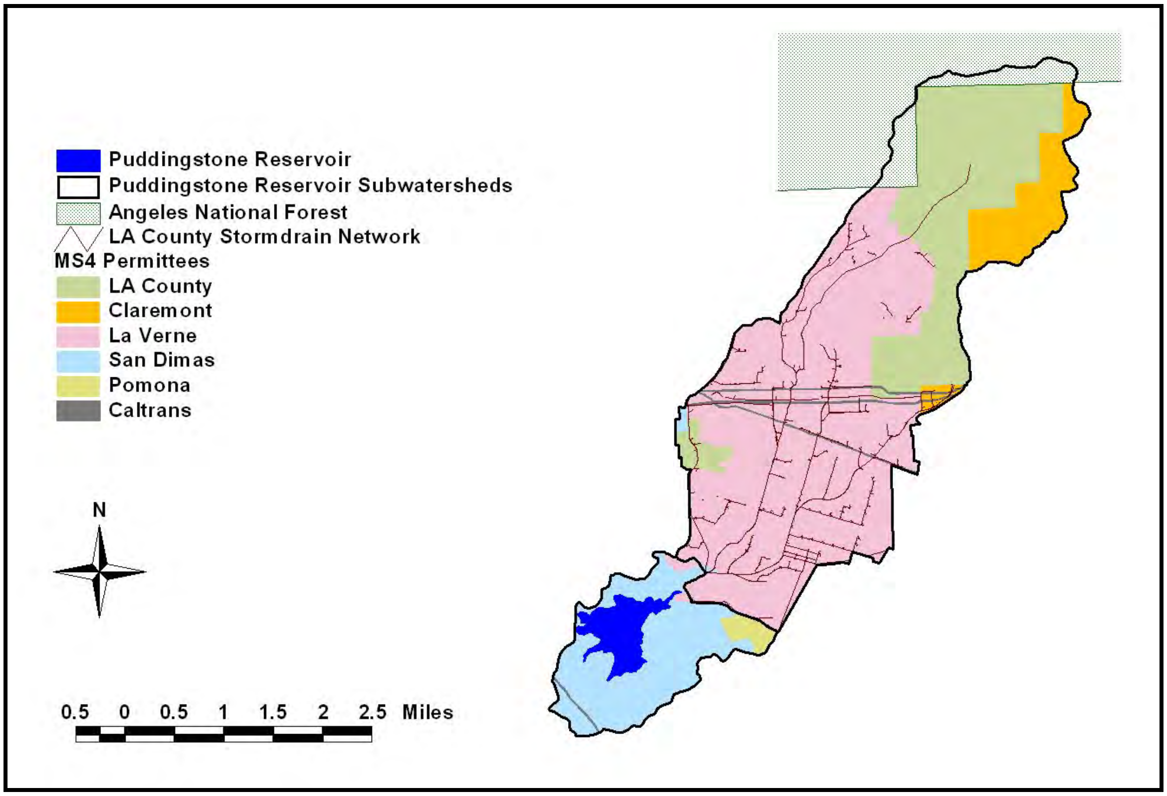

Figure 10-5. **MS4 Permittees and the Storm Drain Network in the Puddingstone Reservoir Subwatersheds**

Figure 10-6. **Storm Drain Discharges to Puddingstone Reservoir**

Figure 10-7. **Natural Drainage Discharge to Puddingstone Reservoir**

Figure 10-8. **Storm Drain Discharge to a Small Depression (that Subsequently Flows to Puddingstone Reservoir)**

10.1.3 Non-MS4 NPDES Dischargers

There are several additional NPDES permits (non-MS4) in the Puddingstone Reservoir watershed (Table 10-1). These include one active discharger covered under a general construction stormwater permit and seven dischargers covered under a general industrial stormwater permit (see Section 3.1 for a detailed discussion of these permit types). These permits were identified by querying excel files of permits from the Regional Board website (Excel files for each watershed are available from this link, www.waterboards.ca.gov/losangeles/water_issues/programs/regional_program/index.shtml#watershed, accessed on October 5, 2009). They are all in the city of La Verne in the northern subwatershed (Figure 10-9) and result in 233 disturbed acres. Specific information is not available regarding these dischargers; however, they are assigned existing loads and wasteload allocations based on their area (industrial stormwater) and disturbed area (construction stormwater).

Table 10-1. Non-MS4 Permits in the Puddingstone Reservoir Watershed

Type of NPDES Permit	Number of Permits	Subwatershed	Jurisdiction	Disturbed Area
General Construction Stormwater (Order No. 99-08-DWQ, CAS000002)	1	Northern	La Verne	36.0 acres
General Industrial Stormwater (Order No. 97-03-DWQ, CAS000001)	7	Northern	La Verne	197 acres

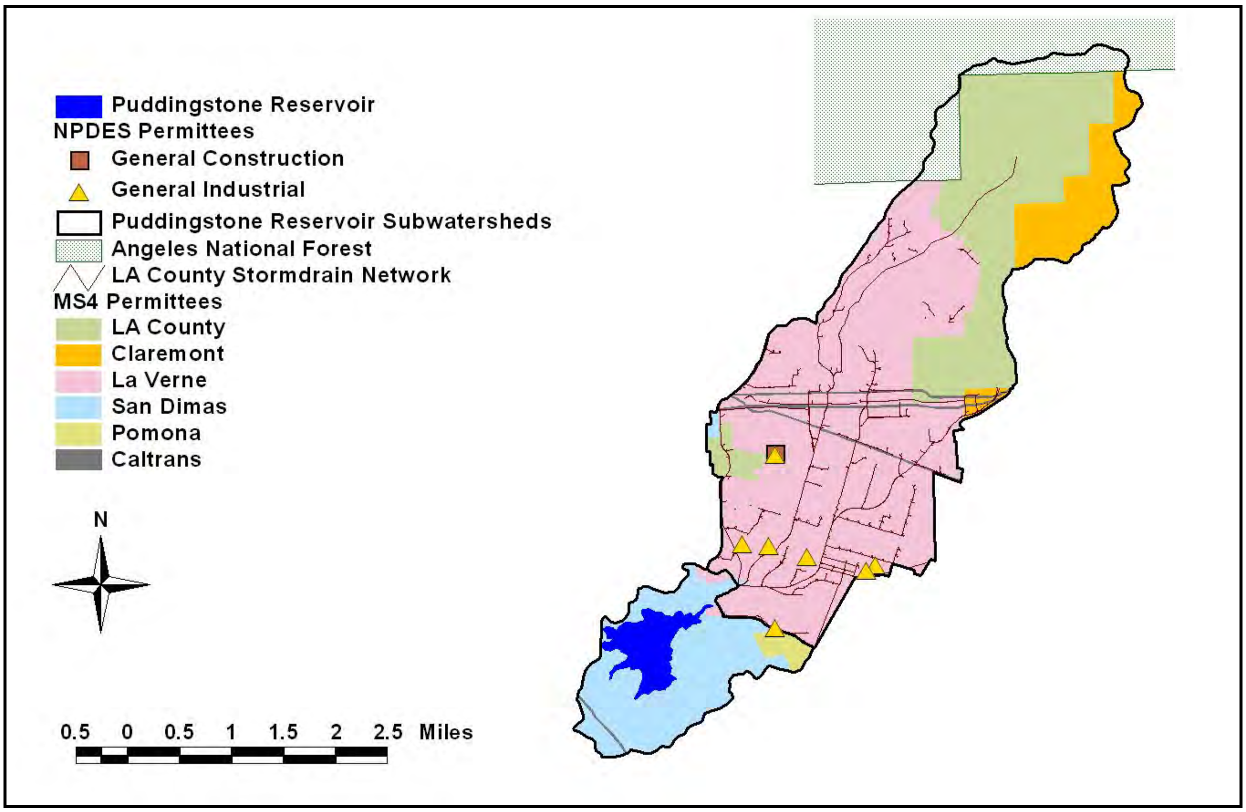

Figure 10-9. **Non-MS4 Permits in the Puddingstone Reservoir Subwatersheds**

10.1.4 Land Uses and Soil Types

Several of the analyses for the Puddingstone Reservoir watershed include source loading estimates obtained from the San Gabriel River Basin LSPC Model discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the San Gabriel River Basin LSPC model are largely residential and shrub and brush rangeland and are shown in Figure 10-10 (based on SCAG 2000 land use data). Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that some of the areas classified by the LSPC model as agriculture or strip mines were inaccurate. Inaccuracies in land use assignment were corrected for each subwatershed and jurisdiction to reflect the more recent SCAG 2005 dataset and current satellite imagery. All areas within the Caltrans jurisdiction were simulated as transportation. Table 10-2 and Table 10-3 summarize the land use areas for each TMDL subwatershed and jurisdiction.

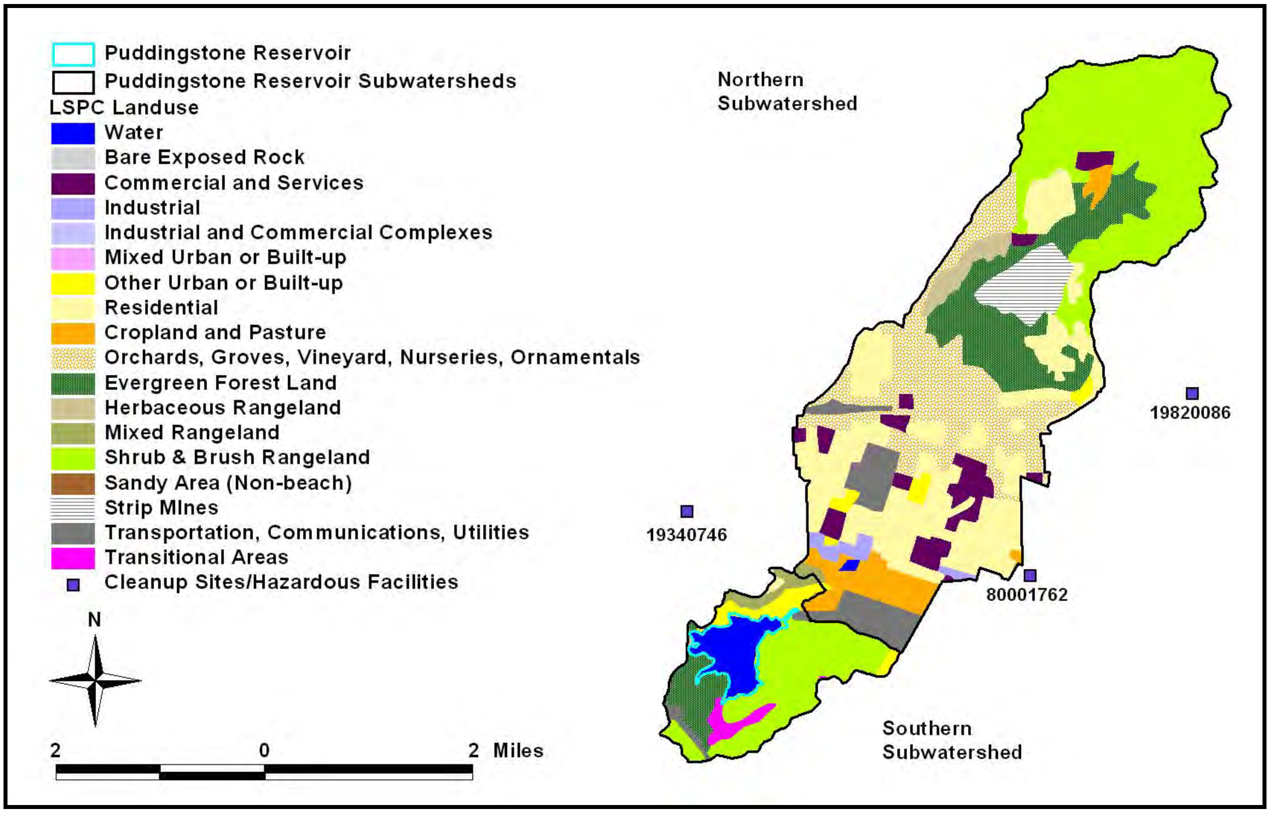

Figure 10-10. **LSPC Land Use Classes for the Puddingstone Reservoir Subwatersheds**

Table 10-2. Land Use Areas (ac) Draining the Northern Subwatershed of Puddingstone Reservoir

Land Use	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Commercial and services	0	38.8	295	0.291	11.0	0	0	**345**
Cropland and pasture	2.91	22.5	199	0	0	0	0	**225**
Evergreen forest land	42.9	378	376	0	0	0	0	**797**
Herbaceous rangeland	0	0	123	0	0	0	0	**123**
Industrial	0	0	82.3	0	0	0	0	**82.3**
Mixed rangeland	0	21.5	111	1.08	1.95	0	0	**135**
Other urban or built-up	8.07	9.24	58.2	0.005	2.90	0	0	**78.4**
Residential	28.4	467	2,469	0.260	10.0	0	0	**2,975**

Land Use	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Shrub & brush rangeland	496	926	19.7	0.097	0.53	0	293	**1,736**
Transportation, communications, utilities	0	0.97	346	3.55	2.12	110	0	**463**
Transitional areas	0	0	0	0	0	0	0	**0**
Total	**578**	**1,865**	**4,079**	**5.28**	**28.5**	**110**	**293**	**6,959**

Table 10-3. Land Use Areas (ac) Draining the Southern Subwatershed of Puddingstone Reservoir

Land Use	La Verne	Pomona	San Dimas	Caltrans	Total
Commercial and services	0	0	0	0	**0**
Cropland and pasture	0	0	0	0	**0**
Evergreen forest land	0	0	184	0	**184**
Herbaceous rangeland	0	0	4.33	0	**4.33**
Industrial	0	0	0	0	**0**
Mixed rangeland	23.7	0	48.5	0	**72.2**
Other urban or built-up	1.35	19.1	101	0	**122**
Residential	0	0	10.7	0	**10.7**
Shrub & brush rangeland	0.006	62.1	602	0	**664**
Transportation, communications, utilities	8.44	0.616	23.0	11.6	**43.6**
Transitional areas	0	0	68.2	0	**68.2**
Total	**33.5**	**81.8**	**1,042**	**11.6**	**1,169**

Three Resource Conservation and Recovery Act (RCRA) cleanup sites are located near the Puddingstone Reservoir watershed (these are within one mile of the watershed, as illustrated in Figure 10-10). Information regarding these facilities is summarized in Table 10-4. No additional information regarding potential contaminants of concern is available for one site. The potential contaminants of concern identified at these three sites are not relevant to the nutrients, mercury, chlordane, dieldrin, PCBs or DDT impairments. It is not known whether or not these facilities contributed mercury, chlordane, dieldrin, PCBs, or DDT to Puddingstone Reservoir in the past. None of these sites should be contributing loading under existing conditions.

Table 10-4. RCRA Cleanup Sites near the Puddingstone Reservoir Watershed

Envirostor #	Facility Name	Cleanup Status	Potential Contaminants of Concern
19340746	Cropper's Plating Site	Certified	Chromium III, copper and compounds, organic lead (tetra ethyl lead)
19820086	La Puerta Elementary School	Certified	No data in site summary database for this facility
80001762 (CAD980894562)	Safety-Kleen Corp.	Inactive	Cadmium, chromium, lead, benzenes, TCE, PCE and non-halogenated solvents

Figure 10-11 shows the predominant soils identified by STATSGO (Appendix D, Wet Weather Loading) in the Puddingstone Reservoir subwatersheds. The soil type identified as Zamora-Urban land-Ramona (MUKEY 660480) comprises the largest area. The soil hydrologic group for this soil is not identified in the data set, which typically indicates either water, bedrock, or urban impervious surfaces. There are two hydrologic group C soils in the watershed (Soper-Fontana-Calleguas-Balcom-Anaheim, MUKEY 660477 and Sobrante-Exchequer-Cieneba, MUKEY 660501). These soils are characterized as moderately-fine to fine-textured and have low infiltration rates when wet; they consist chiefly of soils having a layer that impedes downward movement of water. A small part of the watershed contains a hydrologic group A soil (Urban land-Tujunga-Soboba-Hanford, MUKEY 660474), which has low runoff potential and high infiltration rates even when wet. This soil consists chiefly of sand and gravel and is well-drained to excessively-drained. The San Gabriel River Basin LSPC model does not explicitly use hydrologic soil group as a modeling parameter, though the characteristics of the hydrologic soil group influence parameters such as infiltration rate.

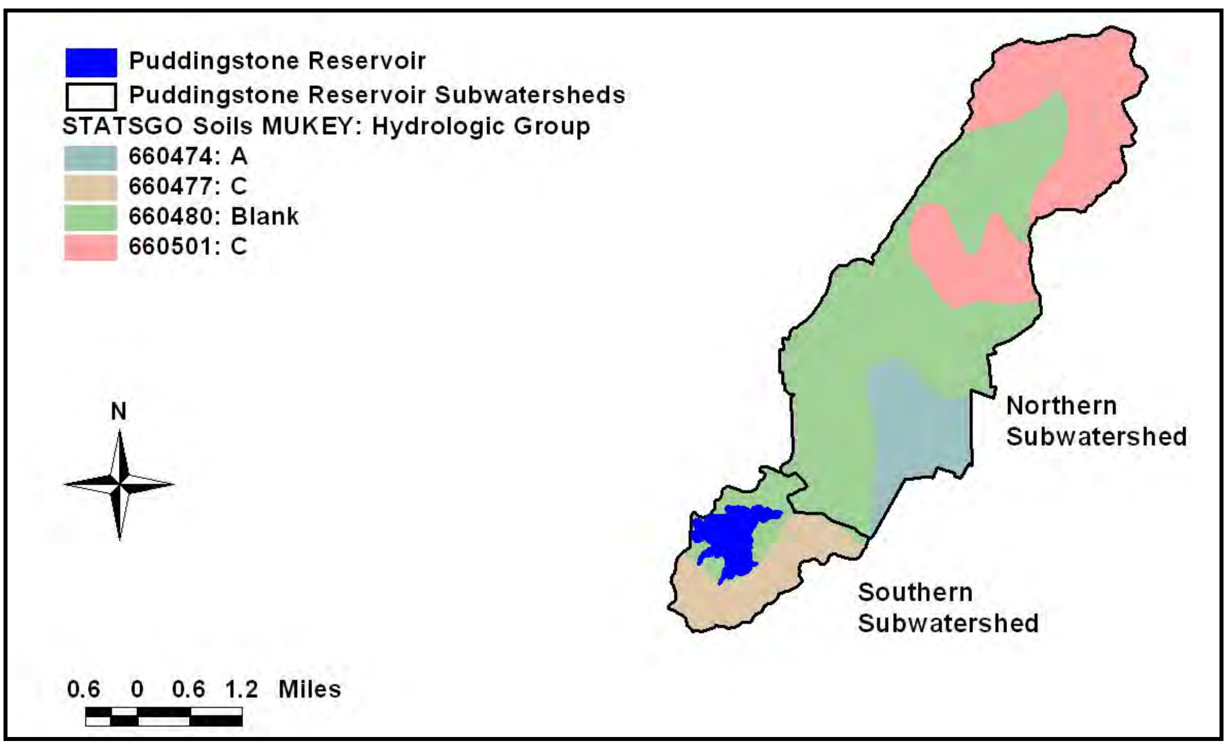

Figure 10-11. **STATSGO Soil Types Present in the Puddingstone Reservoir Subwatersheds**

10.1.5 Additional Inputs

Puddingstone Reservoir does not receive direct inputs from groundwater or potable water sources. Areas around the lake are irrigated with reclaimed water, some of which may reach the reservoir (10.1 percent of the total irrigation volume is assumed to reach the lake). Application of chlorine in the swim beach area may impact pH levels in the lake. The impacts of irrigation and chlorination are discussed in Appendix F (Dry Weather Loading).

10.2 NUTRIENT-RELATED IMPAIRMENTS

A number of the assessed impairments for Puddingstone Reservoir may be associated with nutrients and eutrophication. Nutrient-related impairments for Puddingstone Reservoir include organic enrichment/low dissolved oxygen (DO) (SWRCB, 2010). The loading of excess nutrients enhances algal growth (eutrophication). Algae produce oxygen during photosynthesis but remove oxygen through respiration or decay, resulting in a net depression of DO in the absence of sunlight. Algal photosynthesis can also affect the pH balance of the lake through the removal of carbon dioxide.

10.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated nutrient levels are impairing the REC1/REC2, WARM, and COLD, uses and can potentially impair WILD, MUN, GWR and RARE uses by stimulating algal growth that may form mats that impede recreational and drinking water use, alter pH and dissolved oxygen (DO) levels, alter biology that impair aquatic life, and cause odor and aesthetic problems.

10.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Puddingstone Reservoir. The following targets apply to the organic enrichment/low DO impairment (see Section 2 for additional details and Table 10-5 for a summary):

- The Basin Plan addresses excess aquatic growth in the form of a narrative objective for nutrients. Excessive nutrient concentrations (e.g., nitrogen and phosphorous) in a waterbody can lead to nuisance effects such as algae, odors, and scum. The objective specifies, "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growth causes nuisance or adversely affects beneficial uses." The Regional Board has not adopted numeric targets for biostimulatory nutrients or chlorophyll *a* in Puddingstone Reservoir; however, as described in Tetra Tech (2006), summer (May to September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L are selected as the maximum allowable level consistent with full support of contact recreational use and are also consistent with supporting warm water aquatic life. The mean chlorophyll *a* target must be met at half of the Secchi depth during the summer (May – September) and annual averaging periods.
- The Basin Plan states "at a minimum the mean annual dissolved oxygen concentrations of all waters shall be greater than 7 mg/L, and no single determinations shall be less than 5.0 mg/L,

except when natural conditions cause lesser concentrations." In addition, the Basin Plan states, "the dissolved oxygen content of all surface waters designated as WARM shall not be depressed below 5 mg/L as a result of waste discharges" and "the dissolved oxygen content of all surface waters designated as COLD shall not be depressed below 6 mg/L as a result of waste discharges." Deep lakes that thermally stratify during the summer months, such as Puddingstone Reservoir, must meet the DO target in the epilimnion of the water column.

The epilimnion is the upper stratum of more or less uniformly warm, circulating, and fairly turbulent water during summer stratification. The epilimnion floats above a cold relatively undisturbed region called the hypolimnion. The stratum between the two is the metalimnion and is characterized by a thermocline, which refers to the plane of maximum rate of decrease of temperature with respect to depth. For the purposes of these TMDLs the presence of stratification will be defined by whether there is a change in lake temperature greater than 1 degree Celsius per meter. Deep lakes, such as Puddingstone Reservoir, must meet the DO and pH targets in the water column from the surface to 0.3 m above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the DO and pH targets must be met in the epilimnion, the portion of the water column above the thermocline.

- The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." Deep lakes that thermally stratify during the summer months, such as Puddingstone Reservoir, must meet the pH target in the epilimnion of the water column.

Nitrogen and phosphorus target concentrations are based on simulation of allowable loads with the NNE BATHTUB model (Section 10.2.6). Based on the calibrated model for Puddingstone Reservoir, the target nutrient concentrations within the lake are

- 0.71 mg-N/L summer average (May – September) and annual average
- 0.071 mg-P/L summer average (May – September) and annual average

Table 10-5. Nutrient-Related Numeric Targets for Puddingstone Reservoir

Parameter	Numeric Target	Notes
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 6 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA's 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA's recommended criteria. For this reason, EPA's recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.71 mg-N/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model
Total Phosphorous	0.071 mg-P/L summer average (May – September) and annual average	Based on simulation of allowable loads from the NNE BATHTUB model

10.2.3 Summary of Monitoring Data

This section briefly summarizes the nutrient-related monitoring data for Puddingstone Reservoir. Appendix G (Monitoring Data) contains more detailed information regarding water quality sampling in the lake.

Puddingstone Reservoir was monitored for water quality in 1992 and 1993 in support of the Urban Lakes Study near the center of the northern half of the lake. TKN ranged from 0.3 mg-N/L to 6.9 mg-N/L, although concentrations greater than 1.2 mg-N/L only occurred at depths greater than or equal to 8 meters. Ammonium ranged from 0.1 mg-N/L to 5.3 mg-N/L with 39 measurements less than the detection limit (0.01 mg-N/L); concentrations did not exceed 0.2 mg-N/L except at depths greater than or equal to 8 meters. Each of the 75 measurements of nitrite was less than the detection limit (0.01 mg-N/L), and 23 nitrate samples were less than the detection limit (0.01 mg-N/L). The maximum concentration of nitrate observed was 2 mg-N/L. Forty-nine of 75 samples of orthophosphate were less than the detection limit (0.01 mg-P/L), and the maximum concentration observed was 1.7 mg-P/L. Total phosphorus was similar with 45 measurements less than the detection limit (0.01 mg-P/L) and a maximum observed concentration of 1.3 mg-P/L. Concentrations of orthophosphate and total phosphorus did not exceed 0.2 mg-P/L except at depths greater than or equal to 14 meters. pH ranged from 7.4 to 9.0, and TOC ranged from 2.8 mg/L to 8.2 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 4 µg/L to 22 µg/L with an average of 13 µg/L.

The 1996 Water Quality Assessment Report contains summary information regarding the DO impairment which was listed as not supporting the aquatic life use. DO was measured 187 times with concentrations ranging from 0.1 mg/L to 14.9 mg/L. However, the accompanying database does not contain these measurements so no information regarding location, time, depth, or temperature can be compared. There are some temperature and pH measurements in the database that were collected from December 1977 through March 1978. Temperature ranged from 11.1 ºC to 11.7 ºC, and pH ranged from 6.6 to 7.6.

More recent monitoring of nutrients in Puddingstone Reservoir occurred on November 18, 2008 at four locations in the lake. All samples of ammonia, TKN, nitrate, nitrite, orthophosphate, and total phosphate collected at the four lake stations were below the detection limits of 0.1 mg-N/L, 1 mg-N/L, 0.1 mg-N/L, 0.1 mg-N/L, 0.4 mg-P/L, and 0.5-P mg/L, respectively. Chlorophyll *a* ranged from 11.3 µg/L to 21.4 µg/L.

Puddingstone Reservoir was sampled in February 2009 by USEPA and the Regional Board. The field notes report that approximately 300 gallons of chlorine are pumped into the swim beach area each week during the summer. The edges of the lake are sometimes treated for weeds. Samples were collected from a depth of 1.5 meters at two locations. Secchi depths were 0.76 meters at all locations. Ammonia samples ranged from 0.03 mg-N/L to 0.04 mg-N/L. TKN ranged from 1.3 mg-N/L to 1.7 mg-N/L. Nitrite ranged from 0.02 mg-N/L to 0.05 mg-N/L, and nitrate ranged from 0.02 mg-N/L to 0.26 mg-N/L. Orthophosphate ranged from 0.016 mg-P/L to 0.062 mg-P/L; total phosphorus ranged from 0.098 mg-P/L to 0.121 mg-P/L. Chlorophyll *a* measurements were high during this event and ranged from 66.1 µg/L to 113.5 µg/L. These chlorophyll *a* results are anomalously high compared to later measurements taken during the summer, however, these levels were measured one week after a major rain events that likely delivered high nutrient loads to the lake. Reported concentrations of DO decreased from over 6 mg/L at the surface to 0 mg/L at 3 meters to 4 meters. pH ranged from 7.6 to 9.4 at each station. Temperature at these two stations ranged from 11.3 ºC to 14.6 ºC. Field operators found DO readings suspicious and sent the meter off for repair (Greg Nagle, USEPA Region IX, personal communication, 5/22/09). These DO results were excluded from the relevant data set based on poor quality assurance.

In July 2009, Puddingstone Reservoir was sampled at two locations. Ammonia, nitrite, nitrate, and orthophosphate concentrations were less than the detection limits of 0.03 mg-N/L, 0.01 mg-N/L, 0.01 mg-N/L, and 0.0075 mg-P/L, respectively. Total phosphorus were 0.041 mg-P/L and 0.164 mg-P/L, though the field duplicate for the higher sample was 0.048 mg-P/L. Chlorophyll *a* concentrations were 25.1 µg/L and 27.3 µg/L. DO concentrations were above 8 mg/L throughout the epilimnion. pH ranged from 8.52 to 8.92.

In summary, chlorophyll *a* concentrations are typically above the summer average target concentration of 20 µg/L. Although conditions in February 2009 may have been anomalous (i.e., winter concentrations were significantly higher than all other chlorophyll *a* concentrations), the concentrations measured during the July 2009 event averaged 26 µg/L. Based on the July 2009 profile measurements, DO is meeting the target COLD concentration of 6 mg/L throughout the epilimnion. Readings collected in February may have been collected with a malfunctioning meter. Exceedances of the allowable range for pH (6.5 to 8.5) have been observed as well. The nutrient TMDLs for Puddingstone Reservoir presented in Section 10.2.6 account for summer season critical conditions by assessing loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These reductions in nutrient loading are expected to alleviate pH, odor, DO, and ammonia problems associated with excessive nutrient loading and eutrophication.

10.2.4 Source Assessment

The majority of nutrient loading to Puddingstone Reservoir originates from the surrounding watershed (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading) including irrigation (10.1 percent of the total irrigation volume is assumed to reach the lake). Loading due to direct deposition from the atmosphere is discussed in Appendix E (Atmospheric Deposition). The northern subwatershed comprises 85.6 percent of the drainage area and contributes 86 percent and 90 percent of the total phosphorus and total nitrogen loads, respectively, to Puddingstone Reservoir. The majority of the remaining load originates from the southern subwatershed. All existing loads to Puddingstone Reservoir are summarized in Table 10-6.

Table 10-6. Summary of Average Annual Flows and Nutrient Loading to Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Northern	Caltrans	State Highway Stormwater[1]	141	253 (3.6)	1,603 (3.4)
Northern	Claremont	MS4 Stormwater[1]	206	256 (3.6)	1,786 (3.8)
Northern	County of Los Angeles	MS4 Stormwater[1]	773	1,124 (15.9)	7,299 (15.6)
Northern	La Verne[2]	MS4 Stormwater[1]	2,361	4,209 (59.5)	25,332 (54.0)
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater [1]	258	409 (5.8)	3,008 (6.4)
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater [1]	47.1	74.7 (1.1)	550 (1.2)
Northern	Pomona	MS4 Stormwater[1]	5.48	9.6 (0.1)	60.9 (0.1)
Northern	San Dimas	MS4 Stormwater[1]	26.5	47.2 (0.7)	294 (0.6)
Northern	Angeles National Forest	Stormwater[1]	34.6	10.3 (0.1)	301 (0.6)

Subwatershed	Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Southern	Caltrans	State Highway Stormwater[1]	12.4	22.5 (0.3)	148 (0.3)
Southern	La Verne	Runoff	13.0	19.4 (0.3)	147 (0.3)
Southern	Pomona	Runoff	25.1	34.5 (0.5)	276 (0.6)
Southern	San Dimas	Runoff	229	272 (3.8)	2,433 (5.2)
Southern	County of Los Angeles	Parkland Irrigation	163	337 (4.8)	3,425 (7.3)
Lake Surface		Atmospheric Deposition[3]	366	NA	209 (0.4)
Total			**4,661**	**7,078**	**46,872**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

[3] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

10.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. To simulate the impacts of nutrient loading on Puddingstone Reservoir, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion, based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Puddingstone Reservoir, the following inputs apply: surface area of 252 acres, average depth of 24.6 ft, and volume of 6,200 ac-ft. Based on the phosphorus turnover ratio for this lake (Walker, 1987), the annual averaging period is appropriate (i.e., annual loads are input to the model rather than summer season loads).

The NNE BATHTUB Tool was calibrated to average summer season water quality data observed over twice the typical Secchi depth (2*1.15 m = 2.3 m). To predict the average observed total nitrogen concentration over this depth (1.06 mg-N/L), the calibration factor on the net nitrogen sedimentation rate was set to 1.7. The calibration factor on the net phosphorus sedimentation rate was set to the maximum suggested (2) (Walker, 1987), and the resulting concentration is 0.08 mg-P/L. Although this calibrated sedimentation rate reflects the net effects of phosphorus settling and resuspension, the high calibration factor indicates that settling is the more dominant mechanism in this system, and internal phosphorus loading is likely insignificant relative to the other sources of loading. The reductions in external phosphorus loading in the lake required by this TMDL should lead to further suppression of internal loading. To simulate the average observed chlorophyll *a* concentration, the calibration factor on concentration was set to 1.5 for a predicted concentration of 26 µg/L.

10.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Following calibration of the NNE BATHTUB Tool (Section 10.2.5), the allowable loading combinations of nitrogen and phosphorus were calculated using Visual Basic's GoalSeek function (Appendix A, Nutrient TMDL Development). The loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 0.71 mg-N/L summer average (May – September) and annual average
- 0.071 mg-P/L summer average (May – September) and annual average

The loading capacities for total nitrogen and total phosphorus are 24,190 lb-N/yr and 5,181 lb-P/yr, respectively. These loading capacities can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load (divided among WLAs and LAs) is 46.4 percent of the existing load of 46,872 lb-N/yr, or 21,771 lb-N/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. WLAs and LAs are developed assuming equal percent load reductions in all sources. The resulting TMDL equation for TN is then:

24,190 lb-N/yr = 18,756 lb-N/yr + 3,015 lb-N/yr + 2,419 lb-N/yr

For total phosphorus, the allocatable load (divided among WLAs and LAs) is 65.9 percent of the existing load of 7,078 lb-P/yr, or 4,663 lb-P/yr. This value represents 90 percent of the loading capacity, while the MOS is 10 percent of the loading capacity. The resulting TMDL equation for TP is:

5,181 lb-P/yr = 4,226 lb-P/yr + 437 lb-P/yr + 518 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on simulation of allowable loads with the NNE BATHTUB model (see Section 10.2.5). These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

10.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs at their point of discharge. The wasteload allocations for most point sources are mass-based; however, the wasteload allocations for stormwater discharges that are covered under general industrial and construction stormwater permits are concentration-based. In addition, these TMDLs establish alternative wasteload allocations for total phosphorous and alternative wasteload allocations for total nitrogen (collectively, "Approved Lake Management Plan Wasteload Allocations"). The Approved Lake Management Plan Wasteload allocations are concentration-based and are described in Section 10.2.6.1.2. The Approved Lake Management Plan Wasteload allocations will supersede the wasteload allocations in Section 10.2.6.1.1 if the conditions described in Section 10.2.6.1.2 are met.

Under either wasteload allocation scheme responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on the Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

10.2.6.1.1 Wasteload Allocations

The northern subwatershed drains to a series of storm drains prior to discharging to Puddingstone Reservoir. Therefore, all loads associated with this drainage area are assigned WLAs. The loads attributed to the Caltrans areas in the southern subwatershed and the general construction and industrial stormwater permits also receive WLAs. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Total phosphorus WLAs represent a 34.1 percent reduction in existing loading, and total nitrogen WLAs represent a 53.6 percent reduction in existing loading. These loading values (in pounds per year) represent the TMDLs wasteload allocations (Table 10-7).

Each WLA applies at the point of discharge. As noted in Table 10-7 below, the concentration-based WLAs will be used to evaluate compliance with the allocations for the current discharges authorized by the general industrial stormwater permit and the construction stormwater permit and any future discharges in the watershed authorized by the general industrial and construction stormwater permits. The phosphorous and nitrogen WLA concentrations were calculated by dividing the allowable load (in lb/yr; Table 10-7) by their respective estimated flow rates (258 ac-ft/yr and 47 ac-ft/yr for industrial and construction sites, respectively; Table 10-6) and applying the appropriate conversion factors to yield concentrations in mg/L.

Table 10-7. Wasteload Allocations of Phosphorus and Nitrogen Loading to Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Wasteload Allocation Total Phosphorus[4] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Wasteload Allocation Total Nitrogen[4] (lb/yr)
Northern	Caltrans	State Highway Stormwater[1]	253	167	1,603	745
Northern	Claremont	MS4 Stormwater[1]	256	169	1,786	829
Northern	County of Los Angeles	MS4 Stormwater[1]	1,124	741	7,299	3,390
Northern	La Verne[2]	MS4 Stormwater[1]	4,209	2,772	25,332	11,766
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	409	269 (0.4 mg/L P)[3]	3,008	1,397 (2.0 mg/L N)[3]
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	74.7	49 (0.4 mg/L P)[3]	550	255 (2.0 mg/L N)[3]
Northern	Pomona	MS4 Stormwater[1]	9.57	6.30	60.9	28.3

Subwatershed	Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Wasteload Allocation Total Phosphorus[4] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Wasteload Allocation Total Nitrogen[4] (lb/yr)
Northern	San Dimas	MS4 Stormwater[1]	47.2	31.1	294	137
Northern	Angeles National Forest	Stormwater[1]	10.3	6.8	301	140
Southern	Caltrans	State Highway Stormwater[1]	22.5	14.8	148	68.2
Total			**6,415**	**4,226**	**40,382**	**18,756**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations (see footnote #3).

[3] For these responsible jurisdictions, the concentration-based WLA will be use to evaluate compliance.

[4] Each wasteload allocation must be met at the point of discharge.

10.2.6.1.2 Alternative "Approved Lake Management Plan Wasteload Allocations"

Concentration-based WLAs not exceeding the concentration listed in Table 10-8 are effective and supersede corresponding WLAs for a responsible jurisdiction in Table 10-7 if:

1. The responsible jurisdiction requests that concentration-based wasteload allocations not to exceed the concentrations established in Table 10-8 apply to it;
2. The responsible jurisdiction provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause each of the following to be met: the applicable water quality criteria for ammonia, dissolved oxygen and pH; the chlorophyll *a* targets listed in Table 10-5; and the requested concentration-based WLAs. Responsible jurisdictions may work together to develop, submit and implement the Lake Management Plan. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The responsible jurisdiction may use monitoring data and modeling to show that the water quality criteria, targets and requested WLAs will be met;
3. The Regional Board Executive Officer approves the request and applies concentration-based wasteload allocations for total nitrogen and total phosphorus. These wasteload allocations are not to exceed the concentrations in Table 10-8 as a summer average (May-September) and annual average, and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based WLA must be met in the lake.

Table 10-8. Alternative Wasteload Allocations of Phosphorus and Nitrogen in Puddingstone Reservoir if an Approved Lake Management Plan Exists

Subwatershed	Responsible Jurisdiction	Input	Maximum Allowable Wasteload Allocation Total Phosphorus[3] (mg-P/L)	Maximum Allowable Wasteload Allocation Total Nitrogen[3] (mg-N/L)
Northern	Caltrans	State Highway Stormwater[1]	0.1	1.0
Northern	Claremont	MS4 Stormwater[1]	0.1	1.0
Northern	County of Los Angeles	MS4 Stormwater[1]	0.1	1.0
Northern	La Verne[2]	MS4 Stormwater[1]	0.1	1.0
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	0.1	1.0
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	0.1	1.0
Northern	Pomona	MS4 Stormwater[1]	0.1	1.0
Northern	San Dimas	MS4 Stormwater[1]	0.1	1.0
Northern	Angeles National Forest	Stormwater[1]	0.1	1.0
Southern	Caltrans	State Highway Stormwater[1]	0.1	1.0

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each concentration-based wasteload allocation must be met in the lake.

10.2.6.2 Load Allocations

These TMDLs establish load allocations (LAs) and alternative LAs for total phosphorous and total nitrogen. The alternative LAs will be effective and supersede the LAs listed in Table 10-9 if the conditions described in Section 10.2.6.2.2 are met.

10.2.6.2.1 Load Allocations

Loads associated with the southern subwatershed are assigned LAs. Total phosphorus LAs represent a 34.1 percent reduction in existing loading, and total nitrogen LAs represent a 53.6 percent reduction in existing loading. LAs are provided for each responsible jurisdiction and input and must be met at the point of discharge. These loading values (in pounds per year) represent the TMDLs load allocations (Table 10-9).

Table 10-9. Load Allocations of Phosphorus and Nitrogen Loading to Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Existing Total Phosphorus Load (lb-P/yr)	Load Allocation Total Phosphorus[1] (lb-P/yr)	Existing Total Nitrogen Load (lb-N/yr)	Load Allocation Total Nitrogen[1] (lb/yr)
Southern	La Verne	Runoff	19.4	12.8	147	68.2
Southern	Pomona	Runoff	34.5	22.7	276	128
Southern	San Dimas	Runoff	272	179	2,433	1,130
Southern	County of Los Angeles	Parkland Irrigation	337	222	3,425	1,591
Lake Surface		Atmospheric Deposition[2]	NA	NA	209	97.3
Total			**663**	**437**	**6,490**	**3,015**

[1] Each load allocation must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

10.2.6.2.2 Alternative "Approved Lake Management Plan Load Allocations"

The load allocation for any responsible jurisdiction listed in Table 10-9 will be superseded, and the load allocation for that responsible jurisdiction in Table 10-10 will apply, if:

1. The responsible jurisdiction requests that concentration-based load allocations not to exceed the concentrations established in Table 10-10 apply to it.
2. The responsible jurisdiction provides to USEPA and the Regional Board a Lake Management Plan describing actions that will be implemented and cause the applicable water quality criteria for ammonia, dissolved oxygen and pH to be met. The plan must also show that the chlorophyll *a* targets listed in Table 10-5 and the alternative total nitrogen and phosphorus targets will be met. Responsible jurisdictions may work together to develop, submit and implement the Lake Management Plan. A Lake Management Plan may include the following types of actions: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; and/or fisheries management actions to reduce nutrient availability from sediments. The responsible jurisdiction may use monitoring data and modeling to show that the water quality criteria and targets will be met.
3. The Regional Board Executive Officer approves the request and applies concentration-based load allocations for total nitrogen and total phosphorus. These load allocations are not to exceed the concentrations in Table 10-10 as a summer average (May-September) and annual average, and
4. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each concentration-based load allocations must be met in the lake.

Table 10-10. Alternative Load Allocations of Phosphorus and Nitrogen Loading to Puddingstone Reservoir if an Approved Lake Management Plan Exists

Subwatershed	Responsible Jurisdiction	Input	Maximum Allowable Load Allocation Total Phosphorus[1] (mg-P/L)	Maximum Allowable Load Allocation Total Nitrogen[1] (mg-N/L)
Southern	La Verne	Runoff	0.1	1.0
Southern	Pomona	Runoff	0.1	1.0
Southern	San Dimas	Runoff	0.1	1,0
Southern	County of Los Angeles	Parkland Irrigation	0.1	1.0

[1] Each concentration-based load allocations must be met in the lake.

10.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. To account for the uncertainties concerning the relationship between nutrient loading and the resultant in-lake chlorophyll *a* an explicit MOS is included in these TMDLs. This explicit MOS is set at 10 percent of the loading capacity for total phosphorus and total nitrogen.

10.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are expected to alleviate any DO problems associated with excessive nutrient loading and eutrophication. These TMDLs therefore protect for critical conditions.

10.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of nutrient loading to Puddingstone Reservoir occurs during wet weather events that deliver pollutant loads from the surrounding watershed, the daily maximum allowable loads of nitrogen and phosphorus are calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) to the Reservoir multiplied by the allowable concentrations consistent with achieving the long-term loading targets. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for this San Dimas Creek gage (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99th percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The allowable concentrations for phosphorus and nitrogen were calculated from the annual allowable loads (4,663 lb-P/yr and 21,771 lb-N/yr) divided by the total annual volume delivered to the lake (2,692 ac-ft/yr) (sum of the runoff-associated WLAs and LAs presented in Table 10-7 and Table 10-9, respectively). Multiplying the allowable concentrations (0.637 mg-P/L and 2.97 mg-N/L) by the peak daily flow yields the daily maximum allowable loads which are 131 lb-P/d and 612 lb-N/d. These loads are associated with the MS4 stormwater permittees. For comparison, the existing phosphorus load (7,078 lb-P/yr) would yield an average concentration of 0.967 mg-P/L and a daily load of 199 lb-P/d. The existing nitrogen load (46,872 lb-N/yr) would yield an average concentration of 6.4 mg-N/L and a daily load of 1,318 lb-N/d. As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

10.2.6.6 Future Growth

Much of the Puddingstone Reservoir watershed remains in shrub and brush rangeland. As development occurs in this watershed, best management practices (BMPs) will be required such that loading rates are consistent with the allocations established by these TMDLs. Therefore, no load allocation has been set aside for future growth.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.3 MERCURY IMPAIRMENT

The listing information for Puddingstone Reservoir (LARWCB, 1996) indicates that fish tissue data collected by the Toxic Substances Monitoring Program (TSMP) exceeded the fish tissue guideline and forms the basis for this listing. Recent data collected by the Surface Water Ambient Monitoring Program (SWAMP) and the San Gabriel Watershed Council (SGWC) indicate that fish tissue levels of mercury remain elevated.

In 2008, the Southern California Coastal Water Research Project (SCCWRP) published a report titled "Extent of Fishing and Fish Consumption by Fishers in Ventura and Los Angeles County Watersheds." The purpose of the study was to document the fishing habits and consumption rates of fishers in these counties (SCCWRP, 2008). Puddingstone Reservoir was visited five times, during which 95 fishers were observed. Forty fishers were interviewed, and 55 percent of those consume fish caught from this waterbody. Of the 19 sampling sites located in the San Gabriel River Basin, Puddingstone Reservoir had the second highest number of observed fishers, and the highest number of people interviewed who consume fish caught from the survey location. As previously noted, according to the California Fish and Game, the reservoir is periodically stocked with trout (California Department of Fish and Game, 2009).

10.3.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Applicable water quality criteria are also specified in the California Toxics Rule (USEPA, 2000a). The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Concentrations of mercury measured in fish tissue collected from Puddingstone Reservoir indicate that the REC1, REC2, WARM, and COLD, are currently impaired and at high enough concentrations WILD, MUN, GWR, and RARE uses could be impaired.

10.3.2 Numeric Targets

Numeric targets for mercury in Puddingstone Reservoir apply to both the water column and fish tissue. Water column targets are based on beneficial use. For waters designated MUN (existing, potential, or intermittent), the Basin Plan lists a total mercury maximum contaminant level of 0.002 mg/L, or 2 μg/L. The California Toxics Rule includes total mercury human health criteria for the consumption of "water and organisms" or "organisms only" as 0.050 μg/L and 0.051 μg/L, respectively (USEPA, 2000a). California often implements these values on a 30 day average. The "water and organisms" target applies to Puddingstone because it is designated as an asterisked existing use in the Basin Plan. Because the human health criterion for the consumption of "water and organisms" is the most restrictive criterion, a total mercury water column target of 0.050 μg/L (50 ng/L) is the appropriate target.

In addition, a water column target for dissolved methylmercury of 0.081 ng/L is applicable for Puddingstone Reservoir. This value was calculated by dividing the fish tissue guideline (0.22 ppm) with a national bioaccumulation factor (for dissolved methylmercury) of 2,700,000 applicable for trophic level 4 fish (and multiplying by a factor of 10^6 to convert from milligrams to nanograms).

The fish contaminant goal (FCG) for methylmercury defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 220 ppb or 0.22 ppm (wet weight). This concentration is protective of human and wildlife consumers of trophic level four fish. The target length for comparison to this target is 350 mm (13.8 inches) in largemouth bass. Refer to Section 2 of this report for more information regarding these targets.

10.3.3 Summary of Monitoring Data

Total mercury concentrations in the water column of Puddingstone Reservoir have been measured since 1992. In-lake water column mercury concentrations were measured in July and September 1992 as part of the Urban Lakes Study. All 21 measurements were less than the detection limit of 0.5 μg/L (500 ng/L). As the detection limit of this dataset is 10 times higher than the water quality criterion for mercury (50 ng/L), it is difficult to assess compliance in terms of a water column concentration.

More recent samples from November 2008, February 2009, and July 2009 were collected and analyzed with ultra-clean methods and detection limits no greater than 0.5 ng/L. All total mercury samples collected during these events ranged from 0.26 ng/L to 2.52 ng/L and were more than one order of magnitude less than the water column target. Total methylmercury concentrations ranged from 0.025 ng/L to 0.127 ng/L, and one of four samples exceeded the dissolved target concentration of 0.081 ng/L. The average observed methyl mercury concentration (0.065 ng/L) is less than the dissolved target concentration (0.081 ng/L).

Mercury concentrations in the fish tissue of largemouth bass have been measured in Puddingstone Reservoir since 1986 by the TSMP, SGWC, and SWAMP. Figure 10-12 shows the total mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length. As expected, fish tissue mercury concentrations increase with length. Concentrations exceed 0.22 ppm in all individual or composite samples greater than 345 mm. Twenty-three individual and five composite samples exceed the fish tissue target; five individual samples and one composite had concentrations less than the target. All of the fish tissue data were reported as total mercury concentrations, of which over 90 percent are expected to be in the methyl form (USEPA, 2001a). These total mercury data were compared to the methylmercury fish contaminant guidelines, resulting in conservative assessments.

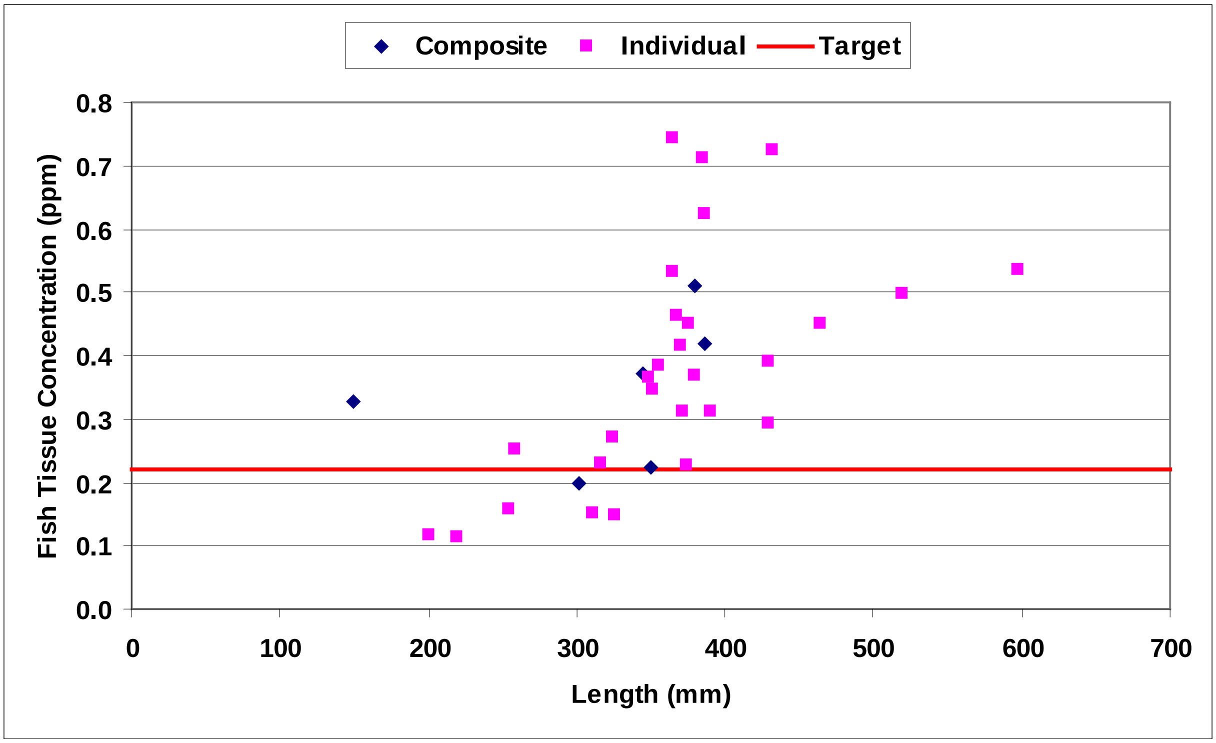

Figure 10-12. **Mercury Concentrations in Largemouth Bass Collected from Puddingstone Reservoir (1986-2007)**

10.3.4 Source Assessment

There are several potential sources of mercury loading in the Puddingstone Reservoir watershed. The majority of loading results from atmospheric deposition to the lake surface. Upland areas are the second largest source; these loads are delivered from tributaries and storm drains in either the water column or sediments. Irrigation of surrounding parklands may contribute loading as well.

Table 10-11 summarizes total mercury loading from the major sources in the watershed. Estimation of watershed loading from MS4 permittees and irrigation of parkland (10.1 percent of the total irrigation volume is assumed to reach the lake) are discussed in more detail in Appendices D and F (Wet and Dry Weather Loading, respectively), Section 10 of both appendices). The atmospheric deposition component of the mercury load is discussed in Appendix E (Atmospheric Deposition). Atmospheric deposition is the largest contributor (47.3 percent) of mercury to Puddingstone Reservoir. The second largest contributor is the MS4 loading from the northern subwatershed (43.6 percent), which contributes loading during wet and dry periods.

Table 10-11. Summary of Existing Total Mercury Loading to Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Total Annual Hg Load (g/yr)	Percent of Load
Northern	Caltrans	State Highway Stormwater[1]	1.32	1.85
Northern	Claremont	MS4 Stormwater[1]	1.26	1.78
Northern	County of Los Angeles	MS4 Stormwater[1]	5.24	7.36
Northern	La Verne[2]	MS4 Stormwater[1]	19.9	27.9
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	2.41	3.38
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	0.44	0.62
Northern	Pomona	MS4 Stormwater[1]	0.049	0.07
Northern	San Dimas	MS4 Stormwater[1]	0.204	0.29
Northern	Angeles National Forest	Stormwater[1]	0.234	0.33
Southern	Caltrans	MS4 Stormwater[1]	0.096	0.13
Southern	La Verne	MS4 Stormwater[1]	0.097	0.14
Southern	Pomona	MS4 Stormwater[1]	0.166	0.23
Southern	San Dimas	MS4 Stormwater[1]	1.57	2.20
Southern	County of Los Angeles	Parkland Irrigation	4.55	6.39
Lake Surface		Atmospheric Deposition[3]	33.7	47.3
Total			**71.2**	**100**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

[3] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

10.3.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. Specifically, models of watershed loading of mercury are combined with an estimated rate of bioaccumulation in the lake. This enables a translation between the numeric target (expressed as a fish tissue concentration of mercury) and mercury loading rates. The loading capacity is then determined via the linkage analysis as the mercury loading rate that is consistent with meeting the target fish tissue concentration.

Neither data nor resources are available to create and calibrate detailed lake response models for mercury cycling in Puddingstone Reservoir. The TMDL target is based on achieving acceptable concentrations in fish. In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of methylmercury in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of methylmercury from lake sediments. In lakes with high sedimentation rates, fish tissue concentrations are therefore likely to respond approximately linearly to reductions in the watershed methylmercury and total mercury load. Two studies have summarized sedimentation rates for Puddingstone Reservoir. According to the Reservoir Sedimentation Database (accessed 6/5/2009), the average annual historical sedimentation rate measured from 1915 to 1941 for Puddingstone Reservoir was 16 ac-ft per year (approximately 0.76 inches per year). The Department of Boating and Waterways and State Coastal Conservancy (2002) reports that the average annual sedimentation rate measured in Puddingstone Reservoir from 1925 to 1980 was 31 ac-ft per year (approximately 1.5 inches per year).

Nationally, authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between methylmercury in water and methylmercury in fish tissue normalized to length. However, this relationship is well-approximated by a linear relationship for the ranges of fish tissue concentration of concern for these impaired lakes. Until such time as a lake response model for mercury is constructed, and sufficient calibration data are collected, an assumption of an approximately linear response of fish tissue concentrations to changes in external loads is sufficient for the development of a TMDL. For a more detailed discussion of the linkage analysis between mercury loading and fish body burden, see Section 3.2.3 of this TMDL report.

10.3.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum load consistent with meeting the numeric target of 0.22 ppm for mercury in largemouth bass. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix C (Mercury TMDL Development).

Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass. To do this, a linear regression analysis was performed on tissue concentrations versus length for Puddingstone Reservoir. The resulting regression equation is

$$Hg(fish) = -0.04001 + 0.001149 \cdot Len, \quad R^2 = 0.32$$

where *Hg(fish)* is the total mercury concentration in largemouth bass (ppm) and *Len* is length in mm. The regression analysis is shown in Figure 10-13, along with the one-sided 95 percent upper confidence limits on mean predictions about the regression line (95 percent UCL) and the 95 percent upper prediction intervals on individual predicted concentrations (95 percent UPI). The UPI gives the confidence limit on the individual predictions for a given length while the UCL gives the confidence limit on the average of the predictions for a given length. This regression has a non-zero intercept and should not be considered valid for lengths less than 150 mm.

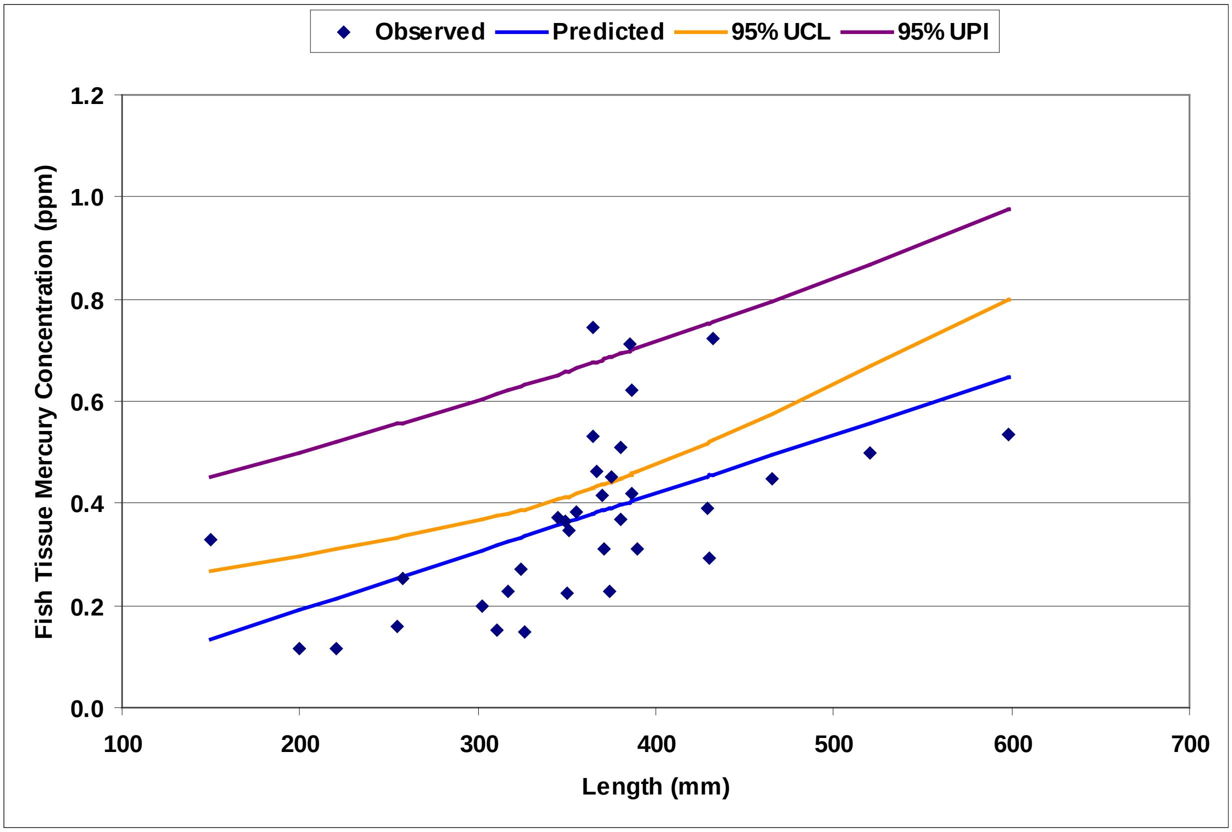

Figure 10-13. **Regression Analysis of Mercury in Puddingstone Reservoir Largemouth Bass**

For mercury, long-term cumulative exposure is the primary concern. Therefore, it is appropriate to use the 95 percent UCL rather than the UPI to provide a Margin of Safety on the appropriate age class. Use of the UCL provides an explicit Margin of Safety because it represents an upper confidence bound on the long-term exposure concentration.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in Puddingstone Reservoir. The TMDL target is established for a 350 mm largemouth bass (see Section 2.2.8). The predicted mercury concentration based on the UCL equation for this length is compared to the target concentration to determine the required reduction in mercury loading, which includes an explicit Margin of Safety as described above.

For Puddingstone Reservoir, the fraction of the existing load consistent with attaining the target (the loading capacity) is the ratio of the target (0.22 ppm) to the best estimate of current average concentrations in the target fish population. The difference between the direct regression estimate and the 95 percent UCL provides the Margin of Safety. Therefore, the allocatable fraction of the existing load (the loading capacity less the Margin of Safety) is the ratio of the target to the 95 percent UCL. The resulting loading capacities and allocatable loads are expressed as fractions of the existing load as summarized in Table 10-12. This analysis indicates that a 46.6 percent reduction in mercury loading will be required to bring fish tissue concentrations in 350 mm largemouth bass (see Section 2.2.8) down to 0.22 ppm.

Table 10-12. Estimated Total Mercury Loading Capacity and Allocatable Load (as Fractions of the Existing Load)

Parameter	Value
Target Concentration (ppm)	0.22
Target Length (mm)	350
Predicted Mercury Concentration at Target Length (ppm)	0.362
95th Percent UCL (ppm)	0.412
Loading Capacity (ratio of target to predicted value)	0.608
Allocatable Load (ratio of target to 95th Percent UCL)	0.534
Required Reduction in Existing Load (1 minus allocatable fraction)	0.466
Margin of Safety Fraction (loading capacity fraction minus allocatable fraction)	0.074

The loading capacity can also be expressed as grams per year (g/yr) using the existing load presented in Table 10-11 and the calculated fractions of the existing load (Table 10-12). Specifically, the loading capacity is 60.8 percent of the existing load of 71.2 g/yr, or 43.3 g/yr. This value can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and margin of safety (MOS) using the equation below.

$$TMDL = \sum WLA + LA + MOS$$

$$43.3g / yr = \sum 16.6g / yr + 21.4g / yr + 5.3g / yr$$

The allocatable load (divided among WLAs and LAs) is 53.4 percent of the existing load of 71.2 g/yr, or 38.0 g/yr. This value represents 88 percent of the loading capacity, while the MOS is 12 percent of the loading capacity. Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

10.3.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). In the Puddingstone Reservoir watershed, WLAs are required for all permittees in the northern subwatershed and Caltrans areas in the southern subwatershed. This TMDL establishes wasteload allocations (WLAs) at their point of discharge. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Table 10-13 summarizes the existing total mercury loads and WLAs for these sources. The WLAs are a 46.6 percent reduction from the existing loads. These loading values (in grams per year) represent the

TMDL wasteload allocations (Table 10-13). In addition to the WLAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L applies.

All responsible jurisdictions must meet the WLAs as a mass load except for storm water permittees under general industrial and construction stormwater permits that are receiving concentration-based WLAs. Each mass based or concentration based wasteload allocation must be met at the point of discharge. In Table 10-13 below, stormwater permittees under general industrial and construction stormwater permits must meet the concentration values to achieve compliance with the WLAs. The WLA concentrations are a 46.6 percent reduction of the existing concentrations associated with these sources, which are calculated by dividing the existing load (in g/yr; see Table 10-13) by the estimated flow rates (258 ac-ft/yr and 47 ac-ft/yr for industrial and construction sites, respectively) and applying the appropriate conversion factors to yield concentrations in ng/L.

Table 10-13. Wasteload Allocations of Total Mercury to the Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Area (ac)	Existing Annual Hg Load (g/yr)	Wasteload Allocation[4] (g/yr)
Northern	Caltrans	State Highway Stormwater[1]	110	1.32	0.702
Northern	Claremont	MS4 Stormwater[1]	578	1.26	0.674
Northern	Count of Los Angeles	MS4 Stormwater[1]	1,865	5.24	2.79
Northern	La Verne[2]	MS4 Stormwater[1]	3,846	19.9	10.6
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	197	2.41	1.29 (4.0 ng/L Hg)[3]
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	36.0	0.44	0.235 (4.0 ng/L Hg)[3]
Northern	Pomona	MS4 Stormwater[1]	5.28	0.0488	0.026
Northern	San Dimas	MS4 Stormwater[1]	28.5	0.204	0.109
Northern	Angeles National Forest	Stormwater[1]	293	0.234	0.125
Southern	Caltrans	State Highway Stormwater[1]	11.6	0.0960	0.051
Total				**31.1**	**16.6**

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations (see footnote #3).

[3] For these responsible jurisdictions, the concentration-based WLA will be used to evaluate compliance.

[4] Each mass-based and concentration-based wasteload allocations must be met at the point of discharge.

10.3.6.2 Load Allocations

Load allocations (LAs) are assigned to the non-Caltrans permittees in the southern subwatershed as well as park irrigation and atmospheric deposition. Table 10-14 summarizes the existing total mercury loads and LAs for these sources. The LAs are a 46.6 percent reduction from the existing loads. These loading values (in grams per year) represent the TMDL load allocations (Table 10-14) and each load allocation must be met at the point of discharge. In addition to the LAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L applies.

Table 10-14. Load Allocations of Total Mercury to the Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Existing Annual Hg Load (g/yr)	Load Allocation[1] (g/yr)
Southern	La Verne	Runoff	0.097	0.0517
Southern	Pomona	Runoff	0.166	0.0887
Southern	San Dimas	Runoff	1.57	0.836
Southern	County of Los Angeles	Parkland Irrigation	4.55	2.43
Lake Surface		Atmospheric Deposition[2]	33.7	18.0
Total			**40.1**	**21.4**

[1] Each mass-based load allocations must be met at the point of discharge.
[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

10.3.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL includes both an implicit and explicit MOS for Puddingstone Reservoir. The implicit MOS includes comparing the total mercury concentration reported for fish tissue samples to the methylmercury fish tissue target. Most mercury in fish tissue is in the methyl form, but not all, so this is a conservative assumption.

In this TMDL, an explicit MOS is also included by selecting the 95 percent UCL to represent the existing mean fish tissue concentration rather than the regression predicted mean (Figure 10-13). Use of the UCL provides a margin of safety because it represents an upper confidence bound on the long-term exposure concentration. For Puddingstone Reservoir, the fraction of the existing load set aside for the explicit MOS is 0.074, or 5.3 g/yr, which represents 12 percent of the loading capacity.

10.3.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target in the northern lake system and maintaining existing water quality in the southern lake system. Because fish bioaccumulate mercury, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, annual mercury loading is more important for the attainment of standards than instantaneous or daily concentrations, and the TMDL is proposed in terms of annual loads. Mercury load is primarily delivered to the reservoir during storm runoff events, so high flows do represent a critical in terms of peak loading rates.

However, the greatest impact to fish occurs when methylmercury, a more biologically available form of mercury, is at its greatest concentration. Bacterially mediated methylation of mercury varies seasonally and typically results in the greatest methylmercury concentrations in the water column in the late summer. However, the impact of seasonal and other short-term variability in loading is damped out by the biotic response since the target concentrations in tissues of edible sized game fish integrate exposure over a number of years. Additionally, this TMDL includes a methylmercury water column target applicable year round. This TMDL therefore protects for critical conditions.

10.3.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. Although it is long-term cumulative load rather than daily loads of mercury that are driving the bioaccumulation of mercury in fish in Puddingstone Reservoir, this TMDL does present a maximum daily load according to the guidelines provided by USEPA (2007). The daily maximum allowable load of mercury to Puddingstone Reservoir is calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) to the reservoir multiplied by the allowable concentration for mercury consistent with achieving the long-term loading target. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDL must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for this San Dimas Creek gage (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99th percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The event mean concentration for mercury was calculated from the allowable load (38.0 g-Hg/yr; sum of the WLAs and LAs presented in Table 10-13 and Table 10-14, respectively) and the average annual simulated stream flow generated by the LSPC model (2,692 ac-ft). The resulting concentration (11.4 ng/L) times the peak flow to Puddingstone Reservoir (38.2 cfs) yields a total maximum daily load of 1.06 g-Hg/d. For comparison, the existing load (71.2 g-Hg/yr) would yield an event mean concentration of 21.4 ng/L and a total maximum daily load of 2.0 g-Hg/d. As described above, in order to achieve fish tissue targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

10.3.6.6 Future Growth

Much of the Puddingstone Reservoir watershed remains in shrub and brush rangeland. As development occurs in this watershed, best management practices (BMPs) will be required such that loading rates are consistent with the allocations established by this TMDL. Therefore, no load allocation has been set aside for future growth.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.4 PCBs Impairment

Polychlorinated biphenyls (PCBs) consist of a family of many related congeners. The individual congeners are often referred to by their "BZ" number. Environmental analyses may address individual congeners, homologs (groups of congeners with the same number of chlorine atoms), equivalent concentrations of the commercial mixtures of PCBs known by the trade name Aroclors, or total PCBs. The environmental measurements and targets described in this section are in terms of total PCBs, defined as the "sum of all congener or isomer or homolog or Aroclor analyses" (CTR, 40 CFR 131.38(b)(1) footnote v).

The PCB impairment of Puddingstone Reservoir affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. PCBs are no longer in production. While some loading of PCBs continues to occur in watershed runoff, the primary source of PCBs in the water column and aquatic life in Puddingstone Reservoir is from historic loads stored in the lake sediments. Like other organochlorine compounds, PCBs accumulate in aquatic organisms and biomagnify in the food chain. As a result, low environmental exposure concentrations can result in unacceptable levels in higher trophic level fish in the lake.

10.4.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of PCBs are currently impairing the REC1, REC2, WARM, and COLD uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD, MUN, GWR and RARE uses could become impaired.

10.4.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of PCBs listed in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets used for PCBs are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for PCBs in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0005 mg/L, or 0.5 μg/L, total PCBs in water. The Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this report is designated WARM, at a minimum, and must meet this requirement. A chronic criterion for the sum of PCB compounds in freshwater systems to protect aquatic life is included in the CTR as 0.014 µg/L (USEPA, 2000a). The CTR also provides a human health-based water quality criterion for the consumption of both water and organisms and organisms only of 0.00017 µg/L (0.17 ng/L). The human health criterion of 0.17 ng/L is the most restrictive applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) for total PCBs in sediment is 59.8 µg/kg dry weight dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The existing sediment PCB concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for PCBs defined by OEHHA (2008) is 3.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For total PCBs, the corresponding sediment concentration target determined using the BSAF is 0.59 µg/kg dry weight, as described in detail in Section 10.4.5. All applicable targets are shown below in Table 10-15. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 10-15. PCB Targets Applicable to Puddingstone Reservoir

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	3.6
Sediment (µg/kg dry weight)	Consensus-based TEC	59.8
Sediment (µg/kg dry weight)	BSAF-derived target	0.59
Water (ng/L)	CTR	0.17

Note: Shaded cells represent the selected targets for this TMDL.

10.4.3 Summary of Monitoring Data

This section summarizes the monitoring data for Puddingstone Reservoir related to the PCB impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

For PCBs, as well as other organochlorine compounds, sample analyses include both a detection limit and a reporting limit. For example, a typical detection limit for total PCBs in sediment analyzed by UCLA is 0.5 µg/kg dry weight, while the reporting limit is 5 µg/kg dry weight.

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the fall of 2008. Of four samples (two in Live Oak Wash and two in-lake stations), PCBs were below detection limits (1.5 ng/L to 1.52 ng/L) in two samples; in one of the Live Oak Wash samples PCB congeners were detected, but below reporting limits of 15.23 ng/L. One in-lake station had a reportable measurement (17.95 ng/L) of the PCB congener BZ-5.

Water samples from Puddingstone Reservoir were also collected by USEPA and/or the Regional Board on November 18, 2008 at five stations (four in-lake stations and one station in Live Oak Wash), February 24, 2009 at one storm drain station, and July 16, 2009 at four stations (Live Oak Wash, a storm drain, and two in-lake locations). PCBs at all stations were generally below the detection limit of 1 ng/L with three exceptions, including an in-lake concentration of 555 ng/L in November 2008, which is above the CTR water column target of 0.17 ng/L. A summary of the water column data is shown in Table 10-16.

Table 10-16. Summary of Water Column Samples for PCBs in Puddingstone Reservoir

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples above Detection Limit	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	[2.78][1]	4	1	1
PR-14 (Northeast Reservoir Side)	(0.63)	2	0	0
PR-15 (Western Reservoir Side)	191	3	2	0
PR-16 (Southern Reservoir Side)	(0.5)[2]	2	0	0
PR-17 (Western Reservoir Side near Shoreline)	(0.5)	1	0	0
PR-SD (Storm drain in northeast reservoir area)	(0.5)	1	0	0
PR-SD2 (Storm drain in northeast reservoir area)	(0.5)	1	0	0
In-Lake Average[2] (PR-14, 15, 16, 17)	48.20			
CTR Water Column Target	0.17			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Pollutant concentrations associated with suspended sediments in the lake were analyzed at two in-lake stations as well as Live Oak Wash during the fall of 2008 by UCLA. During the dry weather sampling event, PCBs were detected but below reporting limits (2.11 µg/kg to 36.23 µg/kg dry weight) at each location.

A wet weather composite sample at Live Oak Wash, did not detect any total PCB concentrations (detection limit of 1.57 µg/kg dry weight); an additional grab sample at the outlet of Live Oak Wash was collected 90 minutes into the wet weather and had no detectable concentration of total PCBs (detection limit of 2.70 µg/kg dry weight). Water column samples were also collected during this event (a time series composite and a single time point sample), but not analyzed.

PCBs were analyzed for three porewater samples collected at two in-lake stations. Each sample detected PCB-31; however, concentrations were less than the reporting limit (150 ng/L). Total suspended solids from the porewater samples were also analyzed for PCBs. Two samples were less than the detection limit (0.20 µg/kg to 0.53 µg/kg dry weight) for all PCB congeners. One sample detected PCB-31 at levels less than the reporting limit (3.01 µg/kg dry weight).

UCLA also collected bed sediment samples at two in-lake locations (total of three individual samples) in Puddingstone Reservoir in fall 2008. PCB congeners were detected in one sediment sample (average 10.8 µg/kg dry weight at PR-14), while the other samples were below detection limits (0.39 µg/kg to 1.58 µg/kg dry weight).

Sediment sampling was also conducted by USEPA and the Regional Board at six stations on July 16, 2009 (Live Oak Wash, two in-lake stations, two storm drain stations, and one natural drainage). PCBs were quantified at five of the six stations (one of the stormdrain samples had a concentration of 194.7 µg/kg dry weight and exceeded the sediment consensus-based TEC of 59.8 µg/kg dry weight). A summary of the sediment data is shown in Table 10-17. The lake-wide average of 4.99 µg/kg dry weight is below the concentration associated with inputs (50.3 µg/kg dry weight), and both are less than the consensus-based TEC of 59.8 µg/kg dry weight.

Table 10-17. Summary of Sediment Samples for PCBs in Puddingstone Reservoir

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limit	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	5.1	1	1	0
PR-14 (Northeast Reservoir Side)	5.4	2	1	0
PR-15 (Western Reservoir Side)	[3.67][1]	2	1	1
PR-16 (Southern Reservoir Side)	[5.75]	2	2	2
PR-19 (Natural drainage on South Side)	(0.50)	1	0	0
PR-19SD (Storm drain on South Side)	194.7	1	1	0
PR-SD2 (Storm drain in northeast reservoir area)	[1.00]	1	1	1
In-Lake Average[2] (PR-14, 15, 16)	4.99			
Influent Average	50.3			
Consensus-based TEC	59.8			

[1] Total PCBs in a sample represents the sum of all quantified PCB congeners, including results reported below the method reporting limit. If all congeners were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no PCBs were quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Eight fish samples (composites of filets from five fish) were collected and analyzed for PCBs as Aroclor equivalents between 1986 and 1999. In 1986, a largemouth bass and common carp sample reported 0 ppb (the detection limits for the historical fish samples are not reported) and 590 ppb wet weight, respectively, while in 1987 another common carp sample had a concentration of 160 ppb wet weight and a bullhead sample reported a zero concentration. In 1988, the reported concentration associated with a brown bullhead sample was 66 ppb wet weight. Three largemouth bass samples had concentrations of 54 ppb, 65 ppb, and 13 ppb wet weight in 1991, 1992, and 1999, respectively. The average reported PCB concentration in all samples from the 1980s and 1990s was 118.5 ppb, including the reported zeros. Results from the individual samples are shown in Appendix G (Monitoring Data).

More recently, SWAMP collected samples in September 2004 and June 2007. Considering only data collected in the past 10 years, the average concentration of total PCBs in largemouth bass was 20.6 ppb wet weight (average lipid fraction of 0.98 percent) and the average concentration of total PCBs in common carp was 30.2 ppb wet weight (average lipid fraction of 3.6 percent). The recent fish-tissue data for Puddingstone Reservoir are summarized in Table 10-18.

Table 10-18. Summary of Recent Fish Tissue Samples for PCBs in Puddingstone Reservoir

Sample Date	Fish Species	Total PCBs (ppb wet weight)[1]
9/22/2004	Largemouth Bass	29.1
9/22/2004	Largemouth Bass	16.0
9/22/2004	Largemouth Bass	35.9
9/22/2004	Largemouth Bass	17.9
9/22/2004	Common Carp	6.5
9/22/2004	Common Carp	49.3
9/22/2004	Common Carp	36.8
9/22/2004	Common Carp	28.3
6/6/2007	Largemouth Bass	18.7
6/6/2007	Largemouth Bass	5.9
2004-2007 Average - Largemouth Bass		20.6
2004 Average - Common Carp		30.2
FCG		3.6

[1] Composite sample of filets from either five (largemouth bass) or three individuals (carp).

In sum, recent fish tissue samples collected from Puddingstone are all elevated above OEHHA fish consumption guidelines for total PCBs. Concentrations in sediment are, on average, below the consensus-based TEC, although an individual sample exceeded this value. Concentrations in water were above detection limits in two samples (out of 14 individual samples); however, all of the detection limits exceeded the CTR criterion.

10.4.4 Source Assessment

PCBs in Puddingstone Reservoir are primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed PCB concentrations on sediment near inflows to the lake.

Watershed loads of PCBs may arise from spills from industrial and commercial uses, improper disposal, and atmospheric deposition. Industrial and commercial spills will tend to be associated with specific land areas, such as older industrial districts, junk yards, and transformer substations. Improper disposal could have occurred at various locations (indeed, waste PCB oils were sometimes used for dust control on dirt roads in the 1950s). Atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated PCB load within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas, while sources of water that do not contribute sediment load, such as irrigation, are considered to provide no significant PCB loading. The average concentration of PCBs on incoming sediment was estimated to be 50.3 µg/kg dry weight (Table 10-17) and the estimated annual sediment load to Puddingstone Reservoir is 265.5 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load is approximately 12.1 g/yr. Table 10-19 shows the annual PCB load estimated from each jurisdiction.

Table 10-19. Total PCB Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total PCB Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	13.5	0.62	5.10%
Northern	Claremont	MS4 Stormwater[1]	4.5	0.20	1.69%
Northern	County of Los Angeles	MS4 Stormwater[1]	27.7	1.30	10.44%
Northern	La Verne[2]	MS4 Stormwater[1]	168	7.68	63.23%
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	24.8	1.13	9.34%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	4.5	0.21	1.69%
Northern	Pomona	MS4 Stormwater[1]	0.5	0.02	0.18%
Northern	San Dimas	MS4 Stormwater[1]	1.6	0.07	0.62%
Northern	Angeles National Forest	Stormwater[1]	1.4	0.06	0.51%
Southern	Caltrans	State Highway Stormwater[1]	1.4	0.06	0.54%
Southern	La Verne	Runoff	1.2	0.06	0.47%
Southern	Pomona	Runoff	1.7	0.08	0.63%

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total PCB Load (g/yr)	Percent of Total Load
Southern	San Dimas	Runoff	14.8	0.68	5.59%
Southern	County of Los Angeles	Parkland Irrigation	0.0	0.00	0.00%
Total Load from Watershed			**265.5**	**12.12**	**100.00%**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of PCBs directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

10.4.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of PCBs into Puddingstone Reservoir consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of PCBs in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. PCBs are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of PCBs will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Puddingstone Reservoir are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment PCB concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to the additional uptake of organochlorine compounds from the sediment by bottom-feeding fish. For PCBs in Puddingstone Reservoir the ratios of the FCG to the existing fish concentrations (Table 10-18) are:

$$\text{TL4: } 3.6/20.6 = 0.1750$$

$$\text{TL3: } 3.6/30.2 = 0.1191$$

The lower ratio, obtained for the TL3 fish, is applied to the observed in-lake sediment concentration of 4.99 µg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 0.59 µg/kg dry weight. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of PCBs are likely to have declined steadily since the cessation of production and use of the chemical. The resulting fish tissue-based target concentrations of PCBs in the sediment of Puddingstone Reservoir is shown in Table 10-20.

Table 10-20. Fish Tissue-Based PCB Concentration Targets for Sediment in Puddingstone Reservoir

Total PCB Concentration	Sediment (µg/kg dry weight)
Existing	4.99
BSAF-derived Target	0.59
Required Reduction	88.2%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 59.8 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.17 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 2,245 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 12.12 g/yr, or 0.5 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of PCBs in Puddingstone Reservoir is primarily due to the storage of historic loads of PCBs in the lake sediment.

10.4.6 TMDL Summary

Because PCB impairment in Puddingstone Reservoir is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake. The PCB TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 0.59 µg/kg dry weight total PCBs. The wasteload allocations and load allocations are also equal to 0.59 µg/kg dry weight total PCBs in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

10.4.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for total PCBs ("Alternative WLAs if the Fish Tissue Target is Met") described in Section 10.4.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 10.4.6.1.1 if the conditions described in Section 10.4.6.1.2 are met.

10.4.6.1.1 Wasteload Allocations

In the Puddingstone Reservoir watershed, wasteload allocations (WLAs) are required for all permittees in the northern subwatershed and Caltrans areas in the southern subwatershed. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

PCBs in water flowing into Puddingstone Reservoir are below detection limits, and most PCB load is expected to move in association with sediment. Therefore, suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for PCBs in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved PCBs and PCBs associated with suspended sediment. The existing concentration on sediment entering the lake is 50.9 µg/kg dry weight. Therefore, a reduction of (50.3 – 0.59)/50.3 = 98.8 percent is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 10-21 and each wasteload allocation must be met at the point of discharge.

Table 10-21. Wasteload Allocations for Total PCBs in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	0.59	0.17
Northern	Claremont	MS4 Stormwater[1]	0.59	0.17
Northern	County of Los Angeles	MS4 Stormwater[1]	0.59	0.17
Northern	La Verne[2]	MS4 Stormwater[1]	0.59	0.17
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	0.59	0.17
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	0.59	0.17
Northern	Pomona	MS4 Stormwater[1]	0.59	0.17
Northern	San Dimas	MS4 Stormwater[1]	0.59	0.17
Northern	Angeles National Forest	Stormwater[1]	0.59	0.17

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Southern	Caltrans	State Highway Stormwater[1]	0.59	0.17

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

10.4.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 10-21 will be superseded, and the wasteload allocations in Table 10-22 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 10-22, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 10-22. Alternative Wasteload Allocations for Total PCBs in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	59.8	0.17
Northern	Claremont	MS4 Stormwater[1]	59.8	0.17
Northern	County of Los Angeles	MS4 Stormwater[1]	59.8	0.17
Northern	La Verne[2]	MS4 Stormwater[1]	59.8	0.17
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	59.8	0.17
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	59.8	0.17
Northern	Pomona	MS4 Stormwater[1]	59.8	0.17

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for PCBs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for PCBs in the Water Column[3] (ng/L)
Northern	San Dimas	MS4 Stormwater[1]	59.8	0.17
Northern	Angeles National Forest	Stormwater[1]	59.8	0.17
Southern	Caltrans	State Highway Stormwater[1]	59.8	0.17

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

10.4.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for total PCBs ("Alternative LAs if the Fish Tissue Target is Met") described in Section 10.4.6.2.2. The alternative load allocations will supersede the load allocations in Section 10.4.6.2.1 if the conditions described in Section 10.4.6.2.2 are met.

10.4.6.2.1 Load Allocations

Load allocations (LAs) are assigned to the non-Caltrans permittees in the southern subwatershed, and lake bottom sediments. Additionally, the TMDL establishes load allocations for PCBs in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved PCBs and PCBs associated with suspended sediment. No load is allocated to atmospheric deposition of PCBs. The legacy PCB stored in lake sediment is the major cause of use impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (County of Los Angeles) should achieve a PCB concentration of 0.59 µg/kg dry weight in lake bottom sediments (Table 10-23). Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-23. Load Allocations for Total PCBs in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Load Allocation for PCBs Associated with Suspended Sediment or Lake Bottom Sediments (µg/kg dry weight)
Southern	La Verne	Runoff[1]	0.59
Southern	Pomona	Runoff[1]	0.59
Southern	San Dimas	Runoff[1]	0.59
Southern	County of Los Angeles	Parkland Irrigation[1]	0.59
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	0.59

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.4.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 10-23 will be superseded, and the load allocations in Table 10-24 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 3.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 10-24, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-24. Alternative Load Allocations for Total PCBs in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	59.8
Southern	Pomona	Runoff[1]	59.8
Southern	San Dimas	Runoff[1]	59.8
Southern	County of Los Angeles	Parkland Irrigation[1]	59.8
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	59.8

[1] Each load allocation must be met at the point of discharge.
[2] The load allocation must be met in the lake.

10.4.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

10.4.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate PCBs, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

10.4.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the PCB WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99^{th} percentile flow for San Dimas Creek (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99^{th} percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The event mean concentration of sediment in stormwater (45.5 mg/L) was calculated from the estimated existing watershed sediment load of 265.5 tons/yr (Table 10-19) divided by the total annual wet weather flow volume delivered to the lake (4,295 ac-ft/yr). Multiplying the sediment event mean concentration by the 99^{th} percentile peak daily flow (38.2 cfs) yields a daily maximum sediment load from stormwater of 4,249 kg/d (4.7 tons/d). Applying the wasteload allocation concentration of 0.59 µg total PCBs per dry kg of sediment yields the stormwater daily maximum allowable load of 0.0025 g/d of total PCBs. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

10.4.6.6 Future Growth

USEPA regulates PCBs under the Toxic Substances Control Act (TSCA), which generally bans the manufacture, use, and distribution in commerce of the chemicals in products at concentrations of 50 parts per million or more, although TSCA allows USEPA to authorize certain uses, such as to rebuild existing electrical transformers during the transformers' useful life. Therefore, no additional allowance is made for future growth in the PCB TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.5 CHLORDANE IMPAIRMENT

Total chlordane consists of a family of related chemicals, including cis- and trans-chlordane, oxychlordane, trans-nonachlor, and cis-nonachlor. Observations and targets discussed in this section all refer to total chlordane. Chlordane was used as a pesticide in field, commercial, and residential uses. Chlordane is no longer in production, but persists in the environment from legacy loads.

The chlordane impairment of Puddingstone Reservoir affects the beneficial uses related to recreation, municipal water supplies, wildlife health, and fish consumption. While some loading of chlordane continues to occur in watershed runoff, the primary source of chlordane in the water column and aquatic life in Puddingstone Reservoir is from historic loads stored in the lake sediments. Chlordane, like other organochlorine compounds, accumulates in aquatic organisms and biomagnifies in the food chain. As a result, low environmental concentrations can result in unacceptable levels in higher trophic level fish in the lake. The approach for chlordane is similar to that for PCBs.

10.5.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of chlordane are currently impairing the REC1, REC2, WARM, and COLD uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impairing sport fishing recreational uses. At high enough concentrations WILD, MUN, GWR and RARE uses could become impaired.

10.5.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of chlordane in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), for chlordane defined by the Office of Environmental Health Hazard Assessment (OEHHA) for fish consumption. The numeric targets used for chlordane are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for chlordane in the Basin Plan are associated with a specific beneficial use. For waters designated MUN, the Basin Plan lists a maximum contaminant level of 0.0001 mg/L, or 0.1 µg/L. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Puddingstone Reservoir is also designated WARM, COLD, and RARE, and must at least meet this requirement. Acute and chronic criteria for chlordane in freshwater systems are defined by the California Toxics Rule as 2.4 µg/L and 0.0043 µg/L, respectively (USEPA, 2000a). The CTR also includes human health criteria for the consumption of water and organisms and for the consumption of organisms only as 0.00057 µg/L and 0.00059 µg/L, respectively (USEPA, 2000a). Because the human health criterion for the consumption of water and organisms is the most restrictive criterion applicable to Puddingstone Reservoir, a water column target of 0.00057 µg/L (0.57 ng/L) is the appropriate target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) for chlordane is 3.24 µg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." The existing sediment chlordane concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for chlordane defined by OEHHA (2008) is 5.6 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For chlordane, the corresponding sediment concentration target determined using the BSAF is 0.75 µg/kg dry weight, as described in Section 10.5.5. All applicable targets are shown below in Table 10-25. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 10-25. Total Chlordane Targets for Puddingstone Reservoir

Media	Source	Target
Fish (ppb wet weight)	OEHHA FCG	5.6
Sediment (ng /dry g)	Consensus-based TEC	3.24
Sediment (µg/kg dry weight)	BSAF-derived target	0.75
Water (ng/L)	CTR	0.57

Note: Shaded cells represent the selected targets for this TMDL.

10.5.3 Summary of Monitoring Data

This section summarizes the monitoring data related to the chlordane impairment in Puddingstone Reservoir. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the fall of 2008. These analyses measured cis- and trans-chlordane, but not oxychlordane or nonachlor. Of four samples (two in Live Oak Wash and two in-lake stations), chlordane was consistently below the detection limits (1.5 ng/L to 1.57 ng/L; the detection limit for chlordane is higher than the water column criterion of 0.57 ng/L).

Water samples from Puddingstone Reservoir were also collected by USEPA and/or the Regional Board on November 18, 2008 at five stations (four in-lake stations and one station in Live Oak Wash) and July 16, 2009 at four stations (Live Oak Wash, a storm drain, and two in-lake locations). These analysis did include oxychlordane and nonachlor. Chlordane concentrations at all stations were below the detection limit of 1 ng/L, which is above the CTR water column target of 0.57 ng/L. A summary of the water column data is shown in Table 10-26.

Table 10-26. Summary of Water Column Samples for Total Chlordane in Puddingstone Reservoir

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
PR-11 (Live Oak Channel)	(0.63)[2]	4	0
PR-14 (Northeast Reservoir Side)	(0.63)	2	0

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples Above Detection Limits[1]
PR-15 (Western Reservoir Side)	(0.60)	3	0
PR-16 (Southern Reservoir Side)	(0.50)	2	0
PR-17 (Western Reservoir Side near Shoreline)	(0.50)	1	0
PR-SD2 (Storm drain in northeast reservoir area)	(0.50)	1	0
In-Lake Average (PR-14, 15, 16, 17)[3]	(0.56)		
CTR Water Column Target	0.57		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that the sample is based only on the detection limits of the samples, and that no chlordanes were quantified in any of the collected samples.

[3] Overall average is the average of individual station averages.

Pollutant concentrations associated with suspended sediments in the lake were analyzed at two in-lake stations as well as Live Oak Wash during the fall of 2008 by UCLA. Concentrations of chlordane in the suspended sediment samples were less than the detection limits (2 µg/kg to 36 µg/kg dry weight) at the two in-lake stations; chlordane was detected but not at reportable amounts in the Live Oak Wash suspended sediment sample. A grab sample at the outlet of Live Oak Wash that was collected 90 minutes into a wet weather event had no detectable results (detection limit of 2.70 µg/kg dry weight); the composite sample for this event was also less than the detection limit (1.57 µg/kg dry weight). Water column samples were collected during this event (a time series composite and a single time point sample) as well, but not analyzed. Chlordane concentrations were also analyzed in porewater; all samples were less than the detection limit of 15 ng/L. The suspended sediments associated with the porewater had concentrations less than detection limits (0.2 µg/kg to 0.53 µg/kg dry weight).

UCLA collected bed sediment samples at two in-lake locations (total of three individual samples) in Puddingstone Reservoir in fall 2008. As with the water column analyses by UCLA, these report cis- and trans-chlordane, but not oxychlordane or nonachlor. Total chlordane was consistently below detection limits (0.39 µg/kg to 1.58 µg/kg dry weight). Sediment sampling was conducted by USEPA and the Regional Board at six stations on July 16, 2009 (Live Oak Wash, two in-lake stations, two storm drain stations, and one natural drainage). Total chlordane (including oxychlordane and nonachlor) was quantified at each of the six stations with values ranging from 1.1 µg/kg to 6.5 µg/kg dry weight (two of the six samples had a concentration exceeding the sediment consensus-based TEC of 3.24 µg/kg dry weight). A summary of the sediment data is shown in Table 10-27. The lake-wide average of 2.15 µg/kg dry weight is less than the concentration associated with inputs (5.11 µg/kg dry weight), and the lake-wide average is less than the consensus-based TEC of 3.24 µg/kg dry weight.

Table 10-27. Summary of Sediment Samples for Total Chlordane in Puddingstone Reservoir

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	10.15	1	1	0
PR-14 (Northeast Reservoir Side)	(0.22)	2	0	0

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
PR-15 (Western Reservoir Side)	3.77	2	1	0
PR-16 (Southern Reservoir Side)	[2.45]	2	1	1
PR-19 (Natural Drainage on South Side)	[2.20]	1	1	1
PR-19SD (Storm Drain on South Side)	[4.50]	1	1	1
PR-SD2 (Storm drain in northeast reservoir area)	[3.60]	1	1	1
In-Lake Average[2] (PR-14, 15, 16)	2.15			
Influent Average	5.11			
Consensus-based TEC	3.24			

[1] Total chlordane in a sample represents the sum of all reported measurements for alpha and gamma chlordane, oxychlordane, and cis- and trans-nonachlor, including results reported below the method reporting limit. If all components were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no chlordane quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

Fish tissue concentrations of total chlordane from Puddingstone Reservoir have been analyzed in largemouth bass, common carp, bullhead, and brown bullhead (SWAMP and TSMP). Eight fish samples (composites of filets from five fish) were collected and analyzed for total chlordane between 1986 and 1999. In 1986, a largemouth bass and common carp sample reported 10.4 ppb and 460 ppb wet weight, respectively, while in 1987 another common carp sample had a concentration of 193.5 ppb wet weight and a bullhead sample reported a concentration of 44.4 ppb wet weight. In 1988, the reported concentration associated with a brown bullhead sample was 48.5 ppb wet weight. Three largemouth bass samples had concentrations of 16.1 ppb, 31.7 ppb, and 2.8 ppb wet weight in 1991, 1992, and 1999, respectively. The average reported chlordane concentration in all samples from the 1980s and 1990s was 100.9 ppb wet weight. Results from the individual samples are shown in Appendix G (Monitoring Data).

More recently, SWAMP collected samples in September 2004 and June 2007. Considering only data collected in the past 10 years, the average concentration of total chlordane in largemouth bass was 8.7 ppb wet weight (average lipid fraction of 0.98 percent), and the average concentration of total chlordane in common carp was 30.2 ppb wet weight (average lipid fraction of 3.6 percent). The recent fish-tissue data for Puddingstone Reservoir are summarized in Table 10-28.

Table 10-28. Summary of Recent Fish Tissue Samples for Total Chlordane in Puddingstone Reservoir

Sample Date	Fish Species	Total Chlordane (ppb wet weight)[1]
9/22/2004	Largemouth Bass	12.4
9/22/2004	Largemouth Bass	5.9

Sample Date	Fish Species	Total Chlordane (ppb wet weight)[1]
9/22/2004	Largemouth Bass	13.6
9/22/2004	Largemouth Bass	7.3
9/22/2004	Common Carp	1.2
9/22/2004	Common Carp	27.3
9/22/2004	Common Carp	20.0
9/22/2004	Common Carp	15.6
6/6/2007	Largemouth Bass	9.3
6/6/2007	Largemouth Bass	3.8
2004-2007 Average - Largemouth Bass		8.7
2004 Average - Common Carp		16.0
FCG		5.6

[1] Composite sample of filets from five (largemouth bass) or three individuals (common carp).

In sum, a majority (80 percent) of recent fish tissue samples collected from Puddingstone are elevated above OEHHA fish consumption guidelines for total chlordane (5.6 ppb; the average concentration is also above the FCG). Concentrations in sediment are, on average, below the consensus-based TEC, although individual samples exceeded this value. Water column samples have all been below detection limits; however, all of the detection limits exceeded the CTR criterion.

10.5.4 Source Assessment

Chlordane in Puddingstone Reservoir is primarily due to historical loading and storing within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed chlordane concentrations on sediment near inflows to the lake. Watershed loads of chlordane may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas, while sources of water that do not contribute sediment load, such as irrigation, are considered to provide no significant chlordane loading. The average concentration of total chlordane on incoming sediment is estimated to be 5.11 µg/kg dry weight (Table 10-27) and the annual sediment load to Puddingstone Reservoir is 265.5 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet weather load of chlordane is approximately 1.23 g/yr (Table 10-29).

Table 10-29. Total Chlordane Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total Chlordane Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	13.5	0.063	5.10%
Northern	Claremont	MS4 Stormwater[1]	4.5	0.021	1.69%
Northern	County of Los Angeles	MS4 Stormwater[1]	27.7	0.128	10.43%
Northern	La Verne[2]	MS4 Stormwater[1]	168	0.778	63.22%
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	24.8	0.115	9.34%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	4.5	0.021	1.69%
Northern	Pomona	MS4 Stormwater[1]	0.5	0.002	0.18%
Northern	San Dimas	MS4 Stormw[1]ater	1.6	0.008	0.62%
Northern	Angeles National Forest	Stormwater[1]	1.4	0.006	0.51%
Southern	Caltrans	State Highway Stormwater[1]	1.4	0.007	0.54%
Southern	La Verne	Runoff	1.2	0.006	0.47%
Southern	Pomona	Runoff	1.7	0.008	0.63%
Southern	San Dimas	Runoff	14.8	0.069	5.59%
Southern	County of Los Angeles	Parkland Irrigation	0.0	0.000	0.00%
Total Load from Watershed			**265.5**	**1.23**	**100.00%**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of total chlordane directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load.

10.5.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of total chlordane into Puddingstone Reservoir. The loading capacity is used to estimate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of total chlordane in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Chlordanes are strongly sorbed to sediments and have long half-lives in sediment and water. Incoming loads of total chlordane will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Puddingstone Reservoir are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment chlordane concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to the additional uptake of organochlorine compounds from the sediment by bottom-feeding fish. For chlordane in Puddingstone Reservoir, the ratios of the FCG to the existing fish concentrations (Table 10-27) are:

$$TL4: 5.6/8.7 = 0.6424$$

$$TL3: 5.6/16.0 = 0.3500$$

The lower ratio, obtained for the TL3 fish, is applied to the observed sediment concentration of 2.15 µg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 0.75 µg/kg dry weight.

The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of total chlordane are likely to have declined steadily since the cessation of production and use of the chemicals. The resulting fish tissue-based target concentration of total chlordane in the sediment of Puddingstone Reservoir is shown in Table 10-30.

Table 10-30. Fish Tissue-Based Chlordane Concentration Targets for Sediment in Puddingstone Reservoir

Total Chlordane Concentration	Sediment (µg/kg dry weight)
Existing	2.15
BSAF-derived Target	0.75
Required Reduction	65.1%

The BSAF-derived sediment target is less than the consensus-based TEC of 3.24 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.57 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 1,379 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 1.23 g/yr, or 0.09 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of chlordane in Puddingstone Reservoir is primarily due to the storage of historic loads of chlordane in the lake sediment.

10.5.6 TMDL Summary

Because chlordane impairment in Puddingstone Reservoir is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake.

The chlordane TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 0.75 µg/kg dry weight chlordane. The wasteload allocations and load allocations are also equal to 0.75 µg/kg dry weight chlordane in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

10.5.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for chlordane (“Alternative WLAs if the Fish Tissue Target is Met”) described in Section 10.5.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 10.5.6.1.1 if the conditions described in Section 10.5.6.1.2 are met.

10.5.6.1.1 Wasteload Allocations

In the Puddingstone Reservoir watershed, wasteload allocations (WLAs) are required for all permittees in the northern subwatershed and Caltrans areas in the southern subwatershed. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Total chlordane concentrations in water flowing into Puddingstone Reservoir are below detection limits, and most chlordane load is expected to move in association with sediment. Therefore, suspended sediment in the water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for chlordane in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved chlordane and chlordane associated with suspended sediment. The existing concentration of sediment entering the lake is 5.11 µg/kg dry weight. Therefore, a reduction of (5.11 – 0.75)/5.11 = 85.3 percent is required on the sediment-associated load from the watershed. The reduction in watershed load is greater than the reduction needed for in-lake sediments because the estimated concentration on influent sediment is greater than the lake-wide average.

The wasteload allocations are shown in Table 10-31 and each wasteload allocation must be met at the point of discharge.

Table 10-31. Wasteload Allocations for Total Chlordane in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	0.75	0.57
Northern	Claremont	MS4 Stormwater[1]	0.75	0.57
Northern	County of Los Angeles	MS4 Stormwater[1]	0.75	0.57
Northern	La Verne[2]	MS4 Stormwater[1]	0.75	0.57
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	0.75	0.57
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	0.75	0.57
Northern	Pomona	MS4 Stormwater[1]	0.75	0.57
Northern	San Dimas	MS4 Stormwater[1]	0.75	0.57
Northern	Angeles National Forest	Stormwater[1]	0.75	0.57
Southern	Caltrans	State Highway Stormwater[1]	0.75	0.57

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

10.5.6.1.2 Alternative Wasteload Allocations if Fish Tissue Targets Are Met

The wasteload allocations listed in Table 10-31 will be superseded, and the wasteload allocations in Table 10-32 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include

a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,

2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 10-32, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 10-32. Alternative Wasteload Allocations for Total Chlordane in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total Chlordane Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Chlordane in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	3.24	0.57
Northern	Claremont	MS4 Stormwater[1]	3.24	0.57
Northern	County of Los Angeles	MS4 Stormwater[1]	3.24	0.57
Northern	La Verne[2]	MS4 Stormwater[1]	3.24	0.57
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	3.24	0.57
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	3.24	0.57
Northern	Pomona	MS4 Stormwater[1]	3.24	0.57
Northern	San Dimas	MS4 Stormwater[1]	3.24	0.57
Northern	Angeles National Forest	Stormwater[1]	3.24	0.57
Southern	Caltrans	State Highway Stormwater[1]	3.24	0.57

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

10.5.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for chlordane ("Alternative LAs if the Fish Tissue Target is Met") described in Section 10.5.6.2.2. The alternative load allocations will supersede the load allocations in Section 10.5.6.2.1 if the conditions described in Section 10.5.6.2.2 are met.

10.5.6.2.1 Load Allocations

Load allocations (LAs) are assigned to the non-Caltrans permittees in the southern subwatershed and lake bottom sediments. Additionally, the TMDL establishes load allocations for chlordane in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved chlordane and chlordane associated with suspended sediment. No load is allocated to atmospheric deposition of total chlordane. The legacy chlordane stored in lake sediment is the major cause of use impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (County of Los Angeles) should achieve a total chlordane concentration of 0.75 µg/kg dry weight in lake bottom sediments (Table 10-33). Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-33. Load Allocations for Total Chlordane in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Load Allocation for Chlordane Associated with Suspended Sediment or Lake Bottom Sediments (µg/kg dry weight)
Southern	La Verne	Runoff[1]	0.75
Southern	Pomona	Runoff[1]	0.75
Southern	San Dimas	Runoff[1]	0.75
Southern	County of Los Angeles	Parkland Irrigation[1]	0.75
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	0.75

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.5.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 10-33 will be superseded, and the load allocations in Table 10-34 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 5.6 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 10-34, and
3. USEPA does not object to the Regional Board’s determination within 60 days of receiving notice of it.

Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-34. Alternative Load Allocations for Total Chlordane in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	3.24
Southern	Pomona	Runoff[1]	3.24
Southern	San Dimas	Runoff[1]	3.24
Southern	County of Los Angeles	Parkland Irrigation[1]	3.24
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	3.24

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.5.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

10.5.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate chlordane, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

10.5.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the total chlordane WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99^{th} percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for San Dimas Creek (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99th percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The event mean concentration of sediment in stormwater (45.5 mg/L) was calculated from the estimated existing watershed sediment load of 265.5 tons/yr (Table 10-29) divided by the total annual wet weather flow volume delivered to the lake (4,295 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (38.2 cfs) yields a daily maximum sediment load from stormwater of 4,249 kg/d (4.7 tons/d). Applying the wasteload allocation concentration of 0.75 µg total chlordane per dry kg of sediment yields the stormwater daily maximum allowable load of 0.0032 g/d of total chlordane. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

10.5.6.6 Future Growth

The manufacture and use of chlordane is currently banned. Therefore, no additional allowance is made for future growth in the chlordane TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.6 DIELDRIN IMPAIRMENT

Dieldrin is a chlorinated insecticide originally developed as an alternative to DDT and was in wide use from the 1950s to the 1970s. Dieldrin in the environment also arises from use of the insecticide aldrin. Aldrin is not itself toxic to insects, but is metabolized to dieldrin in the insect body. The use of both dieldrin and aldrin was discontinued in the 1970s.

The dieldrin impairment of Puddingstone Reservoir affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. Dieldrin, like PCBs, chlordane and DDT, is an organochlorine compound that is strongly sorbed to sediment and lipids and is no longer in production. As such, the approach for dieldrin impairment is similar to that for PCBs, chlordane, and DDT.

10.6.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of dieldrin are impairing the REC1, REC2, WARM, and COLD uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories), and impair sport fishing recreational uses. At high enough concentrations WILD, MUN, GWR and RARE uses could become impaired.

10.6.2 Numeric Targets

The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses. There are no numeric criteria specified for sediment or fish tissue concentrations of dieldrin in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets for dieldrin are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for dieldrin in the Basin Plan are associated with a specific beneficial use. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Acute and chronic criterion for the protection of aquatic life in freshwater systems are included in the CTR for dieldrin as 0.24 µg/L and 0.056 µg/L, respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. The CTR also provides a human health-based water quality criterion for the consumption of organisms only and the consumption of water and organisms as 0.00014 µg/L (0.14 ng/L). The human health criterion of 0.00014 µg/L (0.14 ng/L) is the most restrictive of the applicable criteria specified for water column concentrations and is selected as the water column target.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) of dieldrin in sediment is 0.46 µg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. This target is designed to protect benthic dwelling organisms and explicitly does not consider “the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans).” The estimated existing sediment dieldrin concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for dieldrin defined by the OEHHA (2008) is 0.46 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For dieldrin, the corresponding sediment concentration target estimated using the BSAF approach is 0.22 µg/kg dry weight, as described in detail in Section 10.6.5. All applicable targets are shown below in Table 10-35. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 10-35. Dieldrin Targets Applicable to Puddingstone Reservoir

Medium	Source	Target
Fish (ppb wet weight)	OEHHA FCG	0.46
Sediment (µg/kg dry weight)	Consensus-based TEC	1.9
Sediment (µg/kg dry weight)	BSAF-derived target	0.22
Water (ng/L)	CTR	0.14

Note: Shaded cells represent the selected targets for this TMDL.

10.6.3 Summary of Monitoring Data

This section summarizes the monitoring data for Puddingstone Reservoir related to the dieldrin impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the fall of 2008. All four samples (two in Live Oak Wash and two in-lake stations) were below detection limits for dieldrin (3.05 ng/L to 3.14 ng/L).

Water samples from Puddingstone Reservoir were also collected by USEPA and/or the Regional Board on November 18, 2008 at five stations (four in-lake stations and one station in Live Oak Wash), February 24, 2009 at one storm drain station, and July 16, 2009 at four stations (Live Oak Wash, one storm drain, and two in-lake locations). Dieldrin concentrations at all stations were below the detection limit of 1 ng/L. Although no water column samples have had detectable quantities of dieldrin, the detection limits for these samples (1 ng/L or greater) are higher than the CTR water column target of 0.14 ng/L. A summary of the water column data is shown in Table 10-36.

Table 10-36. Summary of Water Column Samples for Dieldrin in Puddingstone Reservoir

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples above Detection Limit[1]	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	(1.01) [2]	4	0	0
PR-14 (Northeast Reservoir Side)	(1.01) [2]	2	0	0
PR-15 (Western Reservoir Side)	(0.86) [2]	3	0	0
PR-16 (Southern Reservoir Side)	(0.50) [2]	2	0	0
PR-17 (Western Reservoir Side near Shoreline)	(0.50) [2]	1	0	0
PR-SD (Storm drain in northeast reservoir area)	(0.50) [2]	1	0	0
PR-SD2 (Storm drain in northeast reservoir area)	(0.50) [2]	1	0	0
In-Lake Average[3] (PR-14, 15, 16, 17)	(0.72) [2]			
CTR Water Column Target	0.17			

[1] Non-detect samples were included in reported averages at one-half of the sample detection limi.

[2] Numbers in parentheses indicate that the sample is based only on the detection limits of the samples, and that no dieldrin was quantified in any of the collected samples.

[3] Overall average is the average of individual station averages.

Pollutant concentrations associated with suspended sediments in the lake were analyzed at two in-lake stations as well as Live Oak Wash during the fall of 2008 by UCLA, but did not quantify dieldrin at detectable limits (4 µg/kg to 72 µg/kg dry weight). A composite sample during a wet weather event did not detect any dieldrin (detection limit of 3.14 µg/kg dry weight). A grab sample at the outlet of Live

Oak Wash was collected 90 minutes into the wet weather event, which had no detectable results (detection limit of 5.39 µg/kg dry weight).

A wet weather composite sample at Live Oak Wash, did not detect any dieldrin (detection limit of 1.57 µg/kg dry weight); an additional grab sample at the outlet of Live Oak Wash was collected 90 minutes into the wet weather and had no detectable concentration of dieldrin (detection limit of 2.70 µg/kg dry weight). Water column samples were also collected during this event (a time series composite and a single time point sample), but not analyzed.

Dieldrin was analyzed for three porewater samples collected at two in-lake stations. Both samples were less than the detection limit of 30 ng/L. Total suspended solids from the porewater samples were also analyzed for dieldrin, but were less than detection limits of 0.4 – 1.06 µg/kg dry weight.

UCLA also collected bed sediment samples at two in-lake locations (total of three individual samples) in Puddingstone Reservoir in fall 2008. For dieldrin, all the samples were below detection limits (0.77 µg/kg to 3.17 µg/kg dry weight).

Sediment sampling was also conducted by USEPA and the Regional Board at six stations on July 16, 2009 (Live Oak Wash, two in-lake stations, two storm drain stations, and one natural drainage). All samples were less than a detection limit of 1 µg/kg dry weight for dieldrin. Because dieldrin does appear in fish at levels greater than the FCG, and because these body burdens of dieldrin are believed to arise from the sediment, EPA decided to represent statistical estimates for the sediment concentrations of dieldrin by setting the concentration of non-detected samples to the detection limit. A summary of the sediment data is shown in Table 10-37. The lake-wide average of <1.32 µg/kg dry weight for dieldrin is less than the consensus-based TEC of 5.28 µg/kg dry weight.

Table 10-37. Summary of Sediment Samples for Dieldrin in Puddingstone Reservoir

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits[1]	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	(1.0) [1]	1	0	0
PR-14 (Northeast Reservoir Side)	(0.89)	2	0	0
PR-15 (Western Reservoir Side)	(2.08)	2	0	0
PR-16 (Southern Reservoir Side)	(1.0)	1	0	0
PR-19 (Natural drainage on South Side)	(1.0)	1	0	0
PR-19SD (Storm drain on South Side)	(1.0)	1	0	0
PR-SD2 (Storm drain in northeast reservoir area)	(1.0)	1	0	0
In-Lake Average (PR-14,15, 16)[2]	(1.32) [1]			
Influent Average	(1.00)			
Consensus-based TEC	5.28			

[1] All sample results were below detection limits. An upper-bound analysis was performed using the reported sample detection limits for dieldrin. Numbers in parentheses indicate that sample is based only on the detection limits of the samples, and that no dieldrin was quantified in any of the collected samples.

[2] Overall average is the average of individual station averages.

Eight fish samples (composites of filets from five fish) were collected and analyzed for dieldrin between 1986 and 1999. All four largemouth bass and both bullhead samples were below detection limits (the detection limits for the historical fish samples are not reported). However, common carp samples collected in 1986 and 1988 had concentrations of 12 and 5 ppb wet weight, respectively. Results from the individual samples are shown in Appendix G (Monitoring Data).

More recently, SWAMP collected samples in September 2004 and June 2007. Considering only data collected in the past 10 years, the average concentration of dieldrin in largemouth bass was 1.2 ppb wet weight (average lipid fraction of 0.98 percent) and the average concentration of dieldrin in common carp was 2.7 ppb wet weight (average lipid fraction of 3.6 percent). The recent fish-tissue data for Puddingstone Reservoir are summarized in Table 10-38.

Table 10-38. Summary of Recent Fish Tissue Samples for Dieldrin in Puddingstone Reservoir

Sample Date	Fish Species	Dieldrin (ppb wet weight)[1]
9/22/2004	Largemouth Bass	1.7
9/22/2004	Largemouth Bass	0.9
9/22/2004	Largemouth Bass	1.6
9/22/2004	Largemouth Bass	1.2
9/22/2004	Common Carp	0.7
9/22/2004	Common Carp	4.3
9/22/2004	Common Carp	3.4
9/22/2004	Common Carp	2.5
6/6/2007	Largemouth Bass	0.7
6/6/2007	Largemouth Bass	(0.2) [2]
2004-2007 Average - Largemouth Bass		1.2
2004 Average - Common Carp		2.7
FCG		0.46

[1] Composite sample of filets from either five (largemouth bass) or three individuals (carp).

[2] Non-detect samples were included in reported averages at one-half of the sample detection limit (shown in parentheses).

In sum, all but one of the recent fish tissue samples collected from Puddingstone are elevated above OEHHA fish consumption guidelines for dieldrin. Concentrations in sediment are, on average, below the consensus-based TEC, although an individual sample exceeded this value. Concentrations in water were below detection limits; however, all of the detection limits exceeded the CTR criterion.

10.6.4 Source Assessment

Dieldrin present in Puddingstone Reservoir is primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading and direct atmospheric deposition to the lake are considered negligible sources of dieldrin. Stormwater loads from the watershed could not be directly estimated because all sediment and water samples were below detection limits. Watershed loads of dieldrin may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated dieldrin load within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas, while sources of water that do not contribute sediment load, such as irrigation, are considered to provide no significant dieldrin loading. The average concentration of total dieldrin on incoming sediment was estimated to be < 1.0 µg/kg dry weight Table 10-39 – based on the average detection limit of samples), and the annual sediment load to Puddingstone Reservoir is 265.5 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet-weather load of dieldrin is approximately 0.24 g/yr (Table 10-39).

Table 10-39. Dieldrin Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Dieldrin Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	13.5	0.012	5.10%
Northern	Claremont	MS4 Stormwater[1]	4.5	0.004	1.69%
Northern	County of Los Angeles	MS4 Stormwater[1]	27.7	0.025	10.43%
Northern	La Verne[2]	MS4 Stormwater[1]	168	0.152	63.22%
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	24.8	0.022	9.34%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	4.5	0.004	1.69%
Northern	Pomona	MS4 Stormwater[1]	0.5	0.000	0.18%
Northern	San Dimas	MS4 Stormwater[1]	1.6	0.001	0.62%
Northern	Angeles National Forest	Stormwater[1]	1.4	0.001	0.51%
Southern	Caltrans	State Highway Stormwater[1]	1.4	0.001	0.54%
Southern	La Verne	Runoff	1.2	0.001	0.47%
Southern	Pomona	Runoff	1.7	0.002	0.63%
Southern	San Dimas	Runoff	14.8	0.013	5.59%
Southern	County of Los Angeles	Parkland Irrigation	0.0	0.000	0.00%
Total Load from Watershed			**265.5**	**0.24**	**100.00%**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of dieldrin directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load. Direct atmospheric deposition of dieldrin to the lake is accordingly assigned a load allocation of zero.

10.6.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of dieldrin into Puddingstone Reservoir consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted point sources (wasteload allocations) and nonpoint sources (load allocations).

Lake sediments are often the predominant source of dieldrin in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. Dieldrin is strongly sorbed to sediments and has long half-lives in sediment and water. Incoming loads of dieldrin will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data from Puddingstone Reservoir are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. The existing sediment dieldrin concentrations in Puddingstone Reservoir are lower than the consensus-based TEC target, and existing fish tissue concentrations are higher than the fish tissue target. Therefore, a sediment target based on biota-sediment bioaccumulation (a BSAF approach) is calculated from the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to the additional uptake of organochlorine compounds from the sediment by bottom-feeding fish. For dieldrin in Puddingstone Reservoir the ratios of the FCG to the existing fish concentrations (Table 10-38) are:

$$\text{TL4: } 0.46/1.2 = 0.3831$$

$$\text{TL3: } 0.46/2.7 = 0.1692$$

The lower ratio, obtained for the TL3 fish, is applied to the observed in-lake sediment concentration of 1.32 µg/kg dry weight to obtain the site-specific sediment target concentration to achieve fish tissue goals of 0.22 µg/kg dry weight. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations for dieldrin are likely to have declined steadily since the cessation of production and use of the chemical. The resulting fish tissue-based target concentrations of dieldrin in the sediment of Puddingstone Reservoir is shown in Table 10-40.

Table 10-40. Fish Tissue-Based Dieldrin Concentration Targets for Sediment in Puddingstone Reservoir

Total Dieldrin Concentration	Sediment (µg/kg dry weight)
Existing	1.32
BSAF-derived Target	0.22
Required Reduction	83.3%

The BSAF-derived sediment target is less than the consensus-based sediment quality guideline TEC of 1.9 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish.) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target. In addition, the CTR criterion for human health (0.14 ng/L) is the selected numeric target for the water column and protects both aquatic life and human health.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 1,500 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 0.24 g/yr, or 0.02 percent of this amount. Therefore, impairment due to elevated fish tissue concentrations of dieldrin in Puddingstone Reservoir is primarily due to the storage of historic loads of dieldrin in the lake sediment.

10.6.6 TMDL Summary

Because dieldrin impairment in Puddingstone Reservoir is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake. The dieldrin TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 0.22 µg/kg dry weight dieldrin. The wasteload allocations and load allocations are also equal to 0.22 µg/kg dry weight dieldrin in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

10.6.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for dieldrin ("Alternative WLAs if the Fish Tissue Target is Met") described in Section 10.6.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 10.6.6.1.1 if the conditions described in Section 10.6.6.1.2 are met.

10.6.6.1.1 Wasteload Allocations

In the Puddingstone Reservoir watershed, wasteload allocations (WLAs) are required for all permittees in the northern subwatershed and Caltrans areas in the southern subwatershed. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000003
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002

- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

Dieldrin in water flowing into Puddingstone Reservoir is below detection limits, and most dieldrin load is expected to move in association with sediment. Therefore, suspended sediment in water flowing into the lake is assigned waste load allocations. Additionally, the TMDL establishes wasteload allocations for dieldrin in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved dieldrin and dieldrin associated with suspended sediment. The existing concentration on sediment entering the lake is estimated to be 1.0 µg/kg dry weight or less. Therefore, a reduction of up to (1.0 – 0.22)/1.0 = 78 percent is required on the sediment-associated load from the watershed.

The wasteload allocations are shown in Table 10-41 and each wasteload allocation must be met at the point of discharge.

Table 10-41. Wasteload Allocations for Dieldrin in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	0.22	0.14
Northern	Claremont	MS4 Stormwater[1]	0.22	0.14
Northern	County of Los Angeles	MS4 Stormwater[1]	0.22	0.14
Northern	La Verne[2]	MS4 Stormwater[1]	0.22	0.14
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	0.22	0.14
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	0.22	0.14
Northern	Pomona	MS4 Stormwater[1]	0.22	0.14
Northern	San Dimas	MS4 Stormwater[1]	0.22	0.14
Northern	Angeles National Forest	Stormwater[1]	0.22	0.14
Southern	Caltrans	State Highway Stormwater[1]	0.22	0.14

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

10.6.6.1.2 Alternative Wasteload Allocations if Fish Tissue Targets Are Met

The wasteload allocations listed in Table 10-41 will be superseded, and the wasteload allocations in Table 10-42 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,
4. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 10-42, and
2. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 10-42. Alternative Wasteload Allocations for Dieldrin in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Dieldrin Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for Dieldrin in the Water Column[3] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	1.90	0.14
Northern	Claremont	MS4 Stormwater[1]	1.90	0.14
Northern	County of Los Angeles	MS4 Stormwater[1]	1.90	0.14
Northern	La Verne[2]	MS4 Stormwater[1]	1.90	0.14
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	1.90	0.14
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	1.90	0.14
Northern	Pomona	MS4 Stormwater[1]	1.90	0.14
Northern	San Dimas	MS4 Stormwater[1]	1.90	0.14
Northern	Angeles National Forest	Stormwater[1]	1.90	0.14
Southern	Caltrans	State Highway Stormwater[1]	1.90	0.14

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

10.6.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for dieldrin ("Alternative LAs if the Fish Tissue Target is Met") described in Section 10.6.6.2.2. The alternative load allocations will supersede the load allocations in Section 10.6.6.2.1 if the conditions described in Section 10.6.6.2.2 are met.

10.6.6.2.1 Load Allocations

Load allocations (LAs) are assigned to the non-Caltrans permittees in the southern subwatershed and lake bottom sediments. Additionally, the TMDL establishes load allocations for dieldrin in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved dieldrin and dieldrin associated with suspended sediment. No load is allocated to atmospheric deposition of dieldrin. The legacy dieldrin stored in lake sediment is the major cause of use impairment associated with elevated fish tissue concentrations, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (County of Los Angeles) should achieve a dieldrin concentration of 0.22 µg/kg dry weight in lake bottom sediments (Table 10-43). Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-43. Load Allocations for Dieldrin in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	0.22
Southern	Pomona	Runoff[1]	0.22
Southern	San Dimas	Runoff[1]	0.22
Southern	County of Los Angeles	Parkland Irrigation[1]	0.22
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	0.22

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.6.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 10-43 will be superseded, and the load allocations in Table 10-44 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 0.46 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 10-44, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-44. Alternative Load Allocations for Dieldrin in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	1.90
Southern	Pomona	Runoff[1]	1.90
Southern	San Dimas	Runoff[1]	1.90
Southern	County of Los Angeles	Parkland Irrigation[1]	1.90
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	1.90

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.6.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

10.6.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate dieldrin, concentrations in tissues of edible-sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

10.6.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the dieldrin WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99th percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for San Dimas Creek (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99th percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The event mean concentration of sediment in stormwater (45.5 mg/L) was calculated from the estimated existing watershed sediment load of 265.5 tons/yr (Table 10-39) divided by the total annual wet weather flow volume delivered to the lake (4,295 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (38.2 cfs) yields a daily maximum sediment load from stormwater of 4,249 kg/d (4.7 tons/d). Applying the wasteload allocation concentration of 0.22 µg dieldrin per dry kg of sediment yields the stormwater daily maximum allowable load of 0.00086 g/d of dieldrin. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

10.6.6.6 Future Growth

The manufacture and use of dieldrin is currently banned. Therefore, no additional allowance is made for future growth in the dieldrin TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.7 DDT Impairment

Dichlorodiphenyltrichloroethane (DDT) is a synthetic organochlorine insecticide once used throughout the world to control insects. Technically DDT consists of two isomers, 4,4’-DDT and 2,4’-DDT, of which the former is the most toxic. In the environment, DDT breaks down to form two related compounds: DDD (tetrachlorodiphenylethane) and DDE (dichlorodiphenyl-dichloroethylene). DDD and DDE often predominate in the environment and USEPA (2000c) recommends that fish consumption guidelines be based on the sum of DDT, DDD, and DDE – collectively referred to as total DDTs.

The DDT impairment of Puddingstone Reservoir affects beneficial uses related to recreation, municipal water supply, wildlife health, and fish consumption. DDT, like PCBs and chlordane, is an organochlorine compound that is strongly sorbed to sediment and lipids, and is no longer in production. As such, the approach for the DDT impairment is similar to that for PCBs and chlordane.

10.7.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region’s Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Puddingstone Reservoir include REC1, REC2, WARM, WILD, MUN, GWR, COLD, RARE, and AGR. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated levels of DDT are currently impairing the REC1, REC2, WARM, and COLD uses by causing toxicity to aquatic organisms and raising fish tissue concentrations to levels that are unsafe for human consumption (which can result in fish consumption advisories) and impair sport fishing recreational uses. At high enough concentrations WILD, MUN, GWR and RARE uses could become impaired.

10.7.2 Numeric Targets

Targets for DDT are complex because of the many different ways in which the compound is measured. The Basin Plan designates water column concentrations associated with MUN and WARM beneficial uses for several DDTs. There are no numeric criteria specified for sediment or fish tissue concentrations of DDTs listed in the Basin Plan. For the purposes of this TMDL, additional numeric targets for these endpoints are based on the consensus-based sediment quality guidelines defined in MacDonald et al. (2000) and the fish tissue concentration goal, referred to as the fish contaminant goal (FCG), defined by OEHHA (2008) for fish consumption. The numeric targets used for DDTs are listed below. The fish tissue concentration goal was also used to back calculate site-specific targets in sediment, with the most stringent target applying. See Section 2 of this TMDL report for additional details.

The water column criteria for DDT in the Basin Plan are associated with a specific beneficial use. The Basin Plan also contains a narrative criterion that toxic chemicals not be present at levels that are toxic or detrimental to aquatic life (LARWQCB, 1994). Each waterbody addressed in this report is designated WARM, at a minimum, and must meet this requirement. Acute and chronic criteria for 4,4'-DDT in freshwater systems are included in the CTR as 1.1 μg/L and 0.001 μg/L, respectively (USEPA, 2000a). CTR criteria are considered protective of aquatic life. Acute and chronic values for other DDT compounds were not specified. The CTR also includes human health criteria for 4,4'-DDT for the consumption of water and organisms or organisms only as 0.00059 μg/L for both uses (USEPA, 2000a). Because the human health criterion is the most restrictive applicable criterion, a water column target of 0.00059 μg/L (0.59 ng/L) for 4,4'-DDT is the appropriate target. The CTR also specifies a criterion of 0.59 ng/L for 4,4'-DDE (for both consumption of water and organisms or organisms only), while for 4,4'-DDD the criteria are 0.83 ng/L for consumption of water and organisms and 0.84 ng/L for consumption of organisms only. For Puddingstone Reservoir, there is an existing MUN use, so the water and organisms criteria are the appropriate targets. This TMDL the DDT, DDD, and DDE targets in CTR are selected as water column targets.

For sediment, the consensus-based sediment quality guidelines provided in MacDonald et al. (2000) for the threshold effects concentration (TEC) for 4,4'- plus 2,4'-DDT is 4.16 µg/kg dry weight, and the TEC for total DDTs is 5.28 µg/kg dry weight. The consensus-based guidelines have been incorporated into the most recent set of NOAA Screening Quick Reference Tables (SQuiRT) (Buchman, 2008) and are recommended by the State Water Resources Control Board for interpretation of narrative sediment objectives under the 303(d) listing policy. These targets are designed to protect benthic dwelling organisms and explicitly do not consider "the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (i.e., wildlife and humans)." Thus, a separate sediment target calculation based on a biota-sediment accumulation factor (BSAF) is carried out to ensure that fish tissue concentration goals are met.

The fish contaminant goal for total DDTs defined by OEHHA (2008) is 21 ppb wet weight in muscle tissue (filets). Elevated fish tissue concentrations are largely attributable to foodweb bioaccumulation derived from contaminated sediment. A biota-sediment accumulation factor (BSAF) approach is appropriate to correlate sediment and fish tissue targets. For DDTs, the corresponding sediment target concentration determined using the BSAF is 3.94 µg/kg dry weight, as described in further detail in Section 10.7.5. All applicable targets are shown below in Table 10-45. For sediment, the lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

Table 10-45. DDT Targets Applicable to Puddingstone Reservoir

Medium	Source	4,4'-DDT	4,4'-DDT + 2,4'-DDT	DDE[1]	DDD[1]	Total DDTs
Fish (ppb wet weight)	OEHHA FCG					21
Sediment (µg/kg dry weight)	Consensus-based TECs		4.16	3.16[1]	4.88[1]	5.28
Sediment (µg/kg dry weight)	BSAF-derived target					3.94
Water (ng/L)	CTR	0.59		0.59[1]	0.83[1]	

[1] Consensus-based TECs specify sediment targets for total DDE and total DDD. The CTR specifies water column targets specifically for 4,4'-DDE and 4,4'-DDD.

Note: Shaded cells represent the selected targets for this TMDL.

10.7.3 Summary of Monitoring Data

This section summarizes the monitoring data for Puddingstone Reservoir related to the DDT impairment. Additional details regarding monitoring data are discussed in Appendix G (Monitoring Data).

Water column sampling was conducted as part of an organics study performed by UCLA (funded by a grant managed by the Regional Board) in the fall of 2008. These analyses quantified only the 4,4' isomers of DDT, DDD, and DDE. Of four samples (two in Live Oak Wash and two in-lake stations), total DDT was consistently below the detection limits (3.0 ng/L to 3.14 ng/L; the detection limit for total DDT is higher than the water column criterion of 0.59 ng/L).

Water samples from Puddingstone Reservoir were also collected by USEPA and/or the Regional Board on November 18, 2008 at five stations (four in-lake stations and one station in Live Oak Wash) and July 16, 2009 at four stations (Live Oak Wash, storm drain, and two in-lake locations). These analyses included both the 4,4' and 2,4' isomers. Total DDT at all stations was below the detection limit of 1 ng/L, which is above the CTR water column target of 0.59 ng/L. A summary of the water column data is shown in Table 10-46.

Table 10-46. Summary of Water Column Samples for Total DDT in Puddingstone Reservoir

Station	Average Water Concentration (ng/L)	Number of Samples	Number of Samples above Detection Limits[1]
PR-11 (Live Oak Channel)	(1.01)[2]	4	0
PR-14 (Northeast Reservoir Side)	(1.01)	2	0
PR-15 (Western Reservoir Side)	(0.86)	3	0
PR-16 (Southern Reservoir Side)	(0.50)	2	0
PR-17 (Western Reservoir Side near Shoreline)	(0.50)	1	0
PR-SD2 (Storm drain in northeast reservoir area)	(0.50)	1	0
In-Lake Average[3] (PR-14, 15, 16, 17)	(0.72)		
CTR Water Column Target	0.59		

[1] Non-detect samples were included in reported averages at one-half of the sample detection limit.

[2] Numbers in parentheses indicate that the sample is based only on the detection limits of the samples, and that no DDTs were quantified in any of the collected samples.

[3] Overall average is the average of individual station averages.

Pollutant concentrations associated with suspended sediments in the lake were analyzed at two in-lake stations as well as Live Oak Wash during the fall of 2008 by UCLA. Concentrations of total DDT in the suspended sediment samples were less than the detection limits at all three stations (4 µg/kg to 72 µg/kg dry weight). A composite sample during a wet weather event did not detect any DDT (detection limit of 3.14 µg/kg dry weight). A grab sample at the outlet of Live Oak Wash was collected 90 minutes into the wet weather event, which had no detectable results (detection limit of 5.39 µg/kg dry weight). Water column samples were also collected during this event (a time series composite and a single time point sample), but not analyzed. Total DDT concentrations were analyzed in porewater; all samples were less than the detection limit of 30 ng/L. The total suspended sediment associated with the porewater samples also had DDT concentrations less than the detection limits (0.4 µg/kg to 1.06 µg/kg dry weight).

UCLA collected bed sediment samples at two in-lake locations (total of six individual samples) in Puddingstone Reservoir in fall 2008. As with the UCLA water column samples, these included only the 4,4' isomers. Total DDT was consistently below detection limits (0.77 µg/kg to 3.17 µg/kg dry weight). Sediment sampling was also conducted by USEPA and the Regional Board at six stations on July 16, 2009 (Live Oak Wash, two in-lake stations, two storm drain stations, and one natural drainage). Total DDT (including both the 4,4' and 2,4' isomers) was detected at five of the six stations with values ranging from non-detect to 18.6 µg/kg dry weight (four of the six samples had a concentration exceeding the sediment consensus-based TEC of 5.28 µg/kg dry weight). A summary of the sediment data is shown in Table 10-47. The lake-wide average of 7.44 µg/kg is greater than the concentration associated with inputs (5.5 µg/kg), and both are above the consensus-based TEC of 5.28 µg/kg dry weight.

Table 10-47. Summary of Sediment Samples for Total DDT in Puddingstone Reservoir

Station	Average Sediment Concentration (µg/kg dry weight)[1]	Number of Samples	Number of Samples above Detection Limits	Number of Samples between Detection and Reporting Limits
PR-11 (Live Oak Channel)	5.2	1	1	0
PR-14 (Northeast Reservoir Side)	(0.44)	2	0	0
PR-15 (Western Reservoir Side)	10.07	2	1	0
PR-16 (Southern Reservoir Side)	11.8	2	2	0
PR-19 (Natural Drainage on South Side)	7.80	1	1	0
PR-19SD (Storm Drain on South Side)	8.50	1	1	0
PR-SD2 (Storm drain in northeast reservoir area)	(0.50)	1	0	0
In-Lake Average[2] (PR-14, 15, 16)	7.44			
Influent Average	5.50			
Consensus-based TEC[3]	5.28			

[1] Total DDT in a sample represents the sum of all reported measurements for DDT, DDE, and DDD isomers, including results reported below the method reporting limit. If all components were non-detect, the total is represented as one-half the detection limit. Results of any laboratory duplicate analyses of the same sample were averaged. Results for each station represent the average of individual samples. Results in parentheses indicate that the sample average is based only on the detection limits of the samples and that no chlordane quantified in any of the collected samples. Sample averages based only on detected results below the reporting limit plus non-detects are shown in square brackets.

[2] Overall average is the average of individual station averages.

[3] CBSQC TEC is for Total DDTs (DDDs + DDEs + DDTs)

Fish tissue concentrations of total DDT from Puddingstone Reservoir have been analyzed in largemouth bass, common carp, bullhead, and brown bullhead (SWAMP and TSMP). Eight fish samples (composites of filets from five fish) were collected and analyzed for total DDT between 1986 and 1999. In 1986, a largemouth bass and common carp sample reported 16 ppb and 880 ppb wet weight, respectively, while in 1987 another common carp sample had a concentration of 358 ppb wet weight and a bullhead sample reported a concentration of 70 ppb wet weight. In 1988, the reported concentration associated with a brown bullhead sample was 72 ppb wet weight. Three largemouth bass samples had concentrations of 25 ppb, 36 ppb, and 10.7 ppb wet weight in 1991, 1992, and 1999, respectively. The average reported total DDT concentration in all samples from the 1980s and 1990s was 183.5 ppb wet weight. Results from the individual samples are shown in Appendix G (Monitoring Data).

More recently, SWAMP collected samples in September 2004 and June 2007. Considering only data collected in the past 10 years, the average concentration of total DDT in largemouth bass was 24.3 ppb wet weight (average lipid fraction of 0.98 percent), and the average concentration of total DDT in common carp was 39.7 ppb wet weight average lipid fraction of 3.6 percent. The recent fish-tissue data for Puddingstone Reservoir are summarized in Table 10-48.

Table 10-48. Summary of Recent Fish Tissue Samples for Total DDT in Puddingstone Reservoir

Sample Date	Fish Species	Total DDT (ppb wet weight)[1]
9/22/2004	Largemouth Bass	33.7
9/22/2004	Largemouth Bass	15.6
9/22/2004	Largemouth Bass	35.3
9/22/2004	Largemouth Bass	19.4
9/22/2004	Common Carp	2.5
9/22/2004	Common Carp	69.4
9/22/2004	Common Carp	47.7
9/22/2004	Common Carp	39.1
6/6/2007	Largemouth Bass	30.8
6/6/2007	Largemouth Bass	10.8
2004-2007 Average - Largemouth Bass		24.3
2004 Average - Common Carp		39.7
FCG		21

[1] Composite sample of filets from five (largemouth bass) or three individuals (common carp).

In sum, recent fish tissue samples collected from Puddingstone are elevated above OEHHA fish consumption guidelines for total DDT (21 ppb) in three of the six largemouth bass samples and three of the four common carp samples (the average concentrations are also greater than the FCG). Concentrations in sediment are, on average, above the consensus-based TEC, indicating that the lake continues to be impaired by DDT. Water column samples have all been below detection limits; however, all of the detection limits exceeded the CTR criterion.

10.7.4 Source Assessment

Total DDTs present in Puddingstone Reservoir are primarily due to historical loading and storage within the lake sediments, with some ongoing contribution by watershed wet weather loads. Dry weather loading is assumed to be negligible because hydrophobic contaminants primarily move with particulate matter that is mobilized by higher flows. Stormwater loads from the watershed were estimated based on simulated sediment load and observed DDT concentrations on sediment data near inflows to the lake. Watershed loads of DDT may arise from past pesticide applications, improper disposal, and atmospheric deposition. Pesticide applications were most likely associated with agricultural, commercial, and residential areas. Improper disposal could have occurred at various locations, while atmospheric deposition occurs across the entire watershed.

There is no definitive information on specific sources of elevated DDT load within the watershed at this time. Therefore, an average concentration on sediment is applied to all contributing areas, while sources of water that do not contribute sediment load, such as irrigation, are considered to provide no significant DDT loading.

The average concentration of total DDTs on incoming sediment was estimated to be 5.5 µg/kg dry weight (Table 10-47), and the annual sediment load to Puddingstone Reservoir is 265.5 tons/yr (see Appendix D, Wet Weather Loading). The resulting estimated wet-weather load of total DDTs is approximately 1.3 g/yr (Table 10-49).

Table 10-49. Total DDTs Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed (g/yr)

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total DDT Load (g/yr)	Percent of Total Load
Northern	Caltrans	State Highway Stormwater[1]	13.5	0.068	5.10%
Northern	Claremont	MS4 Stormwater[1]	4.5	0.022	1.69%
Northern	County of Los Angeles	MS4 Stormwater[1]	27.7	0.138	10.44%
Northern	La Verne[2]	MS4 Stormwater[1]	168	0.838	63.23%
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	24.8	0.124	9.34%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	4.5	0.022	1.69%
Northern	Pomona	MS4 Stormwater[1]	0.5	0.002	0.18%
Northern	San Dimas	MS4 Stormwater[1]	1.6	0.008	0.62%
Northern	Angeles National Forest	Stormwater[1]	1.4	0.007	0.51%
Southern	Caltrans	State Highway Stormwater[1]	1.4	0.007	0.54%
Southern	La Verne	Runoff	1.2	0.006	0.47%
Southern	Pomona	Runoff	1.7	0.008	0.63%

Subwatershed	Responsible Jurisdiction	Input	Sediment Load (tons/yr)	Total DDT Load (g/yr)	Percent of Total Load
Southern	San Dimas	Runoff	14.8	0.074	5.59%
Southern	County of Los Angeles	Parkland Irrigation	0.0	0.00	0.00%
Total Load from Watershed			**265.5**	**1.32**	**100.00%**

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]The total area for the City of La Verne in the northern subwatershed is 4,079 acres. Discharges governed by the general construction and general industrial stormwater permits are located in the City of La Verne. The disturbed area associated with general construction and general industrial stormwater permittees (233 acres) was subtracted out of the appropriate city area and allocated to these permits.

As described in Appendix E (Atmospheric Deposition), Section E.5, the net atmospheric deposition of total DDT directly to the lake surface is estimated to be close to zero, with deposited loads balanced by volatilization losses. Atmospheric deposition onto the watershed is implicitly included in the estimates of watershed load. Direct atmospheric deposition of total DDT to the lake is accordingly assigned a load allocation of zero.

10.7.5 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity for DDTs in Puddingstone Reservoir consistent with achieving water quality standards. The loading capacity is used to calculate the TMDL and corresponding allocations of that load to permitted, point sources (wasteload allocations) and other nonpoint sources (load allocations). Lake sediments are often the predominant source of DDT in biota. The bottom sediment serves as a sink for organochlorine compounds that can be recycled through the aquatic life cycle. DDT is strongly sorbed to sediment and has a long half-life in sediment and water. Incoming loads of DDT will mainly be adsorbed to particulates from stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

The use of bioaccumulation models and the fish tissue data in Puddingstone Reservoir are discussed in detail in Appendix H (Organochlorine Compounds TMDL Development) and Appendix G (Monitoring Data), respectively. A sediment target to achieve FCGs is calculated based on biota-sediment bioaccumulation (a BSAF approach), using the smaller of the ratio of the FCG to existing fish tissue concentrations obtained from trophic level 4 fish (TL4; e.g., largemouth bass) and bottom-feeding, trophic level 3 fish (TL3; e.g., common carp). In general, the TL3 number is expected to be more restrictive due to the additional uptake of organochlorine compounds from the sediment by bottom-feeding fish. For DDT in Puddingstone Reservoir the ratios of the FCG to the existing fish concentrations (Table 10-48) are:

$$TL4: 21.0/24.3 = 0.8653$$

$$TL3: 21.0/39.7 = 0.5296$$

The smaller ratio, obtained for the TL3 fish, is applied to the estimated lake sediment concentration of 7.44 µg/kg dry weight to obtain the site-specific sediment target concentration to maintain fish tissue goals of 3.94 µg/kg dry weight. The fish tissue-based target concentrations were calculated using only recent data (collected in the past 10 years) because the loads and exposure concentrations of total DDT are likely to have declined steadily since the cessation of production and use of the chemical. The resulting fish tissue-based target concentrations of DDT in sediment of Puddingstone Reservoir is shown in Table 10-50.

Table 10-50. Fish Tissue-Based Total DDTs Concentrations for Sediment in Puddingstone Reservoir

Total DDTs Concentration	Sediment (µg/kg dry weight)
Existing	7.44
BSAF-derived Target	3.94
Required Reduction	47.0%

The BSAF-derived sediment target is less than the consensus-based TEC for total DDTs of 5.28 µg/kg dry weight. (The consensus-based sediment quality guideline is for the protection of benthic organisms, and explicitly does not address bioaccumulation and human-health risks from the consumption of contaminated fish) The lower value of the consensus-based TEC target or the BSAF-derived target is selected as the final sediment target.

The toxicant loading model described in Appendix H (Organochlorine Compounds TMDL Development) can be used to estimate the loading rate that would be required to yield the existing sediment concentration under steady-state conditions. This yields an estimate that a load of 218 g/yr would be required to maintain observed sediment concentrations under steady-state conditions. The estimated current watershed loading rate is 1.32 g/yr, or 0.6 percent of this amount. Thus, concentrations of total DDTs in fish tissue in Puddingstone Reservoir appear to be primarily due to the storage of historic loads of DDT in the lake sediment.

10.7.6 TMDL Summary

Because DDT impairment in Puddingstone Reservoir is predominantly due to historic loads stored in the lake sediment, this impairment is not amenable to a standard, load-based TMDL analysis. Instead, allocations are first assigned on a concentration basis, with the goal of attaining the concentrations identified above for water and sediment, as well as fish tissue. The concentration targets apply to water and sediment entering the lake and within the lake.

The DDT TMDL will be allocated to ensure achievement of the loading capacity. TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation.

$$TMDL = \sum WLA + LA + MOS$$

Note that since this TMDL is being expressed as a concentration in sediment, in this scenario, the loading capacity is equal to 3.94 µg/kg dry weight total DDTs. The wasteload allocations and load allocations are also equal to 3.94 µg/kg dry weight total DDTs in sediment. There is no explicit MOS. Allocations are assigned for this TMDL by requiring equal concentrations of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

10.7.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). This TMDL establishes WLAs at their point of discharge. This TMDL also establishes alternative wasteload allocations for total DDTs (“Alternative WLAs if the Fish Tissue Target is Met”) described in Section 10.7.6.1.2. The alternative wasteload allocations will supersede the wasteload allocations in Section 10.7.6.1.1 if the conditions described in Section 10.7.6.1.2 are met.

10.7.6.1.1 Wasteload Allocations

In the Puddingstone Reservoir watershed, wasteload allocations (WLAs) are required for all permittees in the northern subwatershed and Caltrans areas in the southern subwatershed. Relevant permit numbers are

- County of Los Angeles (including the cities of Claremont, La Verne, Pomona, and San Dimas): Board Order 01-182 (as amended by Order No. R4-2006-0074 and R4-2007-0042), CAS004001
- Caltrans: Order No 99-06-DWQ, CAS000002
- General Construction Stormwater: Order No. 2009-0009-DWQ, CAS000002
- General Industrial Stormwater: Order No. 97-03-DWQ, CAS000001

DDT in water flowing into Puddingstone Reservoir is below detection limits, and most DDT load is expected to move in association with sediment. Therefore, suspended sediment in water flowing into the lake is assigned wasteload allocations. Additionally, the TMDL establishes wasteload allocations for DDT in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved DDT and DDT associated with suspended sediment. The existing concentration of sediment entering the lake is 5.5 µg/kg dry weight. Therefore, a reduction of 28.4 percent [(5.5 – 3.94)/5.5*100] is required on the sediment-associated load from the watershed. The reduction in watershed load is less than the reduction needed for in-lake sediments because the estimated concentration on influent sediment is below the lake-wide average.

The wasteload allocations are shown in Table 10-51 and each wasteload allocation must be met at the point of discharge.

Table 10-51. Wasteload Allocations for DDT in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total DDTs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column[3,4] (ng/L)
Northern	Caltrans	State Highway Stormwater[1]	3.94	0.59[3]
Northern	Claremont	MS4 Stormwater[1]	3.94	0.59
Northern	County of Los Angeles	MS4 Stormwater[1]	3.94	0.59
Northern	La Verne[2]	MS4 Stormwater[1]	3.94	0.59
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	3.94	0.59
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	3.94	0.59
Northern	Pomona	MS4 Stormwater[1]	3.94	0.59
Northern	San Dimas	MS4 Stormwater[1]	3.94	0.59
Northern	Angeles National Forest	Stormwater[1]	3.94	0.59

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total DDTs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column[3,4] (ng/L)
Southern	Caltrans	State Highway Stormwater[1]	3.94	0.59

[1] This input includes effluent from storm drain systems during both wet and dry weather.

[2] Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3] Each wasteload allocation must be met at the point of discharge.

[4] The target water column concentration of 0.59 ng/L specified in the CTR is for 4,4'-DDT. The CTR also specifies targets for DDE and DDD, but does not specify a target for total DDTs. The lowest DDT target is selected for the purposes of representing Total DDTs in this table. If analytical results that resolve individual DDT compounds are available, all of the CTR criteria should be applied individually.

10.7.6.1.2 Alternative Wasteload Allocations if the Fish Tissue Target is Met

The wasteload allocations listed in Table 10-51 will be superseded, and the wasteload allocations in Table 10-52 will apply, if:

1. The responsible jurisdictions submit to USEPA and the Regional Board material describing that the fish tissue target of 21 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative wasteload allocations in Table 10-52, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each wasteload allocation must be met at the point of discharge.

Table 10-52. Alternative Wasteload Allocations for DDT in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total DDTs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column (ng/L)[3,4]
Northern	Caltrans	State Highway Stormwater[1]	5.28	0.59
Northern	Claremont	MS4 Stormwater[1]	5.28	0.59
Northern	County of Los Angeles	MS4 Stormwater[1]	5.28	0.59
Northern	La Verne[2]	MS4 Stormwater[1]	5.28	0.59
Northern	General Industrial Stormwater Permittees (in the city of La Verne)	General Industrial Stormwater[1]	5.28	0.59

Subwatershed	Responsible Jurisdiction	Input	Wasteload Allocation for Total DDTs Associated with Suspended Sediment[3] (µg/kg dry weight)	Wasteload Allocation for 4-4' DDT in the Water Column (ng/L)[3,4]
Northern	General Construction Stormwater Permittees (in the city of La Verne)	General Construction Stormwater[1]	5.28	0.59
Northern	Pomona	MS4 Stormwater[1]	5.28	0.59
Northern	San Dimas	MS4 Stormwater[1]	5.28	0.59
Northern	Angeles National Forest	Stormwater[1]	5.28	0.59
Southern	Caltrans	State Highway Stormwater[1]	5.28	0.59

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]Discharges governed by the general construction and general industrial stormwater permits are currently located in the City of La Verne. Any future discharges governed by the general construction and general industrial stormwater permits will receive the same concentration-based wasteload allocations.

[3]Each wasteload allocation must be met at the point of discharge.

[4]The target water column concentration of 0.59 ng/L specified in the CTR is for 4,4'-DDT. The CTR also specifies targets for DDE and DDD, but does not specify a target for total DDTs. The lowest DDT target is selected for the purposes of representing Total DDTs in this table. If analytical results that resolve individual DDT compounds are available, all of the CTR criteria should be applied individually.

10.7.6.2 Load Allocations

This TMDL establishes load allocations (LAs) at their point of discharge. This TMDL also establishes alternative load allocations for DDTs ("Alternative LAs if the Fish Tissue Target is Met") described in Section 10.7.6.2.2. The alternative load allocations will supersede the load allocations in Section 10.7.6.2.1 if the conditions described in Section 10.7.6.2.2 are met.

10.7.6.2.1 Load Allocations

Load allocations (LAs) are assigned to the non-Caltrans permittees in the southern subwatershed and lake bottom sediments. Additionally, the TMDL establishes load allocations for DDTs in the water column equal to the CTR based water column target. The CTR based water column target includes both dissolved DDTs and total DDTs associated with suspended sediment. No load is allocated to atmospheric deposition of total DDTs. The legacy DDT stored in lake sediment is the major cause of exposure to aquatic organisms and sport fish, and is assigned a load allocation. The in-lake allocation is in concentration terms: specifically, the responsible jurisdiction (County of Los Angeles) should achieve a total DDTs concentration of 3.94 µg/kg dry weight in lake bottom sediments in (Table 10-53). Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-53. Load Allocations for Total DDTs in Puddingstone Reservoir

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	3.94
Southern	Pomona	Runoff[1]	3.94
Southern	San Dimas	Runoff[1]	3.94

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	County of Los Angeles	Parkland Irrigation[1]	3.94
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	3.94

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.7.6.2.2 Alternative Load Allocations if the Fish Tissue Target is Met

The load allocations listed in Table 10-53 will be superseded, and the load allocations in Table 10-54 will apply, if:

1. The responsible jurisdiction submits to USEPA and the Regional Board material describing that the fish tissue target of 21 ppb wet weight has been met for the preceding three or more years. A demonstration that the fish tissue target has been met in any given year must at minimum include a composite sample of skin off fillets from at least five largemouth bass each measuring at least 350mm in length,
2. The Regional Board Executive Officer approves the request and applies the alternative load allocations in Table 10-54, and
3. USEPA does not object to the Regional Board's determination within 60 days of receiving notice of it.

Each load allocation must be met at the point of discharge, except for the lake bottom sediment load allocation which must be met in the lake.

Table 10-54. Alternative Load Allocations for Total DDTs in Puddingstone Reservoir if the Fish Tissue Target is Met

Subwatershed	Responsible Jurisdiction	Input	Load Allocation (µg/kg dry weight)
Southern	La Verne	Runoff[1]	5.28
Southern	Pomona	Runoff[1]	5.28
Southern	San Dimas	Runoff[1]	5.28
Southern	County of Los Angeles	Parkland Irrigation[1]	5.28
Lake Surface	County of Los Angeles	Lake bottom sediments[2]	5.28

[1] Each load allocation must be met at the point of discharge.

[2] The load allocation must be met in the lake.

10.7.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected consensus-based TEC concentration in sediment is considerably lower than the BSAF-derived target.

10.7.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target and protecting benthic biota in sediment. Because fish bioaccumulate DDT, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, overall average loading is more important for the attainment of standards than instantaneous or daily concentrations. WLAs and LAs in this TMDL are assigned as concentrations and protect during all seasons and in both high and low flow conditions. This TMDL therefore protects for critical conditions.

10.7.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. This TMDL includes a maximum daily load estimated according to the guidelines provided by USEPA (2007).

Because the total DDTs WLAs are expressed as concentrations on sediment, the daily maximum allowable load is calculated from the maximum daily sediment load multiplied by the TMDL WLA concentration. The maximum daily sediment load is estimated from the 99th percentile daily flow and the sediment event mean concentration that yields the estimated annual sediment load.

No USGS gage currently exists in the Puddingstone Reservoir watershed. USGS Station 11086400, San Dimas Creek near San Dimas, CA, was selected as a surrogate for flow determination. The 99th percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99th percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99th percentile flow for San Dimas Creek (55 cfs) (Wolock, 2003). To estimate the peak flow to Puddingstone Reservoir, the 99th percentile flow for San Dimas Creek was scaled down by the ratio of drainage areas (8,128 acres/11,712 acres; Puddingstone Reservoir watershed area/San Dimas Creek watershed area at the gage). The resulting peak flow estimate for Puddingstone Reservoir is 38.2 cfs.

The event mean concentration of sediment in stormwater (45.5 mg/L) was calculated from the estimated existing watershed sediment load of 265.5 tons/yr (Table 10-29) divided by the total annual wet weather flow volume delivered to the lake (4,295 ac-ft/yr). Multiplying the sediment event mean concentration by the 99th percentile peak daily flow (38.2 cfs) yields a daily maximum sediment load from stormwater of 4,249 kg/d (4.7 tons/d). Applying the wasteload allocation concentration of 3.94 µg total DDTs per dry kg of sediment yields the stormwater daily maximum allowable load of 0.0167 g/d of total DDTs. This load is associated with the MS4 stormwater permittees. The maximum allowable daily load must be met on all days, and the concentration-based WLAs must be met to ensure compliance with the TMDL.

10.7.6.6 Future Growth

The manufacture and use of DDT is currently banned. Therefore, no additional allowance is made for future growth in the DDT TMDL.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

10.8 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize includes bacteria, nutrients, trash, metals and sediment. Reducing sediment loads would reduce OC pesticides and PCBs as well as mercury delivery to the lake in many instances. More information about this prioritization tool is available at: labmpmethod.org

If necessary, these TMDLs may be revised as the result of new information (See Section 10.9 Monitoring Recommendations). The State Board is in the early stages of developing a Statewide Mercury Policy and Mercury Control Program for Reservoirs. According to CEQA scoping materials, the Policy would define an overall structure for adopting water quality objectives; general implementation requirements; and control plans for mercury impaired water bodies. The final structure of the control program could include a total maximum daily load (TMDL) for mercury in reservoirs along with an implementation plan to achieve the TMDL; or an implementation plan that does not rely on a TMDL. How this upcoming policy and program will affect implementation of this TMDL is unknown at this time.

10.8.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 10-9, Table 10-14, Table 10-23, Table 10-33, and Table 10-53 for nutrients, mercury, PCBs, chlordane, and DDT, respectively.

10.8.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to MS4, Caltrans, and General Industrial and Construction Stormwater permits. Wasteload allocations are expressed in Table 10-7, Table 10-13, Table 10-21, Table 10-31, and Table 10-51 for nutrients, mercury, PCBs, chlordane, and DDT, respectively. The concentration and mass-based wasteload allocations will be incorporated into the Caltrans and Los Angeles County MS4 permits. Concentration-based wasteload allocations will be incorporated into the General Industrial and Construction Stormwater permits.

10.8.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

Puddingstone Reservoir has nutrient-related, mercury, chlordane, DDT, and PCBs impairments. While there are some management strategies that would address all of these impairments (i.e., sediment BMPs placed in upland areas), their differences warrant separate implementation and monitoring discussions.

10.8.3.1 Nutrient-Related Impairments

To address nutrient-related impairments, source reduction and pollutant removal BMPs, designed to reduce sediment loading, could be implemented throughout the watershed as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to Puddingstone Reservoir. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

Education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake or into storm drains that eventually discharge to the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

10.8.3.2 Mercury Impairment

Source reduction and pollutant removal BMPs designed to reduce sediment loading could be implemented throughout the watershed as these management practices will also reduce the mercury loading associated with sediments. However, sedimentation basins or water quality ponds that go anoxic at the sediment-water interface may actually result in increased concentrations of methylmercury. Monitoring of dissolved oxygen levels in these ponds and measurement of total and methylmercury concentrations during warm summer months will assist in the management of these basins to reduce methylmercury loading to Puddingstone Reservoir. Maintaining shallow water levels that do not fluctuate in sedimentation basins will allow penetration of sunlight, which degrades methylmercury, and reduce the wetting and drying conditions that favor methylation.

Dissolved loading associated with storm event runoff also contributes mercury loading to Puddingstone Reservoir. Some of the sediment reduction BMPs may also result in decreased concentrations of mercury in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of mercury from the water column as well as photodegradation. BMPs that provide filtration or infiltration processes may retain dissolved mercury in the upland areas. Additionally, reducing nutrient loading to the lake and improving aeration would likely reduce methylation rates within the lake overall.

Unfortunately, sediment reduction BMPs will not mitigate mercury loading from the largest source in the watershed, atmospheric deposition to the lake surface. Mercury available for deposition in the southwest region typically originates from both local and global sources. In the U.S., mercury emissions from most facilities have been reduced over the past few decades as the best available technology has improved over the years.

To provide reasonable assurances that the assigned allocations will indeed result in compliance with the fish tissue target, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be: 1) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, 2) to document trends over time in mercury loading, and 3) to determine if the load reductions proposed for the TMDL lead to attainment of standards. To assess compliance, it is recommended that a detailed plan be incorporated as part of the implementation plan for this TMDL. This should include annual mercury monitoring of fish tissue as well as quarterly sampling for total and methylmercury in the sediment and water both in-lake and from the tributaries (tributary sampling should also include flow monitoring). It may also be necessary to investigate potential sources of methylmercury loading in the watershed, such as wetlands, sedimentation basins, and areas impacted by forest fires.

In 2008 USEPA modeled mercury air emissions nationally as a tool for tracking airborne mercury to assist in watershed planning. The mercury emission estimates were principally based on 2001 data. The highest modeled impact in California was located in the Long Beach area and the largest single source contributor was the Long Beach South East Resource Recovery facility which combusts municipal waste to produce electricity. Since that time USEPA has promulgated regulations to reduce mercury from solid waste incinerators and the emissions from this facility and another solid waste incinerator in the City of Commerce have been significantly reduced. In addition to these regulations for solid waste combustors, USEPA is in the process of finalizing regulations for Portland Cement plants which also contribute to mercury air loading and deposition in the Los Angeles area.

10.8.3.3 Organochlorine Pesticides and PCBs Impairments

The manufacture and use of chlordane, DDT, dieldrin, and PCBs are currently banned. Therefore, no additional allowances for future growth are needed in the TMDLs. Source control BMPs and pollutant removal are the most suitable courses of action to reduce OC pesticides and PCBs in Puddingstone Reservoir. The TMDL calculations performed for each pollutant (described above in their individual sections) indicated internal lake storage as the greatest contributing source and driving factor affecting fish tissue concentrations. Additionally, the watershed loads for chlordane, dieldrin, and PCBs are less than one percent of the total loading that would be required to maintain the current sediment concentrations in the lake under steady-state conditions. Therefore, the most effective remedial actions and/or implementation efforts will focus on addressing the internal lake storage, such as capping or removal of contaminated lake sediments.

When properly conducted, removal of contaminated lake sediments, or dredging, can be an effective remediation option. The object of sediment dredging is to eliminate the pollutants that have accumulated in sediments at the lake bottom. Dredging is optimal in waterbodies with known spatial distribution of contamination because sediment removal can focus on problem areas. However, no spatial pattern of pollutant contamination was apparent in Puddingstone Reservoir. Removal of the contaminated sediments reduces the pollutants available to the in-lake cycling by discontinuing exposure to benthic organisms, water column loading, and consequential bioaccumulation in higher trophic level fish. Potential negative effects of dredging include increased turbidity and lowered dissolved oxygen concentrations in the short term, and disturbance to the benthic community and reactivation of buried sediment and any associated pollutants.

In some cases, sediment capping may be appropriate to sequester contaminated sediments below an uncontaminated layer of sediment, clay, gravel, or material. Capping is effective in restricting the mobility of OC pesticides and PCBs; however, it is most useful in deep lakes and is likely not a viable solution for some parts of Puddingstone Reservoir. Capping implementation should be restricted to areas with sediments that can support the weight of a capped layer, and to areas where hydrologic conditions of the waterbody will not disturb the cap.

The in-lake options for remediation are costly, but would be the only way to achieve full use support in a short timeframe. It is, however, also true that the OC pesticides and PCBs in question are no longer manufactured and will tend to decline in concentration due to dilution by clean sediment and natural attenuation. Natural attenuation includes the chemical, biological, and physical processes that degrade compounds, or remove them from lake sediments in contact with the food chain, and reduce the concentrations and bioavailability of contaminants. These processes occur naturally within the environment and do not require additional remediation efforts; however, the half-lives of OC pesticides and PCBs in the environment are long, and natural attenuation often requires decades before observing significant improvement.

Loading from the watershed can also be expected to decline over time due to natural attenuation. While reductions are called for in watershed loads, these loads are a small fraction of the historic loads already stored in the lakes. Limited sampling has not identified any hotspots of elevated loading under current conditions. It may, however, be necessary to further investigate potential sources of OC pesticides and PCBs loading in the watershed, such as active and abandoned industrial sites, waste disposal areas, former chemical storage areas, and other potential hotspots.

10.9 Monitoring Recommendations

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations will result in compliance with the chlorophyll *a* and fish tissue targets a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be: 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in algal densities and bloom frequencies and fish tissue mercury and organochlorine compounds concentrations.

10.9.1 Nutrient-Related Impairments

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. Deep lakes, such as Puddingstone Reservoir, must meet the DO and pH targets in the water column from the surface to 0.3 meters above the bottom of the lake when the lake is not stratified. However, when stratification occurs (i.e., a thermocline is present) then the DO and pH targets must be met in the epilimnion, the portion of the water column above the thermocline. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. Wasteload allocations are assigned to stormwater inputs. These sources should be measured near the point where they enter the lakes twice a year for at minimum: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient TMDLs for Puddingstone Reservoir conclude that a 34.1 percent reduction in total phosphorus loading and a 53.6 percent reduction in total nitrogen loading are needed to maintain a summer average chlorophyll *a* concentration of 20 µg/L. As an example of concentrations that responsible jurisdictions may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 10-5), the target concentrations of total phosphorus and total nitrogen may be 0.40 mg-P/L and 1.78 mg-N/L at the outlet of the northern subwatershed. Targeted concentrations for the Caltrans areas in the northern subwatershed may be 0.43 mg-P/L and 1.94 mg-N/L. For the Caltrans areas in the southern subwatershed targeted concentrations may be 0.44 mg-P/L and 2.05 mg-N/L. Similarly, the target concentrations of total phosphorus and total nitrogen may be 0.30 mg-P/L and 1.84 mg-N/L for wet weather runoff from the southern subwatershed; target concentrations in the parkland irrigation return flows to the lake may be 0.50 mg-P/L and 3.59 mg-N/L (10.1 percent of the total irrigation volume is assumed to reach the lake). As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

10.9.2 Mercury Impairment

To assess compliance with the mercury TMDLs, monitoring should include monitoring of largemouth bass (325-375mm in length) fish tissue (skin-off fillets) at least every three years as well as twice yearly sediment and water column sampling. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total mercury, methylmercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids; as well as the following in-lake sediment parameters: total mercury, dissolved methylmercury, total organic carbon, total solids and sulfate. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Additionally, in order to accurately calculate compliance with allocations expressed in yearly loads, monitoring should include flow estimation or monitoring as well as water quality concentration measurements. Wasteload allocations are assigned to stormwater inputs. These sources should be measured near the point where they enter the lakes twice a year for at minimum: total mercury, methyl mercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids.

The mercury TMDL for Puddingstone Reservoir concludes that a reduction in total mercury loading to the lake of 46.6 percent will result in compliance with the fish tissue target of 0.22 ppm. As an example of concentrations that responsible jurisdictions may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 10-5), the target concentration of total mercury may be 3.48 ng/L at the outlet of the northern subwatershed and 3.36 ng/L in runoff from the Caltrans areas in the southern subwatershed. Similarly, the target mercury concentrations may be 2.96 ng/L for wet weather runoff and 12.1 ng/L for parkland irrigation return flows in the southern subwatershed (10.1 percent of the total irrigation volume is assumed to reach the lake). As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved. An in-lake water column dissolved methylmercury target of 0.081 ng/L also applies.

10.9.3 Organochlorine Pesticides and PCBs Impairments

To assess compliance with the organochlorine compounds TMDLs, monitoring should include monitoring of fish tissue at least every three years as well as once yearly sediment and water column sampling. For the OC pesticides and PCBs TMDLs a demonstration that fish tissue targets have been met in any given year must at minimum include a composite sample of skin off fillets from at least five common carp each measuring at least 350mm in length. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total suspended sediments, total PCBs, total chlordane, dieldrin, and total DDTs; as well as the following in-lake sediment parameters: total organic carbon, total PCBs, total chlordane, dieldrin, and total DDTs. Environmentally relevant detection limits should be used (i.e., detection limits lower than applicable target), if available at a commercial laboratory. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Wasteload allocations are assigned to stormwater inputs. These sources should be measured near the point where they enter the lakes once a year during a wet weather event. Sampling should be designed to collect sufficient volumes of suspended solids to allow for the analysis of at minimum: total organic carbon, total suspended solids, total PCBs, total chlordane, dieldrin, and total DDTs. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken.

WLAs and LAs for each pollutant were assigned to the sediment-associated load from the watershed as well as the lake sediments. The concentration-based WLAs and LAs for chlordane, total DDTs, dieldrin, and total PCBs are 0.75 µg/kg dry weight, 3.94 µg/kg dry weight, 0.22 µg/kg dry weight, and 0.59 µg/kg dry weight, respectively. The associated reductions from the watershed load needed to meet the WLAs are 85.3 percent for total chlordane, 28.4 percent for total DDTs, up to 78 percent for dieldrin, and 98.8 percent for total PCBs.

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11 Santa Fe Dam Park Lake TMDLs

Santa Fe Dam Park Lake (#CAL4053100020000303202907) is listed as impaired by copper, lead, and pH SWRCB, 2010). This section of the TMDL report describes the pH impairment and the TMDLs developed to maintain existing water quality (Section 11.2). Nutrient TMDLs are identified here based on existing conditions since nitrogen and phosphorus levels are achieving the chlorophyll *a* target level. Comparison of metals data to their associated hardness-dependent water quality objectives indicates that copper and lead are currently achieving numeric targets at Santa Fe Dam Park Lake; therefore, TMDLs are not included for these pollutants. Analyses are presented below for lead (Section 11.3) and copper (Section 11.4).

11.1 Environmental Setting

Santa Fe Dam Park Lake is located in the San Gabriel River Basin (HUC 18070106) in the Santa Fe Flood Control Basin (Figure 11-1; Figure 11-2). This waterbody was constructed in 1978 by the US Army Corps of Engineers. According to the Los Angeles County Department of Public Works (personal communication Arthur Gotingco, July 13, 2009, County of Los Angeles Dept. of Public Works) and the superintendent of Santa Fe Dam Park Lake, no flood waters are diverted to the park lake (flood water that is diverted from the San Gabriel River enters the spreading grounds and does not reach the lake). In addition, there is no outlet from the lake (personal communication, Chris Graham, County of Los Angeles, March 31, 2010). The park lake is 70.6 ac (surface area based on Southern California Association of Governments [SCAG] 2005 land use) with an average depth of six feet and a total volume of 423.6 ac-ft (volume calculated from depth reported by the lake superintendent and surface area estimated from land use data). Restrooms on the park grounds are connected to the city sewer system. Recreation within Santa Fe Dam Park Lake includes swimming and fishing. The California Fish and Game periodically stock trout. Bird feeding may be another recreational activity at Santa Fe Dam Park Lake; however, it has not been observed during recent fieldwork. There is no known current use of algaecide in this lake. However, lake managers have indicated that the lake has been treated in the past to control nutrients. Additional characteristics of the watershed are summarized below.

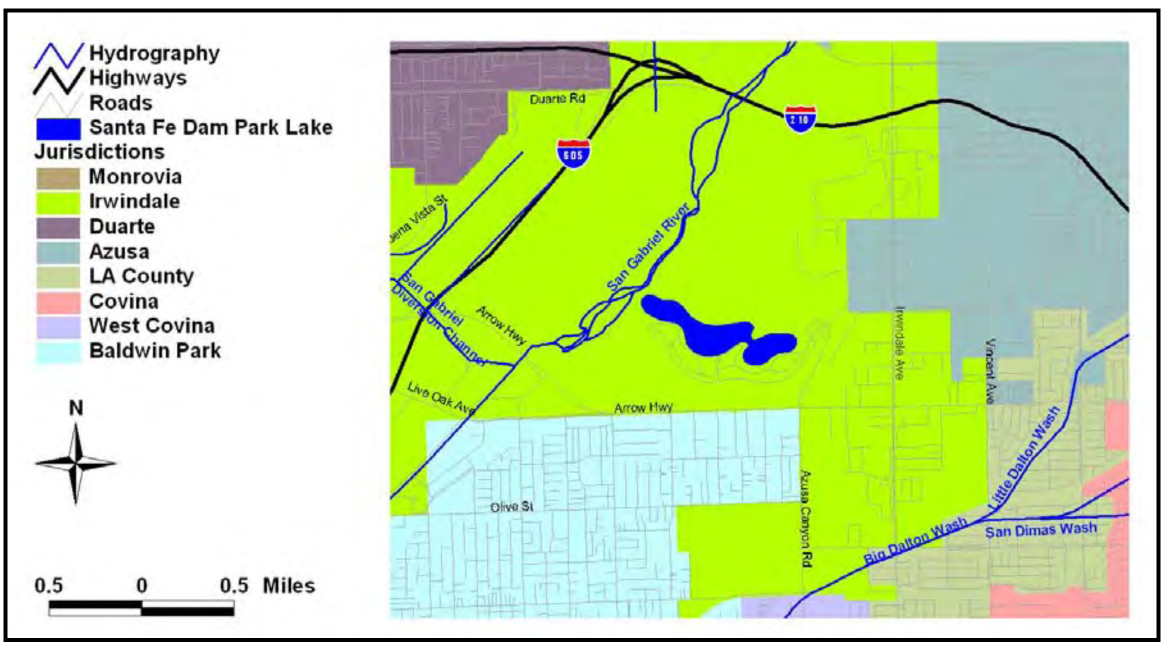

Figure 11-1. **Location of Santa Fe Dam Park Lake**

Figure 11-2. **View of Santa Fe Dam Park Lake**

11.1.1 Elevation, Storm Drain Networks, and Subwatershed Boundaries

The county of Los Angeles subwatershed coverage was sub-delineated based on aerial imagery and a digital elevation model to isolate the drainage area to this lake. Figure 11-3 shows the elevation data for this subwatershed and the resulting 362-acre drainage area (140 meters to 165 meters above sea level). The subwatershed is not drained by a storm drain system, so all loads generated by upland areas will be assigned load allocations except wasteload allocations for the supplemental water additions.

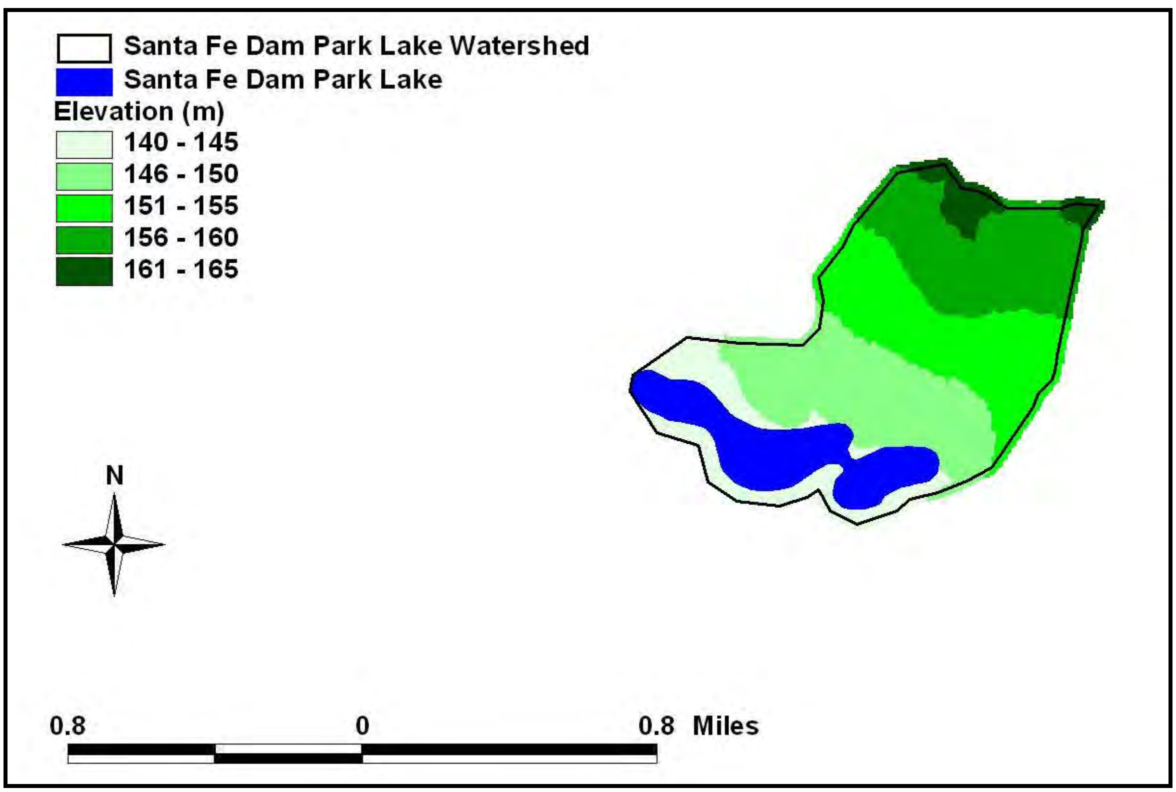

Figure 11-3. **Elevation Data and TMDL Subwatershed Boundaries for Santa Fe Dam Park Lake**

11.1.2 MS4 Permittees

Figure 11-4 shows the MS4 stormwater permittees in the Santa Fe Dam Park Lake subwatershed. The majority of the area is in Irwindale; a small portion is in Azusa. The storm drain coverage used to evaluate whether storm drains are located in the watershed was provided by the county of Los Angeles.

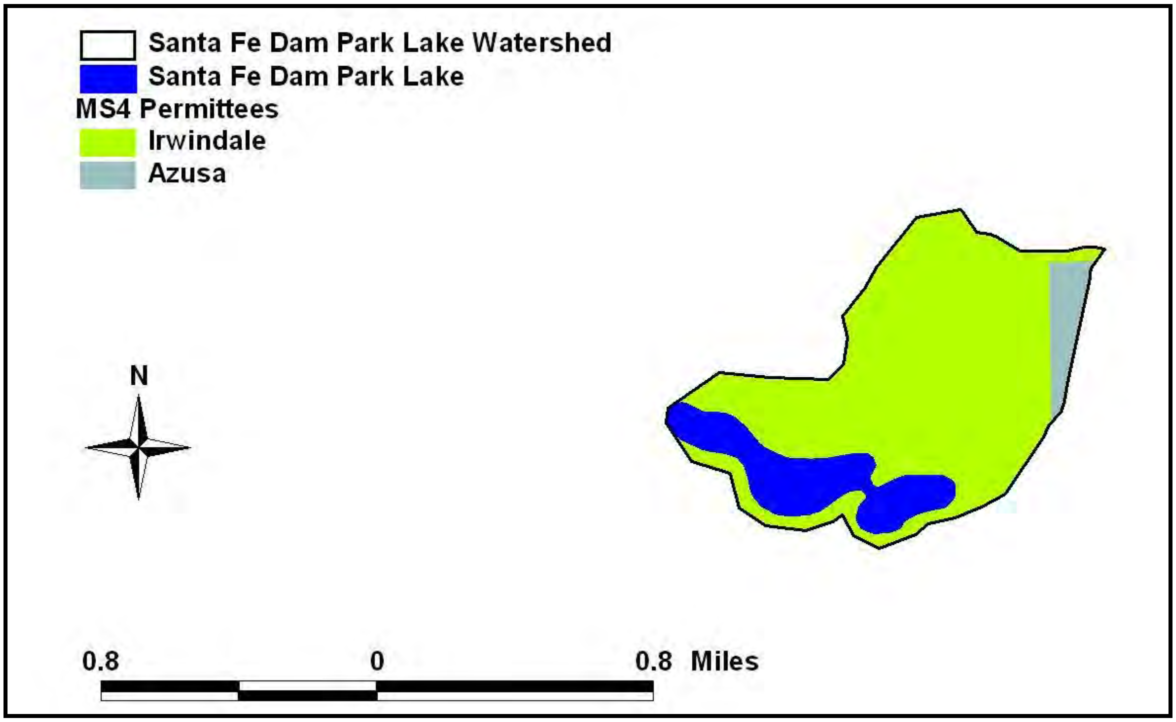

Figure 11-4. **MS4 Permittees in the Santa Fe Dam Park Lake Subwatershed**

11.1.3 Non-MS4 NPDES Dischargers

As of the writing of these TMDLs, there are no non-MS4 NPDES permitted discharges in the Santa Fe Dam Park Lake watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

11.1.4 Land Uses and Soil Types

The analyses for the Santa Fe Dam Park Lake watershed include source loading estimates obtained from the San Gabriel River Basin LSPC Model, discussed in Appendix D (Wet Weather Loading) of this TMDL report. Land uses identified in the San Gabriel River Basin LSPC model for this subwatershed are shown in Figure 11-5.

Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that the portion of area classified by the LSPC model as strip mines had not been mined for some time. The SCAG 2005 database classified this area as vacant; the current satellite imagery shows this area to be re-established shrub/brush rangeland. The 6.25 acres classified by the LSPC model as strip mines were therefore converted to shrub and brush rangeland for this loading analysis. Table 11-1 summarizes the land use areas by jurisdiction.

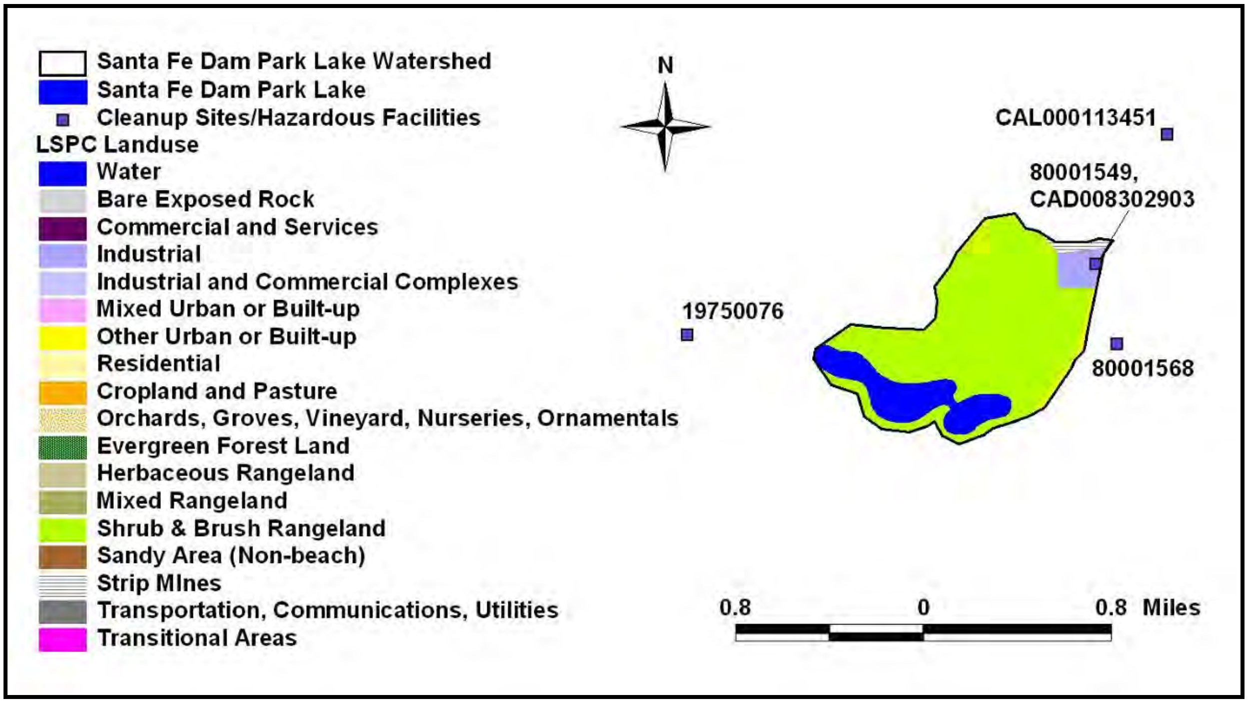

Figure 11-5. **LSPC Land Use Classes for the Santa Fe Dam Park Lake Subwatershed**

Table 11-1. **Land Use Areas (ac) Draining to Santa Fe Dam Park Lake**

Land Use	Azusa	Irwindale	Total
Industrial	11.5	7.16	**18.7**
Other Urban or Built-up	3.94	4.54	**8.48**
Shrub & Brush Rangeland	6.94	328	**335**
Total	**22.4**	**340**	**362**

There is one RCRA cleanup site in the watershed (800001549) and three sites close to the watershed boundary (located within approximately 0.5 miles of the boundary). These are the RCRA cleanup sites in closest proximity to the drainage area and are illustrated in Figure 11-5. Table 11-2 summarizes the information available for these facilities. It is unlikely that these facilities contribute to the pH impairment. Lead is listed as a potential contaminant of concern at three sites below; however, as described below, recent lead samples collected from Santa Fe Dam Park Lake are below the CTR criteria. USEPA recommends the removal of this impairment from this waterbody during the next 303(d) listing cycle.

Table 11-2. **RCRA Cleanup Sites Near the Santa Fe Dam Park Lake Watershed**

Envirostor #	Facility Name	Cleanup Status	Potential Contaminants of Concern
80001549, (CAD008302903)	Veolia ES Technical Solutions	Active, Hazardous Waste Operating Permit	This facility receives a comprehensive suite of hazardous waste including benzenes, dioxins, heavy metals and other toxic organics and inorganics.
80001568	Norac Inc.	Active	Benzoic acid, lead, semi-volatile organics, volatile organics
CAL000113451	Clean Harbors Environmental Services	Evaluation Needed	No data in site summary database for this facility
19750076	Alpha II/Irwindale	No Further Action	Lead, PCBs, Cadmium

The STATSGO soils database shows only one soil type in the watershed, identified as Zamora-Urban land-Ramona (MUKEY 660480). The hydrologic soil group for this soil is not listed in the database.

11.1.5 Additional Inputs

Santa Fe Dam Park Lake receives supplemental flows from groundwater and potable water sources to maintain lake levels. The groundwater is pumped into an artificial stream that then flows into the lake (Figure 11-6). Ten years of monthly usage data were used to estimate the average annual volume pumped from each source. Groundwater and potable water are pumped at average rates of 1,319 ac-ft/yr and 544 ac-ft/yr, respectively. The groundwater source at Santa Fe Dam Park Lake is also used to irrigate 175 acres of parkland and some of this water may reach the lake (9.6 percent of the total irrigation volume is assumed to reach the lake).

In addition to inputs of potable water and groundwater, the swim beach area of Santa Fe Dam Park Lake is disinfected with a 12.5 percent sodium hypochlorite solution (NaOCl) during the summer months. This solution is mixed with lake water in a pump house and then discharged to the lake in the swim area. During the summer, chlorination typically occurs 7 days per week via five pumps. However, due to reduced funding available in 2009, the swim beach was closed Monday through Wednesday and only one chlorine pump was being utilized. The volumes of sodium hypochlorite solution pumped during the summers of 2008 and 2009 were approximately 11,900 gallons each (personal communication, Chris Graham, Los Angeles County Department of Parks and Recreation).

Figure 11-6. **Pumped Groundwater Entering Lake via Artificial Stream**

11.2 PH IMPAIRMENT

Santa Fe Dam Park Lake is listed as impaired by pH. Altered pH chemistry is often a result of elevated nutrient levels that cause excessive growth of algal and plant material. Algal photosynthesis depletes carbon dioxide in the water column, leading to elevated pH during daylight hours, while nighttime respiration releases carbon dioxide, leading to increased concentrations of carbonic acid and depression of pH. However, other circumstances, either natural or anthropogenic, may also lead to impairments in pH. As described below, the pH impairment in Santa Fe Dam Park Lake does not appear to be primarily caused by elevated nutrient levels or chlorination.

11.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. The existing beneficial uses assigned to Santa Fe Dam Park Lake include REC1, REC2, WARM, WILD, WET, and GWR. A potential beneficial use for this lake is MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Elevated pH levels can impair the REC1, REC2, WARM, WILD, and WET uses by causing eye irritation for swimmers and altering the habitat and biota in the lake.

11.2.2 Numeric Targets

The Basin Plan for the Los Angeles Region (LARWQCB, 1994) outlines the numeric targets and narrative criteria that apply to Santa Fe Dam Park Lake. The Basin Plan states that "the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge." The pH target depths for Santa Fe Dam Park Lake are from the surface to 0.3 meters above the lake bottom.

Santa Fe Dam Park Lake is not listed as impaired by algae or eutrophication. However, to determine if elevated nutrient levels, and therefore excessive algal growth, are the cause of the pH impairment in this waterbody, it is useful to compare observed chlorophyll *a* concentrations to target levels. The Regional Board has not adopted numeric targets for chlorophyll *a* in the Santa Fe Dam Park Lake; however, there are applicable narrative criteria that state “Waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growths cause nuisance or adversely affect beneficial uses.” As described in Tetra Tech (2006), summer (May – September) mean and annual mean chlorophyll *a* concentrations of 20 µg/L provide a useful cutoff, representing the maximum allowable level consistent with full support of contact recreational use and are also consistent with supporting warm water aquatic life.

Comparison of dissolved oxygen (DO) and ammonia concentrations to their respective targets is also helpful to determine if algal growth is causing the pH impairment. The instantaneous minimum DO target for Santa Fe Dam Park Lake is 5 mg/L based on the beneficial use designation WARM and must be met from the surface to 0.3 meters above the lake bottom. The Basin Plan expresses ammonia targets as a function of pH and temperature because un-ionized ammonia (NH_3) is toxic to fish and other aquatic life. In order to assess compliance with the standard, the pH, temperature and ammonia must be determined at the same time. For the purposes of setting a target for Santa Fe Dam Park Lake in these TMDLs, a median temperature of 22.3 ºC and a 95th percentile pH of 9.5 were used, as explained in Section 2. The resultant acute (one-hour) ammonia target is 0.70 mg-N/L, the four-day average is 0.41 mg-N/L, and the 30-day average (chronic) target is 0.16 mg-N/L (Note: the median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target varies with the values determined during sample collection.).

Nitrogen and phosphorus target concentrations within the lake are based on existing conditions as explained in Sections 11.2.5 and 11.2.6:

- 0.63 mg-N/L summer season average (May – September) and annual average
- 0.063 mg-P/L summer season average (May – September) and annual average

Table 11-3 presents a summary of the numeric targets for nutrient-related parameters in Santa Fe Dam Park Lake.

Table 11-3. **Nutrient-Related Numeric Targets for Santa Fe Dam Park Lake**

Parameter	Numeric Target	Notes
Ammonia[1]	0.70 mg-N/L acute (one-hour) 0.41 mg-N/L four-day average 0.16 mg-N/L chronic (30-day average)	Based on median temperature and 95th percentile pH
Chlorophyll *a*	20 µg/L summer average (May – September) and annual average	
Dissolved Oxygen	7 mg/L minimum mean annual concentrations and 5 mg/L single sample minimum except when natural conditions cause lesser concentrations	
pH	The pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges. Ambient pH levels shall not be changed more than 0.5 units from natural conditions as a result of waste discharge. (Basin Plan) 6.5 – 9.0 (EPA’s 1986 Recommended Criteria)	The existing water quality criteria for pH is very broad and in cases where waste discharges are not causing the alteration of pH it allows for a wider range of pH than EPA’s recommended criteria. For this reason, EPA’s recommended criteria is included as a secondary target for pH.
Total Nitrogen	0.63 mg-N/L summer average (May – September)	Conservatively based on existing

Parameter	Numeric Target	Notes
	and annual average	conditions, which are maintaining chlorophyll *a* levels below the target of 20 µg/L
Total Phosphorous	0.063 mg-P/L summer average (May – September) and annual average	Based on an in-lake TN to TP ratio of 10, typical of natural systems

[1] The median temperature and 95th percentile pH values were calculated from the observed data and used in the calculation of the acute and chronic targets. These are presented as example calculations since the actual target is the water quality objective which is dependent on pH and temperature. When assessing compliance refer to the water quality objective as expressed in the Basin Plan.

11.2.3 Summary of Monitoring Data

To assess whether or not the pH impairment is due to elevated nutrient levels, analysis of pH, nutrient concentrations, algal densities (chlorophyll *a*), and dissolved oxygen is required. This section briefly summarizes the relevant monitoring data. Appendix G (Monitoring Data) contains more detailed information regarding water quality sampling at Santa Fe Dam Park Lake.

During the summers of 1992 and 1993, the University of Riverside collected water quality data to assess the health of urban lakes in the Los Angeles area. Santa Fe Dam Park Lake was sampled at one station in the southeast section of the lake on 11 days. Total Kjeldahl nitrogen (ammonia plus organic nitrogen; TKN) ranged from 0.3 mg-N/L to 1.1 mg-N/L. Ammonium generally ranged from 0.1 mg-N/L to 0.2 mg-N/L with 21 measurements less than the detection limit (0.01 mg-N/L) and one measurement of 0.4 mg-N/L collected at a depth of 2 meters. All 37 samples of nitrite were less than the detection limit (0.01 mg-N/L), and the majority of nitrate samples (32) were less than the detection limit (0.01 mg-N/L); the maximum observed nitrate concentration was 0.2 mg-N/L. All orthophosphate and total phosphorus concentrations were less than the detection limits (0.01 mg-P/L for both) except one total phosphorus observation which measured 0.1 mg-P/L. pH ranged from 8.0 to 9.6, and TOC ranged from 2.3 mg/L to 3.4 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 1 µg/L to 29 µg/L with an average of 13 µg/L; however, the raw data for chlorophyll *a* were not available.

The 1996 Water Quality Assessment Report (LARWQCB, 1996) states that pH was partially supporting the aquatic life use and not supporting the contact recreation and secondary drinking water uses. Ninety-five measurements of pH were taken, ranging from 7.5 to 9.6 with an average value of 8.7. The associated database did not contain the raw data for these samples.

On March 3rd and August 3rd, 2009, the Regional Board sampled water quality in Santa Fe Dam Park Lake. Overall, both nitrogen and phosphorus levels were very low at the three in-lake stations during both events. TKN ranged from less than the detection limit (0.456 mg-N/L) to 1.1 mg-N/L. Only one sample of ammonia was greater than the detection limit of 0.03 mg-N/L, with a concentration of 0.05 mg-N/L. Nitrite ranged from less than the detection limit (0.01 mg-N/L) to 0.04 mg-N/L; nitrate ranged from less than the detection limit (0.01 mg-N/L) to 0.1 mg-N/L. Phosphate was consistently less than the detection limit of 0.0075 mg-P/L, and total phosphorus ranged from 0.021 mg-P/L to 0.050 mg-P/L. During the winter sampling event, chlorophyll *a* concentrations along the center line of the lake did not exceed 20.5 µg/L, and the average concentration was 17.2 µg/L. In August, chlorophyll *a* was below the detection level of 1 µg/L. pH ranged from 8.6 to 8.8 during both events, with all measurements exceeding the criterion range.

Profile data were collected during the summer monitoring event in the morning and afternoon at two locations in the lake. Over the first 2 meters of depth, DO concentrations were approximately 9.7 mg/L in the morning (~ 9:00 a.m.) and increased to approximately 11.8 mg/L in the afternoon (~4:00 p.m.). pH levels in the morning were approximately 8.5; in the afternoon levels were approximately 8.9. Given that DO concentrations are well above the target of 5 mg/L during the morning hours and pH fluctuations were approximately 0.4 units, algal growth did not appear to be directly causing exceedances of the DO or pH standards during this sampling event. The diurnal change in pH of 0.4 units was less than the allowed change relative to natural conditions of 0.5 units (Section 11.2.2). Additionally, the main discharges to the lake are groundwater and potable water. The pH of the groundwater measured on August 4, 2009 averaged 7.69. Potable water was not measured during a sampling event; however Valley County Water District reported that in 2008 their potable water had an average pH of 7.6 with a range of 7.5-7.7 (Valley County Water District, 2008). The pH standard requires that “the pH of inland surface waters shall not be depressed below 6.5 or raised above 8.5 as a result of waste discharges.” The discharges to the lake do not appear to be causing the elevated pH and therefore the criteria is being met.

On May 4th 2009, Clean Lakes Inc. was contracted by the Los Angeles County Department of Parks and Recreation to conduct baseline water quality monitoring of Santa Fe Dam Park Lake to determine if aquatic weed or algal growth controls were needed. Three locations in the lake were sampled at approximately 10:00 a.m. for water quality at a depth of approximately 1 ft below the water surface. The report stated that the lake did not appear to be impaired with regard to pH, dissolved oxygen, or nutrient levels when compared to the Basin Plan or nutrient TMDLs implemented for other waterbodies in the region. (pH ranged from 7.39 to 7.96 at all locations and depths during this event.) “Light” quantities of pondweed and benthic algae were observed and the lake was noted to have a bluish hue.

On November 17, 2009 the lake was revisited for the collection of metals sampling. pH measurements in the lake ranged from 9.2 to 9.6. pH and DO data were also collected during the December 14, 2009 metals sampling event. At the two midlake stations, pH ranged from 8.63 to 8.90 and DO ranged from 6.1 mg/L to 10.2 mg/L.

On August 12, 2010 the lake was revisited to sample nutrients and metals and to collect diurnal measurements of pH and DO over a 24-hour period. The diurnal sampler placed at SFD-1 measured pH values ranging from 8.75 to 8.97 and DO concentrations ranging from 8.3 mg/L to 9.9 mg/L. At SFD-3, diurnal measurements of pH ranged from 8.82 to 8.97, and DO concentrations ranged from 8.9 mg/L to 11.3 mg/L. TKN ranged from less than the detection limit of 0.5 mg-N/L to 0.594 mg-N/L. Ammonia samples at SFD-1 and SFD-3 were less than the detection limit of 0.03 mg-N/L, and nitrite samples were both detected at 0.035 mg-N/L. Nitrate concentrations were less than the detection limit (0.01 mg-N/L) at SFD-3 and 0.097 mg-N/L at SFD-1. Orthophosphate measurements at both sites were less than the detection limit of 0.0075 mg-P/L; total phosphorus concentrations ranged from 0.023 mg-P/L to 0.129 mg-P/L. Chlorophyll *a* concentrations ranged from 18.4 μg/L to 22.7 μg/L.

In summary, measurements of pH in Santa Fe Dam Park Lake are elevated above the allowable range of 6.5 to 8.5 in both recent and historic datasets. Diurnal sampling shows a variation in pH of less than 0.22 units, indicating that algal and aquatic weed levels are not significantly impacting pH levels. There is no evidence of depressed dissolved oxygen, and limited data on chlorophyll *a* concentrations do not indicate eutrophic conditions (only one measurement exceeded the summer average target of 20 μg/L). There are no dissolved oxygen observations less than the target concentration of 5 mg/L. As explained in greater detail in Section 11.2.5, neither excessive nutrient loading nor chlorination at the lake appear to be causing the elevated pH values that resulted in this listing. Additionally, the main sources of water to these lakes are either groundwater or potable water discharges which account for 97 percent of current flows to the lake. Both of these sources have measured pH values ranging from 7.5 to 7.7 and are not likely to be causing the elevated pH. Based on these multiple lines of evidence, Santa Fe Dam Park Lake is attaining beneficial uses and meets pH water quality standards.

11.2.4 Source Assessment

The elevated pH levels in Santa Fe Dam Park Lake are likely due to natural conditions, as described in Section 11.2.5. Loads of nutrients may also be potentially relevant to the pH impairment, as excess nutrient loads can promote algal growth that depletes CO_2 from the water column and raises pH.

The majority of nutrient loading to Santa Fe Dam Park Lake originates from the groundwater and potable water inputs used to maintain lake levels (Appendix F, Dry Weather Loading). Together these sources account for 82 percent of the total phosphorus load and 95 percent of the total nitrogen load. Other sources include wet weather runoff (Appendix D, Wet Weather Loading), irrigation return flows (9.6 percent of the total irrigation volume is assumed to reach the lake) (Appendix F, Dry Weather Loading) from the surrounding watershed, and atmospheric deposition (Appendix E, Atmospheric Deposition). Table 11-4 summarizes the loadings to the lake.

Table 11-4. **Summary of Average Annual Flows and Nutrient Loading to Santa Fe Dam Park Lake**

Responsible Jurisdiction	Input	Flow (ac-ft)	Total Phosphorus (lb-P/yr) (percent of total load)	Total Nitrogen (lb-N/yr) (percent of total load)
Azusa	Runoff	16.8	27.0 (9.6)	205 (1.6)
Irwindale	Runoff	24.1	23.8 (8.4)	253 (1.9)
County of Los Angeles	Supplemental Water Additions (Groundwater)	1,319	93.3 (33.0)	10,734 (81.5)
County of Los Angeles	Supplemental Water Additions (Potable Water)	544	137 (48.5)	1,789 (13.6)
County of Los Angeles	Parkland Irrigation	16.8	1.2 (0.4)	137 (1.0)
	Atmospheric Deposition (to the lake surface)*	109	NA	51.4 (0.4)
Total		**2,030**	**282**	**13,169**

* Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

Loads of anions and cations are also relevant to the pH balance in the lake, as described in the next section.

11.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions.

In its reaction with chemical compounds in a waterbody, the water molecule (H_2O) dissociates into hydrogen (H^+) and hydroxyl (OH^-) ions. These ions then react with other compounds present in the waterbody. The pH of water is a measure of the proportion of H^+ ions in the water relative to the amount of OH^- ions in the water. Neutral water ($H^+ = OH^-$) has a pH of 7. When more H+ ions are present, the solution is acidic and the pH is less than 7. When more OH^- ions are present, the solution is basic, and the pH is higher than 7.

In natural waterbodies, the presence of carbon dioxide gas (CO_2) in the water is a key driver of pH along with the proportion of cations (positively charged compounds that react with OH^- ions) and anions (negatively charged compounds that react with H^+ ions). Although carbon dioxide makes up a small percentage of the gases in the earth's atmosphere, it is highly soluble in water and is therefore abundant in surface waters. Both groundwater and rainwater have relatively high concentrations of carbon dioxide and may contribute significantly to the carbon stores of a waterbody. Respiration by plants and animals and the decomposition of organic material further increase carbon dioxide levels in the waterbody. When CO_2 is dissolved in water, the pH is typically lowered as CO_2 combines with H_2O to form carbonic acid (H_2CO_3). This acid may dissociate to bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) and hydrogen ions (H^+), and the pH of the water decreases. Photosynthesis reactions consume CO_2 and typically raise the pH of the waterbody.

To determine if the elevated pH measurements in Santa Fe Dam Park Lake are a result of excessive nutrient loading and eutrophication of the waterbody, the California Nutrient Numeric Endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006).

BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. The net sedimentation rates for nitrogen and phosphorus reflect the balance between settling and resuspension of nitrogen and phosphorus within the waterbody. Thus, internal loading is implicitly accounted for in the model. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality.

Target nutrient loads and resulting allocations are determined based on the secondary target – summer mean chlorophyll *a* concentration. The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth. Appendix A (Nutrient TMDL Development) describes additional details on the NNE BATHTUB Tool and its use in determining allowable loads of nitrogen and phosphorus.

In addition to loading rates of nitrogen and phosphorus, the NNE BATHTUB Tool requires basic bathymetry data for the simulation of chlorophyll *a* during the summer. For Santa Fe Dam Park Lake the following inputs apply: surface area of 70.6 acres, average depth of 6 ft, and volume of 423.6 ac-ft. Without adjusting calibration factors in the model, the average annual loads presented in Section 11.2.4 yield total nitrogen, total phosphorus, and chlorophyll *a* concentrations of 1.64 mg-N/L, 0.04 mg-P/L, and 17.8 µg/L, respectively. (Simulated in-lake total nitrogen concentrations are relatively high due to the groundwater input which makes up the majority of the loading and has a concentration of approximately 3 mg-N/L.)

Average conditions for Santa Fe Dam Park Lake with regard to algal stimulation are assessed based on measurements collected between the surface and twice the observed Secchi depth. Average summer observed total nitrogen, total phosphorus, and chlorophyll *a* concentrations over the assessment depth (2 meters) are 0.63 mg-N/L, 0.024 mg-P/L, and 8.5 µg/L, respectively, assuming measurements less than detection are equal to half the detection limit. Even with simulated nitrogen and phosphorus concentrations 1.7 to 2.6 times higher than those observed in the lake (i.e., calibration factors left at 1), simulated chlorophyll *a* (17.8 µg/L) remains below the target concentration of 20 µg/L. Calibrating the NNE BATHTUB Tool would result in lower simulated concentrations of nitrogen, phosphorus, and chlorophyll *a*. Thus, the NNE BATHTUB Tool indicates that Santa Fe Dam Park Lake is not directly impaired by elevated nutrient loads or excessive algal growth. (Since the calibration factor on the net phosphorus sedimentation rate would have been adjusted even lower during calibration, the method described in Appendix A (Nutrient TMDL Development) was used to estimate internal loading. Based on the inflow concentrations, in-lake concentrations, and residence time of this system, the internal loading calculation resulted in a negative number which indicates that settling is more dominant than resuspension, and internal loading of phosphorus is insignificant relative to other sources.) However, the observed algal densities may contribute to depression of CO_2 and weaken the buffering system.

A steady-state, chemical equilibrium model was also set up for Santa Fe Dam Park Lake to determine if elevated pH is due to natural conditions, algal impacts, or the addition of chlorine in the form of sodium hypochlorite, NaOCl, for disinfection of the swim beach area (Section 11.1.5). When NaOCl is added to water, it dissociates into sodium ions (Na^+) and hypochlorite ions (OCl^-). The sodium ion is quickly surrounded by water molecules and is not likely to react further. However, the hypochlorite ion will combine with hydrogen (H^+) that results from the hydrolysis of water ($H_2O = H^+ + OH^-$). The reaction of H^+ with OCl^- results in more hydroxyl ions (OH^-) relative to hydrogen ions (H^+) in the water, and the pH of the solution increases. These acid/base reactions are very rapid and achieve equilibrium within seconds to minutes:

$$NaOCl + H_2O \leftrightarrow HOCl + Na^+ + OH^-$$

The geochemical speciation model, Visual MINTEQ V2.61 (Gustafsson, 2009), was used to investigate the pH conditions in the lake. The model was selected to perform pH simulation based on the available data for Santa Fe Dam Park Lake. The model requires total analytical concentrations and physical inputs to evaluate various geochemical reactions. Visual MINTEQ uses the equilibrium-constant approach to solve chemical equilibrium conditions and uses the same numerical solution method as USEPA's MINTEQA2 (Allison et al., 1991).

The Visual MINTEQ model was used to simulate conditions observed in Santa Fe Dam Park Lake on March 3rd 2009, August 3rd 2009, and August 12th 2010. Average water quality observed on these three days (Table 11-5) was input to the model along with default assumptions regarding the carbon dioxide content of groundwater, atmospheric pressure of carbon dioxide, and concentrations of additional cations and anions based on the available data (calcium, magnesium, and carbonate):

- Because major cation data were not available for the lake, hardness was used to estimate the concentrations of dissolved calcite (Ca) and magnesium (Mg) for each sampling date. The average hardness values ranged from 92.7 mg/L (as $CaCO_3$ mg/L) to 132 mg/L (as $CaCO_3$ mg/L) and were converted to Ca and Mg using the following equation:

  $$\text{Total hardness} = 2.497(Ca^{2+}\ \text{mg/L}) + 4.118(Mg^{2+}\ \text{mg/L})$$

  Assuming the chemical formula of disordered dolomite ($CaMg(CO_3)_2$), the molecular weight ratio of Mg to Ca of 0.6 was applied to magnesium. The calculation resulted in Ca concentrations ranging from 19 mg/L to 27 mg/L and Mg concentrations ranging from 11 mg/L to 16 mg/L. The anion associated with Ca and Mg was assumed to be carbonate according to the dolomite formula. Additional carbonate value was estimated from the observed alkalinity which was assumed to be dominated by carbonate. In addition to carbonate, phosphorus and ammonia could also contribute to alkalinity; however, the relatively low concentrations of these species indicate that these elements are not a major contributor to alkalinity.

- Total organic carbon (TOC) data were used to estimate the possible amount of carbonic acid generated through microbial activities to estimate biodegradable carbon content. Assuming that TOC has the simple sugar glucose structure ($C_6H_{12}O_6$), all TOC values were converted to carbonic acid by microorganism respiration and included in the model.
- Chemical reactions with metal hydroxides associated with clay minerals were not considered in this analysis because the observed TSS concentrations were relatively low.
- Precipitation of saturated calcite solids was not simulated based on the assumption that observed hardness values used to estimate calcium and magnesium concentrations were in the dissolved form and that any potential solid was already precipitated out under the observed water quality conditions.
- Anaerobic decay of organic matter on the lake bottom was not considered in this analysis. However, the lake sediment and adjacent boundary diffusion layer could result in pH changes due to CO_2 generation by microbial activity in conjunction with reduction reactions consuming some of the CO_2 acidity (Morel and Hering, 1993). The depth profile data conducted on August 3rd 2009 show a minimum pH of 7.45 collected at 3 meters from the lake surface. Though the available data do not allow for a quantitative assessment of the influence of microbial activity on pH in the water column, it does indicate that conditions at the sediment-water interface are not acidic and that microbial activity does not appear to have a significant impact on pH throughout the water column.

To test the impacts of algal photosynthesis and respiration on the pH levels, algal densities were assumed at the maximum average levels observed (17 µg/L for the March 3rd 2009 sampling event and 20.05 µg/L for the August 12th 2010 sampling event), and it was also assumed that nitrogen and phosphorus levels were not limiting the rates of photosynthesis or respiration for these organisms.

The average observed pH during the March 3rd event was 8.68. When algal impacts were ignored, the MINTEQ model predicted an average pH of 8.69. When the impacts of algae were simulated, pH was predicted to increase to 8.89 as a result of photosynthesis. During the August 3rd event, chlorophyll *a* levels were less than the detection limit of 1 µg/L. The average observed pH during this event was 8.73 and the average simulated pH was 8.78 (impacts of algae were not included in this scenario since observed algal densities were insignificant). The observed pH during the August 12th 2010 event was 8.8. When algal impacts were ignored, the MINTEQ model predicted an average pH of 8.82. When the impacts of algae were simulated, pH was predicted to increase to 8.94 as a result of photosynthesis.

Table 11-5. **Average Water Quality Conditions in Santa Fe Dam Park Lake on March 3rd and August 3rd 2009 and August 12th 2010**

Date	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chlorophyll *a* (µg/L)	Secchi Depth (m)	Chloride (mg/L)	Temper-ature (°C)	pH	Total Alkalinity (mg/L)	Total Hardness as $CaCO_3$ (mg/L)	TDS (mg/L)	TSS (mg/L)	TOC (mg/L)
3/3/2009	1.02	0.03	0.04	0.08	0.004	0.03	17.17	0.76	27.97	15.50	8.68	115.33	103.72	296.67	7.67	4.67
8/3/2009	0.43	0.02	0.02	0.01	0.004	0.04	0.5	0.54	35.36	27.73	8.73	124.00	131.90	302.67	11.30	3.57
8/12/2010	0.41	0.02	0.04	0.05	0.004	0.08	20.5	0.68	35.7	25.9	8.8	153.00	92.70	241.00	13.90	4.21

Note: Samples reported as less than the detection limit were assumed equal to one-half the detection limit for the purposes of estimating average water quality conditions.

The sensitivity of pH to the addition of NaOCl was tested to examine the impacts of increased concentrations of NaOCl. The original concentration injected into the lake, following mixing of lake water with 12.5 percent NaOCl in the pump house, was assumed 0.05 mg/L based on literature values (Metcalf and Eddy, 2003). During the sensitivity analysis, the concentration was varied from 0 mg/L to 0.50 mg/L. Results of the MINTEQ modeling indicate that the addition of NaOCl has an insignificant impact on pH: the lake volume appears to overwhelm the base effect of hypochlorite ion (OCl^-).

The results of the MINTEQ modeling indicate that the pH of Santa Fe Dam Park Lake is above the target range of 6.5 to 8.5 due to natural conditions including background water quality and gas exchange of carbon dioxide at the air-water interface. While current levels of nutrient loading and algal densities have a minimal impact on pH (less than 0.22 units based on diurnal measurements) and are not responsible for the exceedance of the pH target range, increases in nutrient loading above existing levels could stimulate algal production and result in a problematic increase in pH for this waterbody. Additionally, the main discharges to the lake are groundwater and potable water. The pH of the groundwater at Santa Fe ranged from 7.69 to 7.81 during two sampling events and Valley County Water District reports their potable water had an average pH of 7.6 with a range of 7.5-7.7 (Valley County Water District, 2008). Since the exceedances outside of the target pH range observed in Santa Fe Dam Park Lake are not due to waste discharges or anthropogenic sources, the lake is meeting the pH standard. While carbon dioxide and cations in the water tend to decrease the pH, the presence of anions such as chlorides, silicates, arsenates, and aluminates (Cole, 1994) increase pH. In Santa Fe Dam Park Lake, the elevated pH is likely due to the presence of naturally occurring anions. USEPA concludes that preparing a TMDL for pH is unwarranted at this time and recommends that Santa Fe Dam Park not be identified as impaired by pH in California's next 303(d) list. The nutrient TMDLs will therefore be allocated based on existing conditions as an antidegredation measure to ensure that future loading does not increase algal densities that may further alter the pH of the system.

11.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum nutrient load consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as a summer average. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix A (Nutrient TMDL Development).

Based on simulated and observed levels of chlorophyll *a* in Santa Fe Dam Park Lake, existing levels of nitrogen and phosphorus loading are resulting in attainment of the chlorophyll *a* target concentration and are not significantly impacting the pH. Monitoring data indicate that the average in-lake total nitrogen concentration is 0.63 mg-N/L (Appendix G, Monitoring Data). Because the majority of in-lake phosphorous samples have been less than the detection limits for the analytical laboratory, the phosphorus target concentration is based on an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10. This ratio was selected to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). The corresponding in-lake concentrations of nitrogen and phosphorus are

- 0.63 mg-N/L summer average (May – September) and annual average
- 0.063 mg-P/L summer average (May – September) and annual average

To prevent degradation of this waterbody, nutrient TMDLs are allocated based on existing loading. These TMDLs are broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margins of Safety (MOS) using the general TMDL equation. Note that the MOS is zero as there is no evidence of excess algal growth or significant pH elevation above background conditions.

$$TMDL = \sum WLA + LA + MOS$$

For total nitrogen, the allocatable load is equal to the existing load and is divided among WLAs and LAs. The resulting TMDL equation for total nitrogen is then:

13,169 lb-N/yr = 12,523 lb-N/yr + 646 lb-N/yr + 0 lb-N/yr

For total phosphorus, the allocatable load is equal to the existing load and allocated to WLAs and LAs. The resulting TMDL equation for total phosphorous is then:

282 lb-P/yr = 230 lb-P/yr + 52 lb-P/yr + 0 lb-P/yr

Allocations are assigned for these TMDLs by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on recent and historical monitoring data (see Section 11.2.5). These in-lake concentrations reflect internal cycling processes (see Appendix A, Nutrient TMDL Development) and, therefore, differ from concentrations associated with various inflows. Nutrient concentrations associated with the WLA and LA inputs are described below. These values are provided as examples as they are calculated based on existing flow volumes (they will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

11.2.6.1 Wasteload Allocations

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention and treatment options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. Additionally, persons that apply algaecides as part of an overall lake management strategy must comply with the Aquatic Pesticide General Permit (General Permit Order No. 2004-0009-DWQ, CAG990005).

Local jurisdictions have performed studies on nearby waterbodies that may be considered when evaluating nutrient-reduction strategies for this lake. For example, the City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and construction is currently underway. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). These TMDLs establish WLAs at their point of discharge. There are no MS4 discharges to Santa Fe Dam Park Lake and no other (non-MS4) permitted dischargers in the watershed. The supplemental water addition (groundwater and potable water) inputs are the only sources of nutrient loading to Santa Fe Dam Park Lake that are assigned wasteload allocations (WLAs) (Table 11-6). Note that WLAs are equal to existing loading rates because no reductions in loading are required. These loading values (in pounds per year) represent the TMDLs wasteload allocations (Table 11-6). Each wasteload allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 11-6. **Wasteload Allocations of Phosphorus and Nitrogen Loading to Santa Fe Dam Park Lake**

Responsible Jurisdiction	Input	Total Phosphorus (lb-P/yr)[1]	Total Nitrogen (lb-N/yr)[1]
County of Los Angeles	Supplemental Water Additions (Groundwater)	93.3	10,734
County of Los Angeles	Supplemental Water Additions (Potable Water)	137	1,789
Total		**230**	**12,523**

[1] Each wasteload allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

11.2.6.2 Load Allocations

There are no storm drains that discharge runoff flows into Santa Fe Dam Park Lake. Therefore, all loads associated with the surrounding drainage area are assigned load allocations (LAs) (Table 11-7). Atmospheric deposition is also assigned an LA. These loading values (in pounds per year) represent the TMDLs load allocations (Table 11-7). Each load allocation must be met at the point of discharge. A three-year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

Table 11-7. **Load Allocations of Phosphorus and Nitrogen Loading to Santa Fe Dam Park Lake**

Responsible Jurisdiction	Input	Total Phosphorus (lb-P/yr)[1]	Total Nitrogen (lb-N/yr)[1]
Azusa	Runoff	27.0	205
Irwindale	Runoff	23.8	253
County of Los Angeles	Parkland Irrigation	1.2	137
	Atmospheric Deposition (to the lake surface)[2]	NA	51.4
Total		**52**	**646**

[1] Each load allocation must be met at the point of discharge. A three year average will be used to evaluate compliance. However, if applicable water quality criteria for ammonia, dissolved oxygen and pH, and the chlorophyll *a* target are met in the lake, then the total phosphorous and total nitrogen allocations are considered attained.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

11.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This lake is currently achieving the in-lake chlorophyll a target and TMDLs are being established at the existing loads. This conservative anti-degradation measure is the implicit margin of safety for these TMDLs.

11.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. Critical conditions for nutrient impaired lakes typically occur during the warm summer months when water temperatures are elevated and algal growth rates are high. Elevated temperatures not only reduce the saturation levels of DO, but also increase the toxicity of ammonia and other chemicals in the water column. Excessive rates of algal growth may cause large swings in DO, elevated pH, odor, and aesthetic problems. Loading of nutrients to lakes during winter months are often biologically available to fuel algal growth in summer months. These nutrient TMDLs account for summer season critical conditions by using the NNE Bathtub model to calculate possible annual loading rates consistent with meeting the summer chlorophyll *a* target concentration of 20 µg/L. These TMDLs are based on existing conditions as an anti-degradation measure since nitrogen and phosphorus levels are currently achieving the chlorophyll *a* target level. These TMDLs therefore protect for critical conditions.

11.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. These TMDLs present a maximum daily load according to the guidelines provided by USEPA (2007b). The majority of nutrient loading to Santa Fe Dam Park Lake comes from the supplemental water additions used to maintain lake levels. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDLs must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

The maximum daily loads from the combined supplemental water inputs were calculated by multiplying the highest metered monthly flowrates with the long-term average concentrations consistent with meeting the TMDLs. Allowable concentrations of phosphorus and nitrogen are 0.045 mg-P/L and 2.47 mg-N/L for the combined supplemental water additions (Section 11.2.6.1). The maximum combined metered monthly flow rate is 4,497 ac-ft/mo or 145.1 ac-ft/d (4,497 ac-ft/mo divided by 31 d/mo). Multiplying this maximum daily flowrate by the allowable concentrations yields maximum daily nutrient loads of 17.7 lb-P/d and 975 lb-N/d associated with the combined supplemental water additions. As described above, in order to achieve in-lake nutrient targets as well as annual load-based allocations, the maximum allowable daily loads cannot be discharged to the lake systems every day. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

11.2.6.6 Future Growth

The Santa Fe Dam Park Lake watershed is comprised entirely of parkland/rangeland with a small section of adjacent industrial area. It is not likely that the watershed will be developed and it is expected to remain primarily as open space. No load allocation has been set aside for future growth, and it is unlikely that any dischargers will be permitted in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

11.3 LEAD IMPAIRMENT

Santa Fe Dam Park Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 32 additional samples (12 wet weather) between March 2009 and August 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 32 samples (Appendix G, Monitoring Data). USEPA also collected two sediment samples during the month of August 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Santa Fe Dam Park Lake meets lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Santa Fe Dam Park Lake not be identified as impaired by lead in California's next 303(d) list.

11.4 COPPER IMPAIRMENT

Santa Fe Dam Park Lake was listed as impaired for copper in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (SWRCB, 2005), EPA and local agencies collected 32 additional samples (12 wet weather) between March 2009 and August 2010 to evaluate current water quality conditions. There were zero dissolved copper exceedances in 32 samples (Appendix G, Monitoring Data). USEPA also collected two sediment samples during the month of August 2010 to further evaluate lake conditions. There were zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Santa Fe Dam Park Lake meets copper water quality standards, and USEPA concludes that preparing a TMDL for copper is unwarranted at this time. USEPA recommends that Santa Fe Dam Park Lake not be identified as impaired by copper in California's next 303(d) list.

11.5 IMPLEMENTATION RECOMMENDATIONS

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits, or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that would reduce pollutant loading to this lake include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; increasing flow volume or circulation in the lake; reducing stormwater discharges by improved infiltration; treating stormwater or supplemental water inputs with a wetland system; alum treatment to immobilize nutrients in sediments; dredging in lake sediments; and/or fisheries management actions to reduce nutrient availability from sediments.

If necessary, these TMDLs may be revised as the result of new information (See Section 11.6 Monitoring Recommendations).

11.5.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy. Additionally, South Coast Air

Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 11-7.

11.5.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to supplemental water additions (Table 11-6). These mass-based waste load allocations will be implemented by the Regional Board.

11.5.3 Source Control Alternatives

The nutrient-response analysis for Santa Fe Dam Park Lake indicates that existing levels of nitrogen and phosphorus loading are resulting in attainment of the summer average chlorophyll *a* target concentration of 20 µg/L and are not significantly impacting pH levels in the waterbody. As an antidegradation measure, nitrogen and phosphorus TMDLs are allocated based on existing loading. Future land use changes are not expected in this watershed. In the event that development does occur, source reduction and pollutant removal BMPs, designed to reduce sediment loading, could be implemented as these management practices will also reduce the nutrient loading associated with sediments. Dissolved loading associated with dry and wet weather runoff also contributes nutrient loading to Santa Fe Dam Park Lake. Some of the sediment reduction BMPs may also result in decreased concentrations of nitrogen and phosphorus in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of nutrients from the water column. BMPs that provide filtration, infiltration, and vegetative uptake and removal processes may retain nutrient loads in the upland areas.

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other biofiltration options) to treat the stormwater and supplemental water flows entering the lake, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Implementing these options can reduce the lake's nutrient loads and, in the case of recirculation through constructed wetlands, reduce in-lake nutrient concentrations. The City of Los Angeles has modeled expected nutrient concentration reductions to stormwater flows to Echo Park Lake from constructed wetlands, and plans to finalize the design and begin construction in the near future. Information about this and other City of Los Angeles water quality improvement projects are available on Proposition O website: http://www.lapropo.org/sitefiles/lariver.htm.

Education of park maintenance staff regarding the proper placement, timing, and rates of fertilizer application will also result in reduced nutrient loading to the lake. Park staff should be advised to follow product guidelines regarding fertilizer amounts and to spread fertilizer when the chance of heavy precipitation in the following days is low. Encouraging pet owners who visit the park to properly dispose of pet wastes will also reduce nutrient loading associated with fecal material that may wash directly into the lake. Discouraging feeding of birds at the lake will reduce nutrient loading associated with excessive bird populations.

In order to meet the fine particulate ($PM_{2.5}$) and ozone (O_3) national ambient air quality standards by their respective attainment dates of 2015 and 2024, the South Coast Air Quality Management District and the California Air Resources Board have prepared an air quality management plan that commits to reducing nitrogen oxides (NOx, a precursor to both $PM_{2.5}$ and ozone) by over 85 percent by 2024. These reductions will come largely from the control of mobile sources of air pollution such as trucks, buses, passenger vehicles, construction equipment, locomotives, and marine engines. These reductions in NOx emissions will result in reductions of ambient NOx levels and atmospheric deposition of nitrogen to the lake surface.

11.6 MONITORING RECOMMENDATIONS

Although estimates of the loading capacity and allocations are based on best available data and incorporate a MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur.

To provide reasonable assurances that the assigned allocations result in compliance with the chlorophyll *a* target, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to document trends over time in algal densities and bloom frequencies.

To assess compliance with the nutrient TMDLs, monitoring for nutrients and chlorophyll *a* should occur at least twice during the summer months and once in the winter. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids, total dissolved solids and chlorophyll *a*. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. All parameters must meet target levels at half the Secchi depth. DO and pH must meet target levels from the surface of the water to 0.3 meters above the lake bottom. Additionally, in order to accurately calculate compliance with wasteload allocations to the lake expressed in yearly loads, monitoring should include flow estimation or monitoring as well as the water quality concentration measurements. At Santa Fe Dam Park Lake the only wasteload allocation s are for supplemental water additions. These sources should be monitoring once a year during the summer months (the critical condition) for at minimum; ammonia, TKN or organic nitrogen, nitrate plus nitrite, orthophosphate, total phosphorus, total suspended solids and total dissolved solids.

The nutrient-response analysis for Santa Fe Dam Park Lake indicates that existing levels of nitrogen and phosphorus loading are resulting in attainment of the summer average chlorophyll *a* target concentration of 20 µg/L and are not significantly impacting pH levels in the waterbody. As an antidegradation measure, nitrogen and phosphorus TMDLs are allocated based on existing loading. As an example of concentrations that responsible jurisdiction may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing conditions (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Table 11-4), the target concentrations of phosphorus and nitrogen may be 0.045 mg-P/L and 2.47 mg-N/L for the combined supplemental water additions. The target concentrations of phosphorus and nitrogen in the city of Azusa may be 0.591 mg-P/L and 4.49 mg-N/L. Targeted concentrations may be 0.363 mg-P/L and 3.86 mg-N/L for the city of Irwindale. Targeted concentration in the irrigated parkland return flows to the lake may be 0.026 mg-P/L and 3.0 mg-N/L (9.6 percent of the total irrigation volume is assumed to reach the lake). Assuming an average precipitation depth, the targeted concentration of nitrogen in precipitation may be 0.196 mg-N/L. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved.

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12 Lake Sherwood TMDL

Lake Sherwood (#CAL4042600019990201154540) is listed as impaired by mercury in fish tissue (note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL). Other impairments, for which TMDLs have already been developed, include algae, ammonia, eutrophication, and organic enrichment/low dissolved oxygen (SWRCB, 2010). This section of the TMDL report describes the mercury impairment and the TMDL developed to address it.

12.1 Environmental Setting

Lake Sherwood is located in the Santa Monica Bay Basin (HUC 18070104) between Hidden Valley Wash and Potrero Canyon Creek in Ventura County (Figure 12-1). The lake was created in 1904 from the construction of a dam on the east side of the lake (Figure 12-2). In total, the private lake contains three islands, covers approximately 213 acres and reaches a maximum depth of 30 feet (USEPA, 2003). The lake is primarily fed by watershed runoff but also contains natural springs. Water loss occurs predominantly through evaporation; however, the lake does fill to capacity and discharge to Potrero Canyon Creek during most winters.

The lake was drained for two years during the early 1980s and refilled during 1986 and 1987. The homes surrounding the lake were historically served by individual septic tanks. While the lake was drained they were connected to the Triunfo Sanitation District sewer system. Recreation includes catch and release fishing, boating, and swimming. In addition, a golf course is located on the west end of the lake (Figure 12-3). Bird feeding may be another recreational use at the lake; however, it has not been observed during recent fieldwork. Additional characteristics of the watersheds are summarized below.

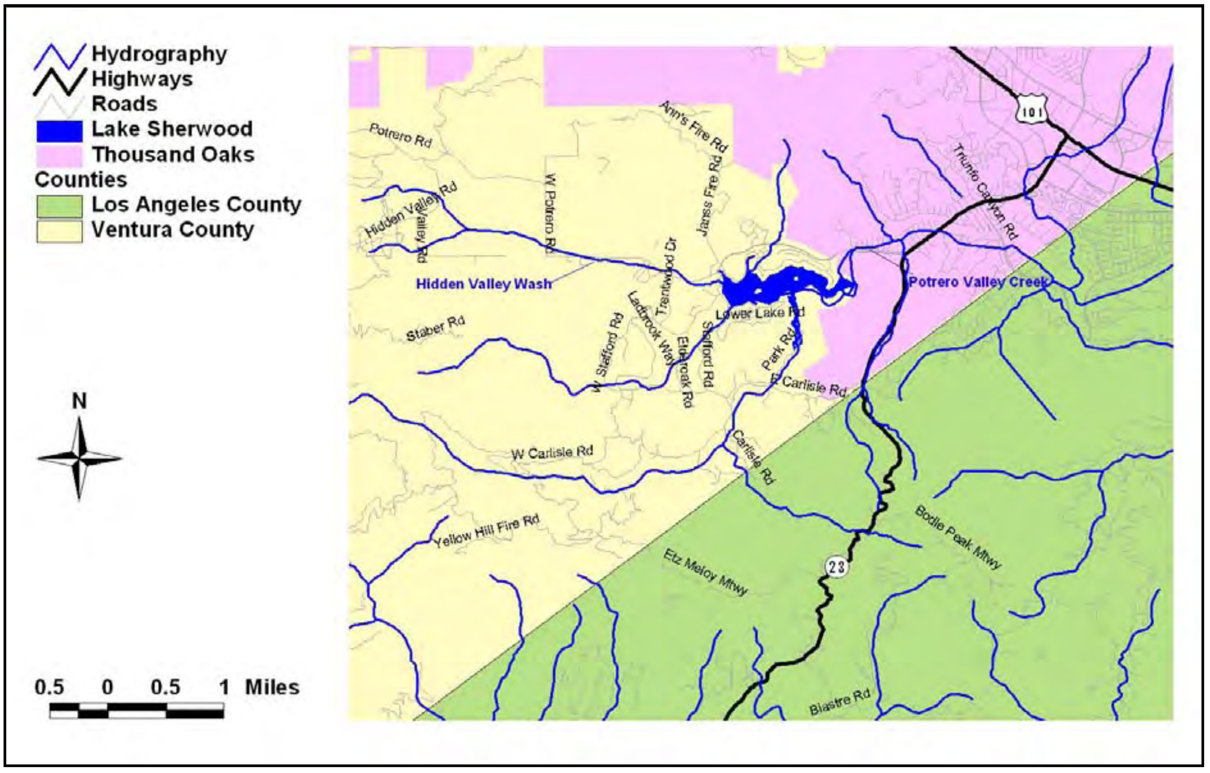

Figure 12-1. **Location of Lake Sherwood**

Figure 12-2. **Lake Sherwood Dam and One of Several Storm Drains**

Figure 12-3. **Creek Winding Through Golf Course and Discharging to Lake through Culverts**

12.1.1 Elevation, Storm Drain Networks, and TMDL Subwatershed Boundaries

Six subwatersheds comprise the drainage area (10,656 acres) to Lake Sherwood, which ranges in elevation from 282 meters to 948 meters (Figure 12-4). TMDL subwatershed boundaries for Lake Sherwood were primarily based on a subwatershed boundary dataset maintained by the county of Los Angeles, which includes the portions of the subwatersheds that intersect with Ventura County. Slight modifications were made to some of the boundaries near the lake, based on aerial photography, to exclude the lake arms from the tributary subwatersheds and to separate the undeveloped and developed areas.

Figure 12-4. **Elevation Data and TMDL Subwatershed Boundaries for Lake Sherwood**

12.1.2 MS4 Permittees

Figure 12-5shows the responsible jurisdictions and entities located in each subwatershed draining to Lake Sherwood. Sherwood Valley Homeowners Association (SVHOA) owns stormdrains in some of the subwatersheds so the area that falls within the Lake Sherwood Overall Plan jurisdiction that is in subwatersheds in which they own stormdrains is also included in Figure 12-5 and further described in Section 12.1.3. The SVHOA is not an MS4 and the stormdrains they own are excluded from the Ventura County MS4 jurisdiction. Ventura County is the only MS4 permittee in the Western subwatershed. The Hidden Valley Wash subwatershed is mostly unincorporated Ventura County with a small portion in Thousand Oaks. The Northern, Near Lake Undeveloped, and Near Lake Developed subwatersheds are comprised of both Ventura County and Thousand Oak areas. The Carlisle Canyon subwatershed contains Ventura and Los Angeles County areas as well as Thousand Oaks, California Department of Transportation (Caltrans), and California State Park areas. Ventura and Los Angeles Counties as well as the City of Thousand Oaks do not maintain a storm drain system in the Lake Sherwood watershed and these areas do not appear to be currently regulated under the existing Ventura County and Los Angeles County MS4 permits. However, there are residential developments in the vicinity of the lake which drain to culverts and storm drains that ultimately discharge to the lake through stormdrains owned by the SVHOA (these areas are further discussed in the next section). Figure 12-6 shows a major storm drain entering Lake Sherwood located at the base of the Northern subwatershed.

All subwatersheds will receive wasteload allocations except for the Carlisle Canyon and Near Lake Undeveloped subwatersheds because these two subwatersheds do not drain to pipes or culverts prior to discharge to the lake. The small Caltrans area in the Carlisle Canyon subwatershed will also receive a wasteload allocation. The new MS4 permit for Ventura County (Order R4 2010-0108, NPDES Permit No. CAS004002, July 8, 2010) unifies MS4 coverage for that county with the Ventura County Watershed Protection District (VCWPD) as Principal Permittee.

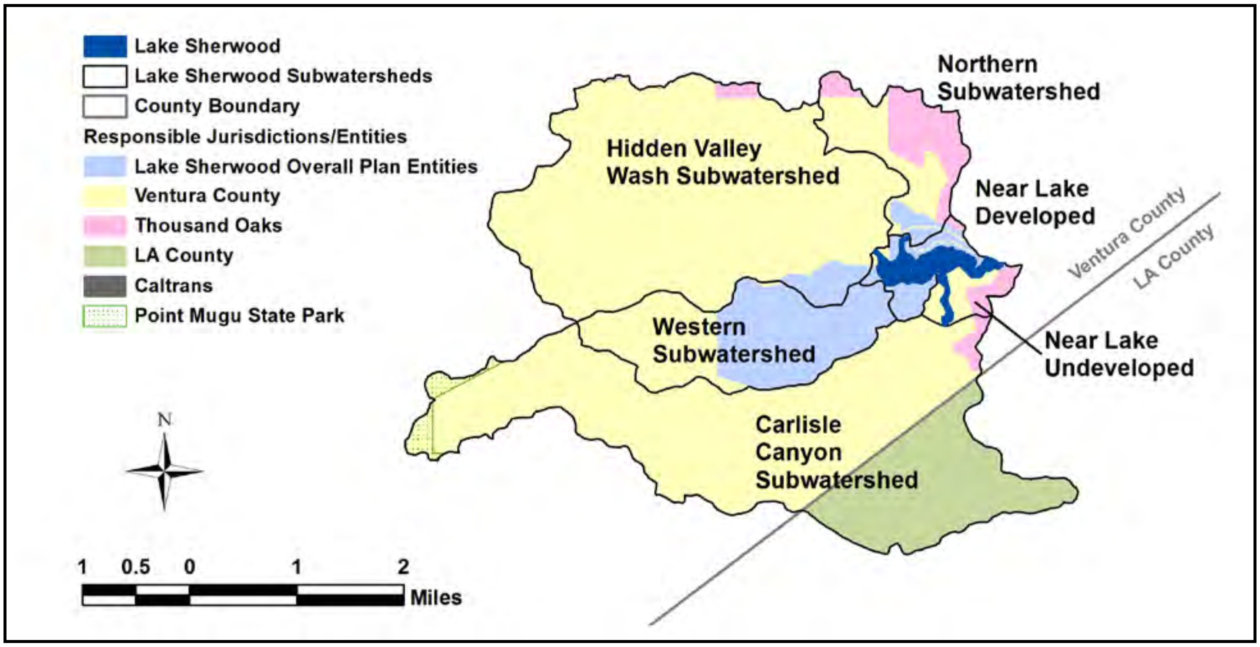

Figure 12-5. **Responsible Jurisdictions / Entities in the Lake Sherwood Subwatersheds**

Figure 12-6. **Major Storm Drain to Lake Sherwood**

12.1.3 Non-MS4 Stormwater Dischargers

Lake Sherwood is in a 1,900 acre planned community that has been primarily developed by Sherwood Development Company but includes homes existing prior to the 1980's when the lake and the undeveloped lands surrounding the lake were purchased by the Sherwood Development Company. Homes built and sold by the Sherwood Development Company fall under the jurisdiction of the Sherwood Valley Homeowners Association (SVHOA), The Glens HOA, Trentwood HOA, Meadows HOA, and/or Northshore HOA. The community has two golf courses, the Sherwood Country Club which is owned by its members and is in the Western subwatershed and one owned by Sherwood Development Company in the Carlisle Canyon subwatershed. Many undeveloped parcels remain and are owned by Sherwood Development Company. Commonly owned parcels as infrastructure are either owned by the SVHOA or will be passed to the SVHOA by the Sherwood Development Company eventually. The multitude of parties within the Lake Sherwood Overall Plan, delineated in Figure 12-7, are given a grouped allocation. The stormdrains that discharge to Lake Sherwood within the residential community surrounding Lake Sherwood were initially thought to belong to Ventura County. However, an October 27, 1998 letter from John C. Crowley, Deputy Director of Public Works, Water Resources and Engineering Department, County of Ventura to Board of Directors, Lake Sherwood Community Services District titled "Transfer of Real Property To Sherwood Valley Homeowner Association, Supervisorial District No. 2" and associated Quitclaim Deeds indicate that the stormdrains in the Lake Sherwood Overall Plan area (see Figure 12-7) are owned by SVHOA. Communication with the County indicates that the only exceptions are any storm drainage infrastructure within the 50 foot wide right of way owned by the County of Ventura along Lake Sherwood Drive. This road is the only road within the 1,900 acre planned residential community owned and maintained by the County. Most, but not all, of the community is gated; gates are located on Stafford Road, Trentwood Drive, Pixton Street, Sandcroft Street, Braxfield Court, Ravesnbury Street and Stonecreek Court. Many roads are owned by the SVHOA or Sherwood Development Company and the following roads are private roads owned by the residents of the surrounding houses: Lower Lake Road, Upper Lake Road, Thorsby, Dirt Road, Hereford, and Giles Road on the south side of the Lake; and David Lane and Trentham on the north side of the Lake.

Lake Sherwood Overall Plan area is shown in Figure 12-7. Additionally, site visits to Lake Sherwood identified many stormwater discharges entering the lake from the surrounding land. The observed stormwater outlets owned by SVHOA discharging to the lake are identified on Figure 12-8. Figure 12-5 illustrates this area in relation to the other stormwater dischargers in the watershed. SVHOA and the Sherwood Development Company will receive joint wasteload allocations.

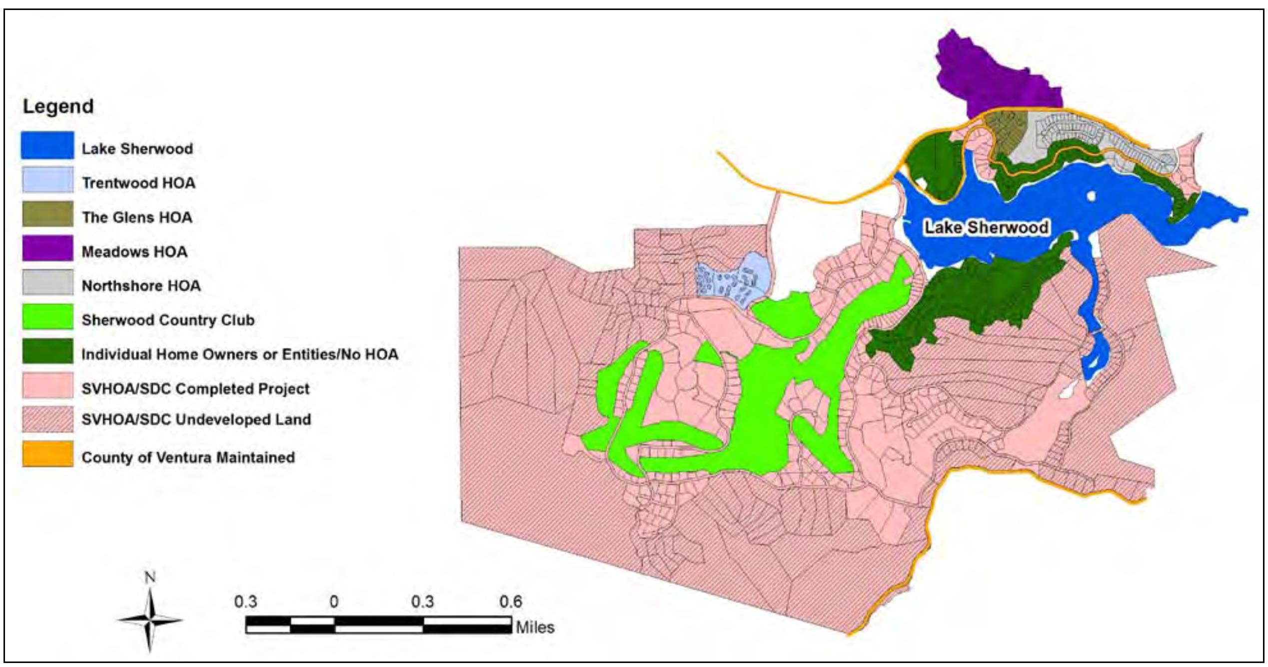

Figure 12-7. **Parcel Map of the Area Included in the Lake Sherwood Overall Plan**

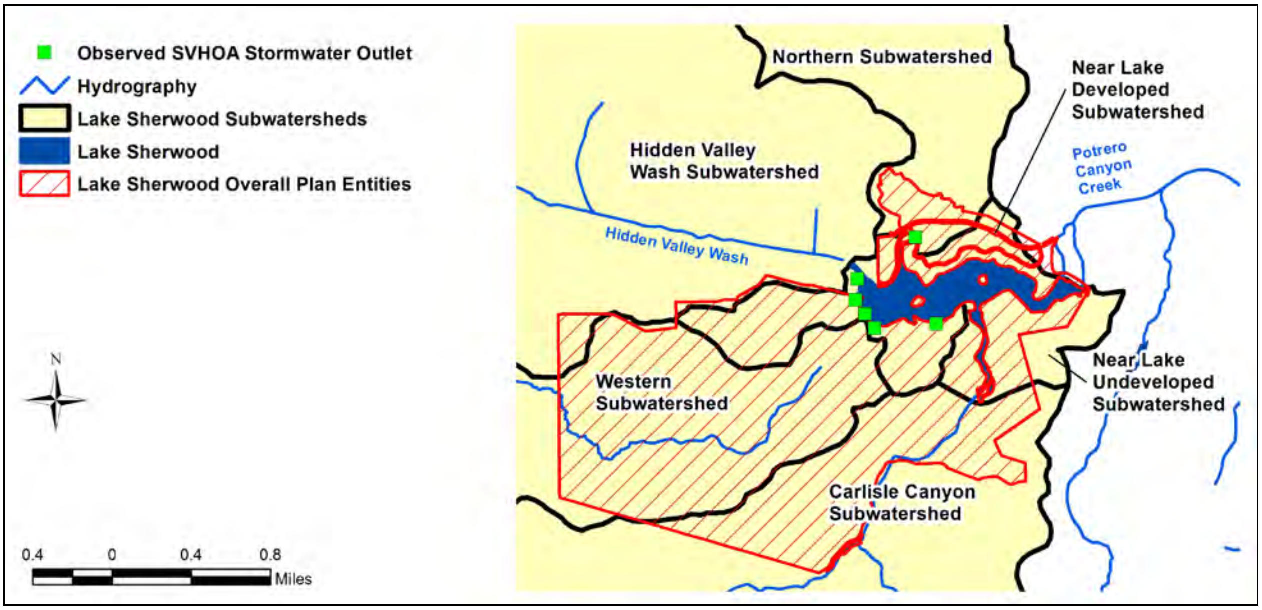

Figure 12-8. **Lake Sherwood Overall Plan and Observed SVHOA Stormwater Outlets**

12.1.4 Non-MS4 NPDES Dischargers

As of the writing of this TMDL, there are no (non-MS4) NPDES permits in the Lake Sherwood watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

12.1.5 Land Uses and Soil Types

Lake Sherwood is located in the Santa Monica Bay Basin and is impaired by mercury. For consistency with the other two mercury impaired lakes addressed by this TMDL (Puddingstone Reservoir and the El Dorado Park lakes), the upland mercury loads will be calculated from tributary monitoring data collected in 2009 and estimates of runoff volumes and sediment loading predicted by an LSPC model (Appendix D, Wet Weather Loading). Though an LSPC model has not been developed for the Santa Monica Bay Basin, the land use coverage for the Los Angeles River Basin LSPC model covers the drainage area to Lake Sherwood and was used to predict runoff volumes and sediment loads by land use to Lake Sherwood.

Land uses identified in the Los Angeles River LSPC model are shown in Figure 12-9. The watershed is comprised of open space, agriculture, residential, and other urban areas. A single parcel of commercial development was identified in the Near Lake Developed subwatershed. Review of SCAG 2005 land use data confirmed that much of the watershed is currently used for agriculture. The area in the Carlisle Canyon subwatershed under the Caltrans jurisdiction was simulated as industrial to estimate sediment loading and runoff volumes from the area associated with this State highway. Table 12-1 through Table 12-6 summarize the land use areas by subwatershed and jurisdiction.

Figure 12-9. **LSPC Land Use Classes for the Lake Sherwood Subwatersheds**

Table 12-1. **Land Use Areas (ac) Draining from the Northern Subwatershed**

Land Use	Ventura County	Thousand Oaks	Lake Sherwood Overall Plan	Total
Agriculture	42	0	0	**42**
Commercial	0	0	0	**0**
Industrial	0	0	0	**0**
Open	301	338	29	**669**
Other Urban	7.2	0	34	**41**
Residential	0.20	0	2	**2**
Total	**351**	**338**	**65**	**754**

Table 12-2. **Land Use Areas (ac) Draining from the Hidden Valley Wash Subwatershed**

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	1,328	0	**1,328**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	2,441	40.4	**2,482**
Other Urban	19.7	0	**20**
Residential	3.97	0	**4**
Total	**3,793**	**40.4**	**3,833**

Table 12-3. **Land Use Areas (ac) Draining from the Western Subwatershed**

Land Use	Ventura County	Lake Sherwood Overall Plan	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	548	587	**1,136**
Other Urban	0	165	**165**
Residential	0	20	**20**
Total	**548**	**772**	**1,321**

Table 12-4. **Land Use Areas (ac) Draining from the Carlisle Canyon Subwatershed**

Land Use	Ventura County	Thousand Oaks	LA County	Caltrans	Point Mugu State Park	Total
Agriculture	5.24	0	0.118	0	0	**5.36**
Commercial	0	0	0	0	0	**0**
Industrial	0	0	0	2.75	0	**2.75**
Open	2,866	50.4	1,149	0	101	**4,166**
Other Urban	34.2	0	0.06	0	0	**34**
Residential	0	0	0	0	0	**0**
Total	**2,905**	**50**	**1,149**	**2.75**	**101**	**4,209**

Table 12-5. **Land Use Areas (ac) Draining from the Near Lake Undeveloped Subwatershed**

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	126	70.9	**197**
Other Urban	0	0	**0**
Residential	0.004	0	**0.004**
Total	**126**	**70.9**	**197**

Table 12-6. **Land Use Areas (ac) Draining from the Near Lake Developed Subwatershed**

Land Use	Ventura County	Thousand Oaks	Lake Sherwood Overall Plan	Total
Agriculture	0	0	0	**0.0**
Commercial	1.13	0	0	**1.1**
Industrial	0	0	0	**0**
Open	15	8.8	143	**167**
Other Urban	3.3	0	110	**113**
Residential	4.4	0	57	**61**
Total	**24**	**8.8**	**310**	**343**

There are no Resource Conservation and Recovery Act (RCRA) contaminated industrial facilities located near the Lake Sherwood watershed. Figure 12-10 shows the predominant soils identified by STATSGO (Appendix D, Wet Weather Loading) in the Lake Sherwood subwatersheds. The most predominant soil type is MUKEY 661018, which is Rock outcrop-Lithic Xerorthents-Hambright Gilroy, a hydrologic group D soil with high runoff potential, very low infiltration rates, and consists chiefly of clay soils. Areas around the lake as well as a large portion of the Hidden Valley Wash subwatershed are comprised of Xerofluvents-Salinas-Pico-Mocho-Metz-Anacapa (soil MUKEY 661012), which is a hydrologic group B soil. These soils have moderate infiltration rates and moderately coarse textures.

Figure 12-10. **STATSGO Soil Types Present in the Lake Sherwood Subwatersheds**

12.1.6 Additional Inputs

Lake Sherwood was included in the 1994 Urban Lakes Study (UC Riverside, 1994). The primary inflows to the lake were identified as several springs and a large tributary on the west end of the lake (Hidden Valley Wash; Figure 12-11 and Figure 12-12) that drains a portion of the Santa Monica Mountains.

Runoff from the surrounding areas also enters the lake. This TMDL accounts for loads delivered from the watershed, but USEPA and the Los Angeles Regional Board were not able to locate or sample the referenced springs during any field reconnaissance or sampling visits. These additional inputs are therefore not considered in the TMDL.

Note: Bridge shown connects to the lake.

Figure 12-11. **Input from Hidden Valley Wash Subwatershed on Northwest Side of Lake Sherwood**

Note: Picture was taken where the wash enters a culvert heading towards the sediment retention basin.

Figure 12-12. **Hidden Valley Wash as it Enters Culvert on Northwest Side of Lake Sherwood**

12.2 Mercury Impairment

The listing information for Lake Sherwood (LARWCB, 1996) indicates that fish tissue data collected by the Toxic Substances Monitoring Program (TSMP) exceeded the fish tissue guideline for mercury and forms the basis for this listing. Continued sampling of largemouth bass from this lake confirms that Lake Sherwood is impaired by mercury.

Lake Sherwood was visited three times in support of the report "Extent of Fishing and Fish Consumption by Fishers in Ventura and Los Angeles County Watersheds" (SCCWRP, 2008). On average, two fishers were observed during each of the three visits. Because Lake Sherwood is a private lake, these fishers were not interviewed, so no direct information is available to determine fishing habits and consumption information for Lake Sherwood. Though no direct information is available regarding consumption of fish caught from Lake Sherwood, a TMDL is required to address the fish tissue impairment because there is the potential to catch and consume fish from this lake.

12.2.1 Beneficial Uses

California state water quality standards consist of the following elements: 1) beneficial uses, 2) narrative and/or numeric water quality objectives, and 3) an antidegradation policy. In California, beneficial uses are defined by the Regional Water Quality Control Boards (Regional Boards) in the Water Quality Control Plans (Basin Plans). Numeric and narrative objectives are specified in each Region's Basin Plan, designed to be protective of the beneficial uses of each waterbody in the region. Applicable water quality criteria are also specified in the California Toxics Rule (USEPA, 2000a). The existing beneficial uses assigned to Lake Sherwood include REC1, REC2, WARM, WILD, WET, GWR, and NAV. A potential beneficial use for Lake Sherwood is MUN. Descriptions of these uses are listed in Section 2 of this TMDL report. Concentrations of mercury measured in fish tissue collected from Lake Sherwood indicate that the REC1, REC2, WARM, and COLD uses are currently impaired. At high enough concentrations WILD, WET, and GWR uses could become impaired.

12.2.2 Numeric Targets

Numeric targets for mercury in Lake Sherwood apply to both the water column and fish tissue. Water column targets are based on beneficial use. For waters designated MUN (existing, potential, or intermittent), the Basin Plan lists a total mercury maximum contaminant level of 0.002 mg/L, or 2 μg/L. The California Toxics Rule includes total mercury human health criteria for the consumption of "water and organisms" or "organisms only" as 0.050 μg/L and 0.051 μg/L, respectively (USEPA, 2000a). California often implements these values on a 30 day average. Because the human health criterion for the consumption of "water and organisms" is the most restrictive criterion, a total mercury water column target of 0.050 μg/L (50 ng/L) is the appropriate target.

In addition, a water column target for dissolved methylmercury of 0.081 ng/L is applicable for Lake Sherwood. This value was calculated by dividing the fish tissue guideline (0.22 ppm) with a national bioaccumulation factor (for dissolved methylmercury) of 2,700,000 applicable for trophic level 4 fish (and multiplying by a factor of 10^6 to convert from milligrams to nanograms).

The fish contaminant goal (FCG) for methylmercury defined by the California Office of Environmental Health Hazard Assessment (OEHHA, 2008) is 220 ppb or 0.22 ppm (wet weight). This concentration is protective of human and wildlife consumers of trophic level four fish. The target length for comparison to this target is 350 mm (13.8 inches) in largemouth bass. Refer to Section 2 of this report for more information regarding these targets.

12.2.3 Summary of Monitoring Data

Total mercury concentrations in the water column of Lake Sherwood have been measured twice to assess compliance with the water quality target. On February 25, 2009, the observed concentration was 3.32 ng/L; on July 15, 2009, the observed concentration was 0.75 ng/L. Both measurements were more than an order of magnitude less than the target (50 ng/L). Total methylmercury concentrations observed during these events were 0.189 ng/L and 0.329 ng/L, and likely exceeded the water column target for dissolved methylmercury (0.081 ng/L). Based on the average observed total methylmercury concentrations, reductions in methylmercury loading of 68.7 percent are needed (Note: the observed data were based on the total fraction, while the water column target is for the dissolved fraction, resulting in more conservative assessments). [Mercury reductions required by the fish tissue data (Section 12.2.5) are higher than 68.7 percent so meeting the reductions for fish tissue should also result in attainment of the water column target for methylmercury.]

The concentrations of mercury observed in largemouth bass have consistently exceeded the fish tissue target since monitoring began in 1991. The TSMP collected three individual specimens in the 1990s with total mercury concentrations ranging from 0.214 ppm to 1.60 ppm. In the summer of 2007, the Surface Water Ambient Monitoring Program (SWAMP) collected 16 individual specimens with total mercury concentrations ranging from 0.219 ppm to 0.802 ppm. The Sherwood Valley HOA sampled five individual fish in 2007; tissue concentrations of total mercury ranged from 0.284 ppm to 0.670 ppm. SWAMP resampled the lake in 2010: five individual samples had concentrations ranging from 0.664 ppm to 1.09 ppm. Figure 12-13 shows the concentration data plotted against fish length; length data for the Sherwood Valley HOA samples were not retained so this data set cannot be plotted. The majority of fish sampled exceed the target concentration of 0.22 ppm. All of the fish tissue data were reported as total mercury concentrations, of which most is expected to be in the methyl form (USEPA, 2001a). These total mercury data were compared to the methylmercury fish tissue guideline, resulting in conservative assessments.

SWAMP also collects total mercury fish tissue concentrations from redear sunfish in Lake Sherwood. These data were not considered in the linkage analysis because redear sunfish are not consumed by humans and, therefore, not relevant for the protection of human health. The composite concentrations (based on five fish per composite) collected on April 19, 2010 range from 0.140 to 0.185 ppm of total mercury.

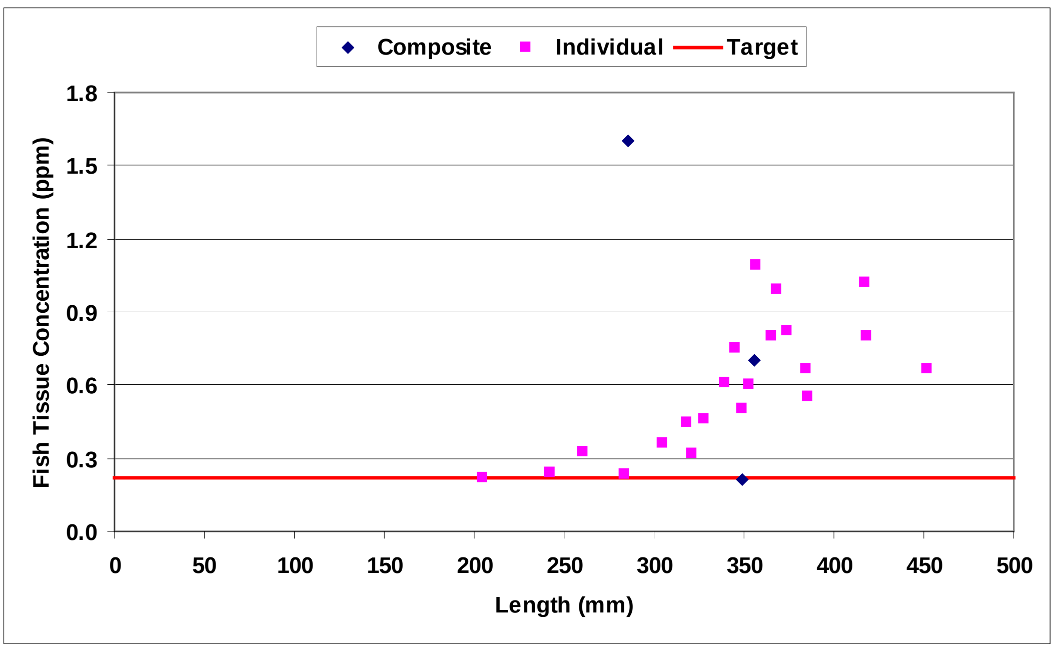

Figure 12-13. **Mercury Concentrations in Largemouth Bass Collected from Lake Sherwood (1991-2010)**

12.2.4 Source Assessment

There are two main components of mercury loading identified in the Lake Sherwood watershed. The majority of loading originates from upland areas and is delivered from tributaries and storm drains in either the water column or sediments. In addition, mercury is deposited from the atmosphere directly to the lake surface.

Watershed loading was determined for six subwatersheds and associated jurisdictions. Estimation of watershed loading during wet weather is discussed in detail in Appendix D (Wet Weather Loading), Section 12; dry weather loading is discussed in Appendix F (Dry Weather Loading), Section 12. Assumptions used to estimate loading from the atmosphere are discussed in Appendix E (Atmospheric Deposition). Table 12-7 summarizes the total mercury loading from each source. On average, 41.7 grams of mercury are delivered to Lake Sherwood each year. The majority of the loading (approximately 45.2 percent) originates from the Hidden Valley Wash subwatershed. Loads from the Near Lake Developed subwatershed are the next largest contributor (17.5 percent).

Table 12-7. **Summary of Existing Total Mercury Loading to Lake Sherwood**

Subwatershed	Responsible Jurisdiction / Entities	Input	Area (ac)	Total Annual Hg Load (g/yr)	Percent of Load
Western	Ventura County	Runoff	549	0.43	1.04
Western	Lake Sherwood Overall Plan Entities*	Runoff	772	2.62	6.29
Hidden Valley Wash	Thousand Oaks	Runoff	40.4	0.031	0.07
Hidden Valley Wash	Ventura County	Runoff	3,793	18.8	45.12
Near Lake Undeveloped	Thousand Oaks	Runoff	70.9	0.043	0.10
Near Lake Undeveloped	Ventura County	Runoff	126	0.077	0.19
Near Lake Developed	Thousand Oaks	Runoff	8.85	0.021	0.05
Near Lake Developed	Ventura County	Runoff	23.8	0.38	0.91
Near Lake Developed	Lake Sherwood Overall Plan Entities*	Runoff	310	6.90	16.55
Northern	Thousand Oaks	Runoff	338	0.786	1.89
Northern	Ventura County	Runoff	351	1.70	4.09
Northern	Lake Sherwood Overall Plan Entities*	Runoff	65.1	1.45	3.47
Carlisle Canyon	Caltrans	State Highway Stormwater[1]	2.75	0.049	0.12
Carlisle Canyon	County of Los Angeles	Runoff	1,149	0.708	1.70
Carlisle Canyon	Thousand Oaks	Runoff	50.4	0.031	0.07
Carlisle Canyon	Ventura County	Runoff	2,905	2.32	5.56
Carlisle Canyon	Point Mugu State Park	Runoff	101	0.06	0.15
Lake Surface		Atmospheric Deposition[2]	137	5.27	12.64
Total				**41.7**	**100**

*Lake Sherwood Overall Plan Entities jointly includes the following: the Glens HOA, Trentwood HOA, Meadows HOA, Northshore HOA, Sherwood Country Club, SVHOA, Sherwood Development Company and individual home owners.

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

12.2.5 Linkage Analysis

The linkage analysis defines the connection between numeric targets and identified pollutant sources and may be described as the cause-and-effect relationship between the selected indicators, the associated numeric targets, and the identified sources. This provides the basis for estimating total assimilative capacity and any needed load reductions. Specifically, models of watershed loading of mercury are combined with an estimated rate of bioaccumulation in the lake. This enables a translation between the numeric target (expressed as a fish tissue concentration of mercury) and mercury loading rates. The loading capacity is then determined via the linkage analysis as the mercury loading rate that is consistent with meeting the target fish tissue concentration.

Neither data nor resources are available to create and calibrate detailed lake response models for mercury cycling in Lake Sherwood. The TMDL target is based on achieving acceptable concentrations in fish. In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of methylmercury in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of methylmercury from lake sediments. In lakes with high sedimentation rates, fish tissue concentrations are therefore likely to respond approximately linearly to reductions in the watershed methylmercury and total mercury load. For Lake Sherwood, reported average annual sedimentation rates measured from 1905 to 1938 ranged from 2.5 to 10 acre-feet per year (0.22 to 0.88 inches per year).

Nationally, authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between methylmercury in water and methylmercury in fish tissue normalized to length. However, this relationship is well-approximated by a linear relationship for the ranges of fish tissue concentration of concern for these impaired lakes. Until such time as a lake response model for mercury is constructed, and sufficient calibration data are collected, an assumption of an approximately linear response of fish tissue concentrations to changes in external loads is sufficient for the development of a TMDL. For a more detailed discussion of the linkage analysis between mercury loading and fish body burden, see Section 3.2.3 of this TMDL report.

12.2.6 TMDL Summary

A waterbody's loading capacity represents the maximum load of a pollutant that can be assimilated without violating water quality standards (40 CFR 130.2(f)). This is the maximum load consistent with meeting the numeric target of 0.22 ppm for mercury in largemouth bass. The methodology for determining the loading capacity is described briefly in this section. For more detail, refer to Appendix C (Mercury TMDL Development).

Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass. To do this, a linear regression analysis was performed on tissue concentrations versus length for Lake Sherwood. The outlier (length = 286 mm, concentration = 1.6 ppm) was removed from the regression to improve fit with the majority of data. The resulting regression equation is

$$Hg(fish) = -0.54236 + 0.003285 \cdot Len, \quad R^2 = 0.490$$

where *Hg(fish)* is the total mercury concentration in largemouth bass (ppm) and *Len* is length in mm. The regression analysis is shown in Figure 12-14, along with the one-sided 95 percent upper confidence limits on mean predictions about the regression line (95 percent UCL) and the 95 percent upper prediction intervals on individual predicted concentrations (95 percent UPI). The UPI gives the confidence limit on the individual predictions for a given length while the UCL gives the confidence limit on the average of the predictions for a given length. This regression has a non-zero intercept and should not be considered valid for lengths less than 200 mm.

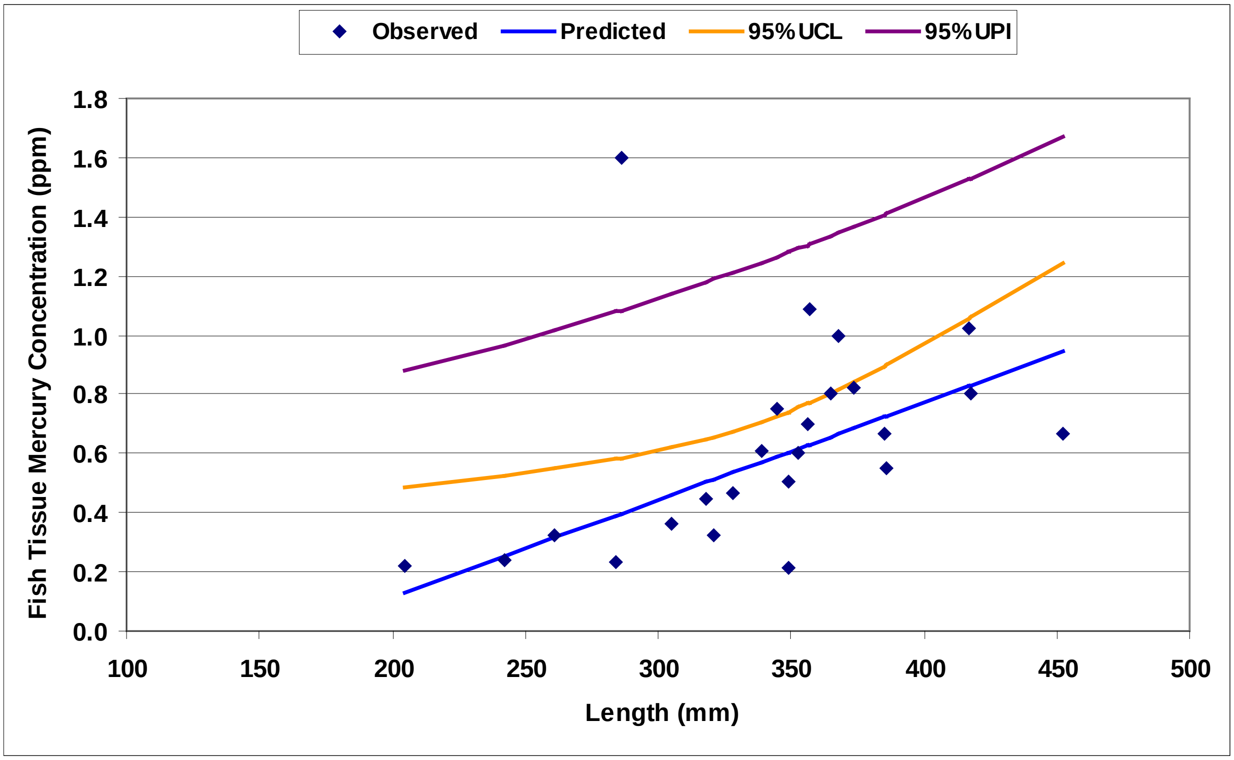

Figure 12-14. **Regression Analysis of Mercury in Lake Sherwood Largemouth Bass (Outlier not included in regression.)**

For mercury, long-term cumulative exposure is the primary concern. Therefore, it is appropriate to use the 95 percent UCL rather than the UPI to provide a Margin of Safety on the appropriate age class. Use of the UCL provides an explicit Margin of Safety because it represents an upper confidence bound on the long-term exposure concentration.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in Lake Sherwood. The TMDL target is established for a 350 mm largemouth bass (see Section 2.2.8). The predicted mercury concentration based on the UCL equation for this length is compared to the target concentration to determine the required reduction in mercury loading, which includes an explicit Margin of Safety as described above.

For Lake Sherwood, the fraction of the existing load consistent with attaining the target (the loading capacity) is the ratio of the target (0.22 ppm) to the best estimate of current average concentrations in the target fish population. The difference between the direct regression estimate and the 95 percent UCL provides the Margin of Safety. Therefore, the allocatable fraction of the existing load (the loading capacity less the Margin of Safety) is the ratio of the target to the 95 percent UCL. The resulting loading capacities and allocatable loads are expressed as fractions of the existing load as summarized in Table 12-8. This analysis indicates that a 70.4 percent reduction in mercury loading will be required to bring fish tissue concentrations in 350 mm largemouth bass (see Section 2.2.8) down to 0.22 ppm.

Table 12-8. **Estimated Total Mercury Loading Capacity and Allocatable Load (as Fractions of the Existing Load)**

Parameter	Value
Target Concentration (ppm)	0.22
Target Length (mm)	350
Predicted Mercury Concentration at Target Length (ppm)	**0.607**
95th Percent UCL (ppm)	**0.744**
Loading Capacity (ratio of target to predicted value)	**0.362**
Allocatable Load (ratio of target to 95th Percent UCL)	**0.296**
Required Reduction in Existing Load (1 minus allocatable fraction)	**0.704**
Margin of Safety Fraction (loading capacity fraction minus allocatable fraction)	**0.067**

The loading capacity can also be expressed as grams per year (g/yr) of total mercury using the existing load presented in Table 12-7 and the calculated fractions of the existing load. Specifically, the loading capacity is 36.2 percent of the existing load of 41.7 g/yr, or 15.1 g/yr. This value can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the equation below.

$$TMDL = \sum WLA + LA + MOS$$

$$15.1\ g/yr = \sum 9.79\ g/yr + 2.51\ g/yr + 2.8\ g/yr$$

The allocatable load (divided among WLAs and LAs) is 29.6 percent of the existing load of 41.7 g/yr, or 12.3 g/yr. This value represents 81.5 percent of the loading capacity, while the MOS is 18.5 percent of the loading capacity. Allocations are assigned for this TMDL by requiring equal percentage reductions of all sources. Details associated with the WLAs, LAs, and MOS are presented in the following three sections.

12.2.6.1 Wasteload Allocations

Federal regulations require that NPDES permits incorporate water quality based effluent limitations (WQBELs) consistent with the requirements and assumptions of any available wasteload allocations (WLAs). Wasteload allocations are required for all waters that discharge to the lake through stormdrains or culverts. Responsible entities located in the Northern, Western, Hidden Valley Wash, and Near Lake Developed subwatersheds discharge to the lake through stormdrains or culverts that are not currently part of the Ventura or Los Angeles County MS4s. The stormdrains in the Northern, Western and Near Lake Developed subwatersheds are owned and operated by the SVHOA. In those subwatersheds any land in the Sherwood Lake Overall Plan area is allocated jointly to the following entities: the Glens HOA, Trentwood HOA, Meadows HOA, Northshore HOA, Sherwood Country Club, SVHOA, Sherwood Development Company and individual home owners. This TMDL establishes WLAs at their point of discharge. A WLA is also required for the Caltrans area in the Carlisle Canyon subwatershed. The relevant permit number is

- Caltrans: Order No 99-06-DWQ, CAS000003.

Table 12-9 summarizes the existing loads and WLAs of total mercury for these sources. The WLAs are a 70.4 percent reduction from the existing loads. These loading values (in grams per year) represent the TMDL wasteload allocations (Table 12-9). In addition to the WLAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L also applies.

Table 12-9. **Wasteload Allocations of Mercury in the Lake Sherwood Watershed**

Subwatershed	Responsible Jurisdiction / Entities	Input	Area (ac)	Existing Annual Hg Load (g/yr)	Wasteload Allocation[2] (g/yr)
Western	Ventura County	Runoff[3]	548	0.43	0.128
Western	Lake Sherwood Overall Plan Entities*	Runoff[3]	772	2.62	0.774
Hidden Valley Wash	Thousand Oaks	Runoff[3]	40	0.03	0.009
Hidden Valley Wash	Ventura County	Runoff[3]	3,793	18.8	5.559
Near Lake Developed	Thousand Oaks	Runoff[3]	9	0.021	0.006
Near Lake Developed	Ventura County	Runoff[3]	24	0.38	0.112
Near Lake Developed	Lake Sherwood Overall Plan Entities*	Runoff[3]	310	6.90	2.039
Northern	Thousand Oaks	Runoff[3]	338	0.786	0.232
Northern	Ventura County	Runoff[3]	351	1.70	0.504
Northern	Lake Sherwood Overall Plan Entities*	Runoff[3]	65	1.45	0.427
Carlisle Canyon	Caltrans	State Highway Stormwater[1]	2.75	0.049	0.014
Total				**33.1**	**9.79**

*Lake Sherwood Overall Plan Entities jointly includes the following: the Glens HOA, Trentwood HOA, Meadows HOA, Northshore HOA, Sherwood Country Club, SVHOA, Sherwood Development Company and individual home owners.

[1]This input includes effluent from storm drain systems during both wet and dry weather.

[2]Each mass-based wasteload allocations must be met at the point of discharge.

[3]This input includes effluent carried through the stormdrain system owned by SVHOA that is not currently included in either the Los Angeles or Ventura County MS4 permits.

12.2.6.2 Load Allocations

Load allocations (LAs) of total mercury are assigned to all sources not subject to permits in the Near Lake Undeveloped subwatershed, non-Caltrans sources in the Carlisle Canyon subwatershed, and atmospheric deposition. Table 12-10 summarizes the existing loads and LAs for these sources. The LAs are a 70.4 percent reduction from the existing loads. LAs are provided for each responsible jurisdiction and input. These loading values (in grams per year) represent the TMDL load allocations (Table 12-10) and each load allocation must be met at the point of discharge. In addition to the LAs presented below for total mercury, an in-lake water column dissolved methylmercury target of 0.081 ng/L also applies.

Table 12-10. **Load Allocations of Mercury in the Lake Sherwood Watershed**

Subwatershed	Responsible Jurisdiction	Input	Area (ac)	Existing Annual Hg Load (g/yr)	Load Allocation[1] (g/yr)
Near Lake Undeveloped	Thousand Oaks	Runoff	70.9	0.043	0.013
Near Lake Undeveloped	Ventura County	Runoff	126	0.077	0.023
Carlisle Canyon	County of Los Angeles	Runoff	1,149	0.708	0.209
Carlisle Canyon	Thousand Oaks	Runoff	50.4	0.031	0.009
Carlisle Canyon	Ventura County	Runoff	2,905	2.32	0.685
Carlisle Canyon	Point Mugu State Park	Runoff	101	0.062	0.018
Lake Surface		Atmospheric Deposition[2]	137	5.27	1.56
Total				**8.51**	**2.51**

[1] Each mass-based load allocations must be met at the point of discharge.

[2] Loads for atmospheric deposition are based on direct precipitation to the lake (calculated by the annual average precipitation multiplied by the surface area of the lake).

12.2.6.3 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL includes both an implicit and explicit MOS for Lake Sherwood. The implicit MOS includes comparing the total mercury concentration reported for fish tissue samples to the methylmercury fish tissue target. Most mercury in fish tissue is in the methyl form, but not all, so this is a conservative assumption. In this TMDL, an explicit MOS is also included by selecting the 95 percent UCL to represent the existing mean fish tissue concentration rather than the regression predicted mean (Figure 12-14). Use of the UCL provides a margin of safety because it represents an upper confidence bound on the long-term exposure concentration. For Lake Sherwood, the fraction of the existing load set aside for the explicit MOS is 0.067, or 2.8 g/yr, which represents 18.5 percent of the loading capacity.

12.2.6.4 Critical Conditions/Seasonality

TMDLs must include consideration of critical conditions and seasonal variation to ensure protection of the designated uses of the waterbody at all times. This TMDL protects beneficial uses by reducing fish tissue concentrations to the FCG target in Lake Sherwood. Because fish bioaccumulate mercury, concentrations in tissues of edible sized game fish integrate exposure over a number of years. As a result, annual mercury loading is more important for the attainment of standards than instantaneous or daily concentrations, and the TMDL is proposed in terms of annual loads. Mercury load is primarily delivered to Lake Sherwood during storm runoff events, so high flows do represent a critical period in terms of peak loading rates.

However, the greatest impact to fish occurs when methylmercury, a more biologically available form of mercury, is at its greatest concentration. Bacterially mediated methylation of mercury varies seasonally and typically results in the greatest methylmercury concentrations in the water column in the late summer.

However, the impact of seasonal and other short-term variability in loading is damped out by the biotic response since the target concentrations in tissues of edible sized game fish integrate exposure over a number of years. Additionally, this TMDL includes a methylmercury water column target applicable year round. This TMDL therefore protects for critical conditions.

12.2.6.5 Daily Load Expression

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. Although it is long-term cumulative load rather than daily loads of mercury that are driving the bioaccumulation of mercury in fish in Lake Sherwood, this TMDL does present a maximum daily load according to the guidelines provided by USEPA (2007). The daily maximum allowable load to Lake Sherwood is calculated from the maximum daily storm volume (estimated from the 99^{th} percentile flow) to the reservoir multiplied by the allowable concentration for mercury consistent with achieving the long-term loading target. These maximum loads are not allowed each day of the year because the annual loads specified by the TMDL must also be achieved. The WLA and LA loads presented above are annual loading caps that cannot be exceeded.

No USGS gage currently exists in the Lake Sherwood watershed. USGS Station 11105500, Malibu Creek at Crater Camp near Calabasas, CA, was selected as a surrogate for flow determination. This gage is located downstream of Lake Sherwood on Malibu Creek. The 99^{th} percentile flow was chosen to represent the peak flow for this drainage. Choosing the 99^{th} percentile flow eliminates errors due to outliers and is reasonable for development of a daily load expression.

The USGS StreamStats program was used to determine the 99^{th} percentile flow for Malibu Creek (355 cfs) (Wolock, 2003). To estimate the peak flow to Lake Sherwood, the 99^{th} percentile flow for Malibu Creek was scaled down by the ratio of drainage areas (10,656 acres/67,200 acres; Lake Sherwood watershed area/Malibu Creek watershed area at the gage). The resulting peak flow estimate for Lake Sherwood is 56.3 cfs.

The event mean concentration for mercury was calculated from the allowable load (12.3 g-Hg/yr; sum of the WLAs and LAs) and the average annual total flow (wet weather plus dry weather 1,492 ac-ft). The resulting concentration (6.70 ng/L) times the peak flow to Lake Sherwood (56.3 cfs) yields a total maximum daily load of 0.922 g-Hg/d associated with the MS4 permittees. For comparison, the existing load (41.7 g-Hg/yr) would yield an event mean concentration of 22.7 ng/L and a daily load of 3.12 g-Hg/d.

12.2.6.6 Future Growth

The majority of land in the Lake Sherwood Watershed is either in agricultural or undeveloped uses. As more development occurs in the watershed, best management practices will need to be employed to maintain the wasteload and load allocations defined in this TMDL. No allocation has been set aside for expansion of permitted discharges in the watershed, such as wastewater treatment facilities, as no facilities currently exist or are planned in the watershed.

If any sources currently assigned load allocations are later determined to be point sources requiring NPDES permits, those load allocations are to be treated as wasteload allocations for purposes of determining appropriate water quality-based effluent limitations pursuant to 40 CFR 122.44(d)(1).

12.3 Implementation Recommendations

Implementation measures may be developed in the future by the Regional Board through an implementation plan, NPDES permits or non-point source enforcement. This section describes USEPA's recommendations to the Regional Board as to the implementation procedures and regulatory mechanisms that could be used to provide reasonable assurances that water quality standards will be met. General information about various lake management strategies can be found in a USEPA document titled *Managing Lakes and Reservoirs (EPA 841-B-01-006)*. Lake management options that can reduce pollutant loading to lakes include but are not limited to: increasing the volume of the lake that is aerated; installing hydroponic islands to remove nutrients; and reducing stormwater discharges by improved infiltration. Additionally, responsible jurisdictions implementing these TMDLs are encouraged to utilize Los Angeles County's Structural Best Management Practice (BMP) Prioritization Methodology which helps identify priority areas for constructing BMP projects. The tool is able to prioritize based on multiple pollutants. The pollutants that it can prioritize include bacteria, nutrients, trash, metals and sediment. Reducing sediment loads would reduce mercury delivery to the lake in many instances. More information about this prioritization tool is available at: labmpmethod.org.

If necessary, these TMDLs may be revised as the result of new information (See Section 12.4 Monitoring Recommendations). The State Board is in the early stages of developing a Statewide Mercury Policy and Mercury Control Program for Reservoirs. According to CEQA scoping materials, the Policy would define an overall structure for adopting water quality objectives; general implementation requirements; and control plans for mercury impaired water bodies. The final structure of the control program could include a total maximum daily load (TMDL) for mercury in reservoirs along with an implementation plan to achieve the TMDL; or an implementation plan that does not rely on a TMDL. How this upcoming policy and program will affect implementation of this TMDL is unknown at this time.

12.3.1 Nonpoint Sources and the Implementation of Load Allocations

Regional Board may regulate nonpoint pollutant sources through the authority contained in sections 13263 and 13269 of the California Water Code, in conformance with the State Water Resources Control Board's Nonpoint Source Implementation and Enforcement Policy, and the Conditional Waiver for Discharges from Irrigated Lands, adopted by the Los Angeles Regional Water Quality Control Board on November 3, 2005. Additionally, South Coast Air Quality Management District has authority to regulate air emissions throughout the basin that affect air deposition. Load allocations are expressed in Table 12-10.

12.3.2 Point Sources and the Implementation of Wasteload Allocations

Wasteload allocations apply to Ventura County, Thousand Oaks, Caltrans as well as the Lake Sherwood Overall Plan Entities (Table 12-9). The mass-based waste load allocations for Caltrans will be incorporated into the Caltrans permit; the Regional or State Board may develop a new permit to cover the previously unpermitted stormwater discharges to the lake from Ventura County, Thousand Oaks and Lake Sherwood Overall Plan Entities.

12.3.3 Source Control Alternatives

Responsible jurisdictions are encouraged to consider the construction of wetland systems and bioswales (or other retention or treatment options) to treat the stormwater, as well as stormwater diversion and infiltration using methods such as porous pavements and rain gardens. Source reduction and pollutant removal BMPs designed to reduce sediment loading are management practices that will also reduce the mercury loading associated with sediments. However, sedimentation basins or water quality ponds that go anoxic at the sediment-water interface may actually result in increased concentrations of methylmercury. This is likely occurring at the mouth of Hidden Valley Wash, where concentrations of methylmercury in the water column and sediments have been observed at levels one order of magnitude greater than other locations in the watershed (see Appendix D, Wet Weather Loading; Appendix G, Monitoring Data).

Dissolved loading associated with stormwater contributes the largest amount of mercury loading to Lake Sherwood. Some of the sediment reduction BMPs may also result in decreased concentrations of mercury in the runoff water. Storage of storm flows in wet or dry ponds may allow for adsorption and settling of mercury from the water column. BMPs that provide filtration or infiltration processes may retain dissolved mercury in the upland areas.

Monitoring of dissolved oxygen levels in these ponds and measurement of total and methylmercury concentrations during warm summer months will assist in the management of these basins to reduce methylmercury loading to Lake Sherwood. Maintaining shallow water levels that do not fluctuate will allow penetration of sunlight, which degrades methylmercury, and reduce the wetting and drying conditions that favor methylation. Existing and ongoing efforts by the Sherwood Valley Homeowner's Association to improve lake water quality by reducing nutrient loading to the lake and improving aeration have likely reduced methylation rates within the lake overall. Further reductions of nutrient levels and improvements to aeration would have the combined benefit of reducing methylation of mercury and implementing load allocations called for in the nutrient TMDL established in 2003.

Unfortunately, sediment reduction BMPs will not mitigate mercury loading from the second largest source in the watershed, atmospheric deposition to the lake surface. Mercury available for deposition in the southwest region typically originates from both local and global sources. In the US, mercury emissions from most facilities have been reduced over the past few decades as the best available technology has improved over the years.

In 2008 USEPA modeled mercury air emissions nationally as a tool for tracking airborne mercury to assist in watershed planning. The mercury emission estimates were principally based on 2001 data. The highest modeled impact in California was located in the Long Beach area and the largest single source contributor was the Long Beach South East Resource Recovery facility which combusts municipal waste to produce electricity. Since that time USEPA has promulgated regulations to reduce mercury from solid waste incinerators and the emissions from this facility and another solid waste incinerator in the City of Commerce have been significantly reduced. In addition to these regulations for solid waste combustors, USEPA is in the process of finalizing regulations for Portland Cement plants which also contribute to reductions in mercury air loading and deposition in the Los Angeles area.

12.4 Monitoring Recommendations

Although estimates of the loading capacity and allocations are based on best available data and incorporate an MOS, these estimates may potentially need to be revised as additional data are obtained. The mass-based loading capacity will be affected by changes in flow volumes; therefore, loading capacities may be reconsidered if significant volume reductions or additions occur. To provide reasonable assurances that the assigned allocations will indeed result in compliance with the fish tissue target, a commitment to continued monitoring and assessment is warranted. The purposes of such monitoring will be 1) to determine compliance with wasteload and load allocations, 2) to determine if numeric targets are being attained, 3) to evaluate whether numeric targets and allocations need to be adjusted to attain beneficial uses, 4) to evaluate the efficacy of control measures instituted to achieve the needed load reductions, and 5) to document trends over time in mercury loading.

To assess compliance with the mercury TMDL, monitoring should include monitoring of largemouth bass (325-375mm in length) fish tissue (skin-off fillets) at least every three years as well as twice yearly sediment and water column sampling in each lake. At a minimum, compliance monitoring should measure the following in-lake water quality parameters: total mercury, dissolved methylmercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids; as well as the following in-lake sediment parameters: total mercury, methylmercury, total organic carbon, total solids and sulfate. Measurements of the temperature, dissolved oxygen, pH and electrical conductivity should also be taken throughout the water column with a water quality probe along with Secchi depth measurement. Additionally, in order to accurately calculate compliance with allocations expressed in yearly loads, monitoring should include flow estimation or monitoring as well as water quality concentration measurements. At Lake Sherwood wasteload allocations are assigned to stormwater inputs from various subwatersheds. These sources should be measured near the point where they enter the lake twice a year for at minimum: total mercury, methyl mercury, chloride, sulfate, total organic carbon, alkalinity, total suspended solids, and total dissolved solids.

It may also be helpful to investigate potential sources of methylmercury loading in the watershed, such as wetlands, sedimentation basins, and any areas impacted by forest fires. Specifically, there are several springs that have been documented to feed the lake, however, sampling of these sources was not possible during USEPA site visits. Springs sometimes have elevated mercury so finding these springs and measuring their mercury loading is a high priority activity to fully characterize all sources to Lake Sherwood. Additionally, sampling conducted for the development of this TMDL found that one of the tributaries to the lake (Hidden Valley Wash) discharged to the lake through a sedimentation basin with high methylation. Further sampling of tributaries to determine other likely methylation sources may be appropriate in order to target BMPs.

The mercury TMDL for Lake Sherwood concludes that a reduction in total mercury loading to the lake of 70.4 percent will result in compliance with the fish tissue target of 0.22 ppm. As an example of concentrations that responsible jurisdictions may need to target in order to meet and comply with the mass-based WLAs and LAs, this discussion provides concentrations calculated based on existing flow volumes (a recalculation is needed if flow volumes change). Assuming flow volumes remain at existing levels (Appendix D, Wet Weather Loading; Appendix F, Dry Weather Loading), targeted concentrations of total mercury in stormwater from the Northern, Western, Hidden Valley Wash, and Near Lake Developed subwatersheds may be 9.94 ng/L, 2.92 ng/L, 10.38 ng/L, and 9.94 ng/L. The targeted concentration in the stormwater from the Caltrans area in the Carlisle Canyon subwatershed may be 4.37 ng/L. Similarly, the targeted concentration of total mercury in runoff from the Near Lake Undeveloped subwatershed may be 2.06 ng/L. The targeted concentration in the runoff from the non-Caltrans areas in the Carlisle Canyon subwatershed may be 2.32 ng/L. As stated above, these concentrations are provided as guidelines; however, mass-based WLAs must be achieved. An in-lake water column dissolved methylmercury target of 0.081 ng/L also applies.

13 Westlake Lake TMDL

Westlake Lake (#CAL4042500019990201153000) is listed as impaired by lead (SWRCB, 2010). (Note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL.) Comparison of metals data to their associated hardness-dependent water quality objectives indicates that lead is currently achieving numeric targets at Westlake Lake. Therefore, a TMDL is not included for this pollutant. These analyses are presented below (see Section 13.2).

13.1 Environmental Setting

Westlake is located in the Santa Monica Bay watershed (HUC 18070104) in Los Angeles and Ventura counties, near the city limits of Thousand Oaks and Westlake Village (Figure 13-1). The private lake was completed in 1976 following the construction of a dam in 1973. According to the Urban Lakes Study (UC Riverside, 1994), the average depth is approximately six feet. Dense residential development surrounds the lake and covers a private, multi-lobed island. Additionally, a small business complex, including a yacht club and restaurants, is located on the southeast shore of the lake. The private lake is the site of heavy recreation, predominately boating and catch and release fishing, as swimming is prohibited (Figure 13-2 and Figure 13-3). Catfish are periodically stocked in the lake (July 17, 2009 USEPA Field Notes).

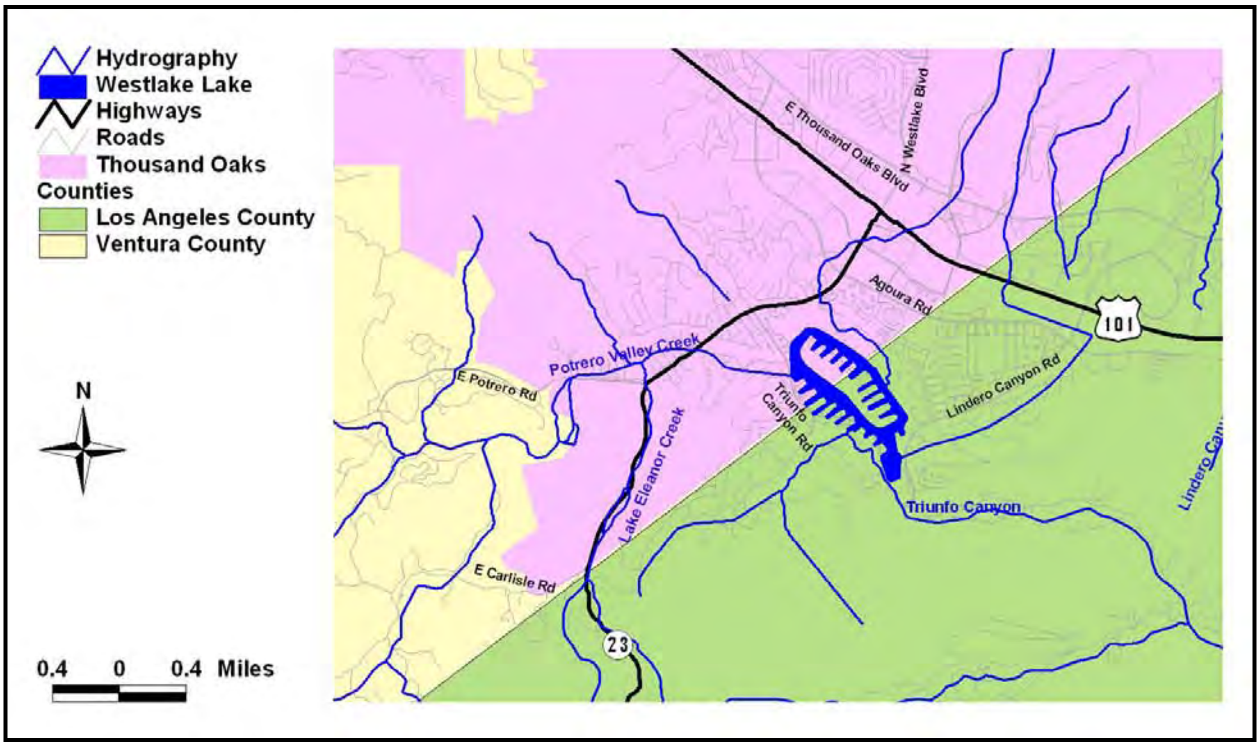

Figure 13-1. **Location of Westlake Lake**

Figure 13-2. **View of Westlake Lake**

Figure 13-3. **Sampling Location is at the Buoys (WL1)**

Westlake was included in the Urban Lakes Study (UC Riverside, 1994). The primary inflows to the lake were identified as Potrero Valley Creek, two wells, six storm drains (Figure 13-4), and runoff from the upstream watershed. Westlake Lake discharges to Triunfo Canyon Creek.

As of the writing of this TMDL, there are no non-MS4 NPDES permitted discharges in the Westlake Lake watershed. This includes non-stormwater discharges (individual and general permits) as well as general stormwater permits associated with construction and industrial activities.

Figure 13-4. **Sampling Location in Front of Storm Drain (WL3)**

13.2 LEAD IMPAIRMENT

Westlake Lake was listed as impaired for lead in 1996 based on an assessment in the Regional Board's Water Quality Assessment and Documentation Report (LARWQCB, 1996). Consistent with project plan recommendations provided in California's Impaired Waters Guidance (2005), EPA and local agencies collected 24 additional samples between March 2009 and October 2010 to evaluate current water quality conditions. There were zero dissolved lead exceedances in 24 samples (Appendix G, Monitoring Data). USEPA also collected two sediment samples during August 2010 to further evaluate lake conditions. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target (Appendix G, Monitoring Data). Therefore, Westlake meets lead water quality standards, and USEPA concludes that preparing a TMDL for lead is unwarranted at this time. USEPA recommends that Westlake Lake not be identified as impaired by lead in California's next 303(d) list.

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Appendix A. Methodology for Nutrient TMDL Development

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A.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure A-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

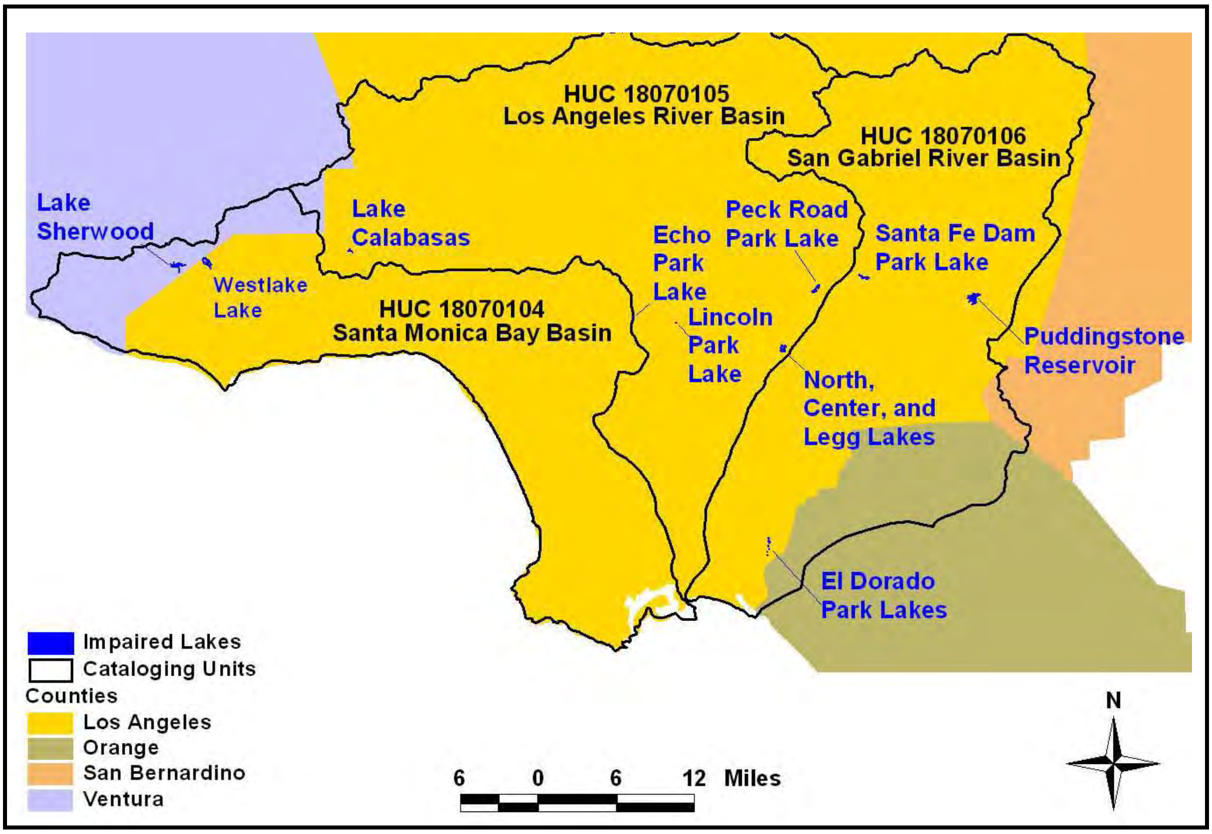

Figure A-1. Location of Impaired Lakes

Eight of these waterbodies have impairments that may be due to elevated nutrient levels: Peck Road Park Lake, Echo Park Lake, Lincoln Park Lake, Lake Calabasas, the El Dorado Park lakes, Legg Lake, Puddingstone Reservoir, and Santa Fe Dam Park Lake. These impairments include algae, ammonia, eutrophication, low dissolved oxygen/organic enrichment, odor, and pH. A steady-state lake response model has been set up for each impaired lake to determine whether or not eutrophication is the primary cause of these impairments. This appendix discusses the problems associated with eutrophication, sources of nutrient loading, and the approach used for determining loading capacities for nitrogen and phosphorus based on observed and simulated levels of chlorophyll *a*.

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A.2 Conceptual Model: Nutrients, Algae, and Eutrophication

Excessive algal growth in the urban lakes of the Los Angeles region has resulted in several waterbodies not supporting their designated beneficial uses associated with aquatic life and recreation (LARWQCB, 1996). Unaesthetic amounts of algal biomass can directly impair swimming and wading recreational uses. Algal growth in some instances has produced algal mats in the lakes (UC Riverside, 1994). Excess growth of algae can also result in loss of invertebrate taxa through habitat alteration (Biggs, 2000). In addition, ammonia, a nitrogen compound, has been measured at concentrations exceeding objectives designed to protect aquatic life (LARWQCB, 1996).

Rates of algal growth depend on the availability of nutrients, light, and other factors. Stimulation of excess algal growth by nutrient loading is referred to as eutrophication. There are many biological responses to nutrients (nitrogen and phosphorus) in lakes. The biologically available nutrients and light will stimulate phytoplankton and or macrophyte growth. As these plants grow, they provide food and habitat for other organisms such as zooplankton and fish. When the aquatic plants die, they will release nutrients (ammonia and phosphorus) back into the water through decomposition. The decomposition of plant material consumes oxygen from the water column; in addition the recycled nutrients are available to stimulate additional plant growth. Physical properties such as light, temperature, residence time, and wind mixing also play integral roles throughout the pathways described.

These typical biological processes can become over-stimulated by the addition of excess nutrients to a waterbody and create a situation in which water quality becomes degraded and beneficial uses are impaired. The following flow chart (Figure A-2) outlines the responses within a lake to excessive nutrient loading and how the beneficial uses will be impacted.

Excessive nutrient loading, from either external or internal processes, can cause excessive phytoplankton and macrophyte growth. The resulting plant biomass may cause increased turbidity, altered planktonic food chains, unaesthetic conditions, reduced dissolved oxygen concentrations, and increased nutrient recycling (Figure A-2). These changes can lead to a cascade of biological responses culminating in impaired beneficial uses.

Typically, excessive plant growth can quickly lead to an altered planktonic community; in many cases the dominant phytoplankton species may become blue-green algae (cyanophytes) and algal blooms may occur, especially in the summer months. These blooms cause fluctuations in dissolved oxygen concentration and pH that can negatively affect aquatic life in the waterbody. Senescence and decay of the biomass present in algal blooms may also cause problems with scum and odors that affect recreational uses of the affected waterbody. Likewise, macrophyte growth may increase and become expansive throughout the lake (Figure A-2). Particularly in shallow lakes, the combination of available nutrients and greater light intensity throughout the water column provides the light that is needed for rapid plant growth. In addition, light can penetrate to the bottom of shallow lakes, promoting macrophyte growth. In comparison, in deep lakes a greater portion of the water column is not able to support photosynthesis as a majority of the water column is below the light penetration depth. Thus, the impacts of nutrient loading and the biological response of planktonic algae and macrophytes are often very apparent in shallow lakes.

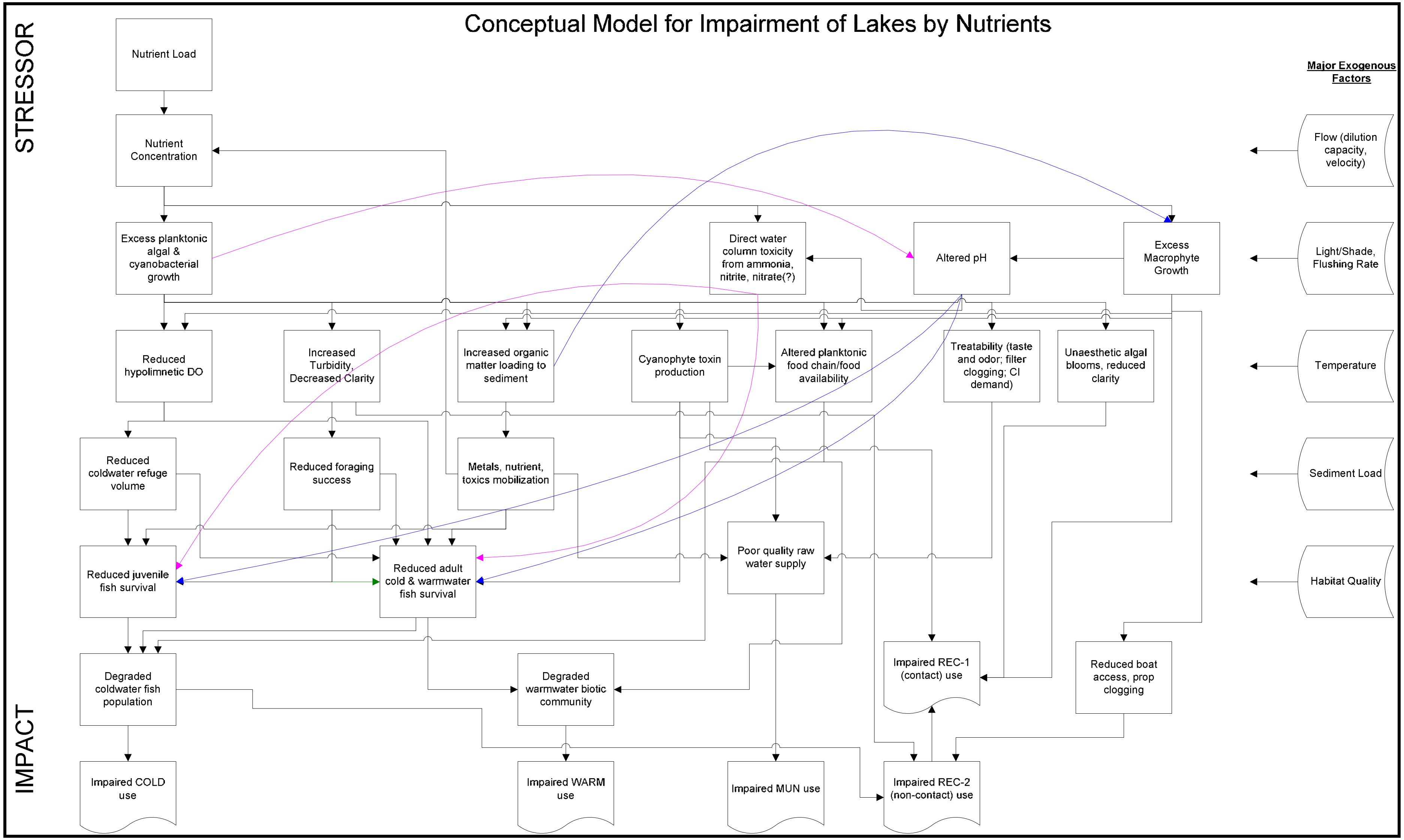

Figure A-2. Conceptual Model for Lakes

As noted above, eutrophication can also lead to increased daytime pH in lakes due to rapid uptake of carbon dioxide by photosynthesizing algae. The elevated pH creates a harmful environment for organisms and can increase the concentration of un-ionized ammonia, potentially leading to direct toxicity to fish and other organisms. Dense algal populations also cause diurnal swings in dissolved oxygen concentrations, as oxygen is released during daytime photosynthesis and consumed during nighttime respiration. Decomposition of algal biomass can consume oxygen and dramatically reduce the oxygen levels found in the lake. Low dissolved oxygen levels can become very stressful for fish and other organisms and may in fact lead to fish kills (Figure A-2). Moreover, as the plant material is decomposed, the nutrients are released and will recycle through the system. Shallow lakes tend to have increased biological productivity because it is likely that the photosynthetic zone and decomposition zone of the water column overlap, creating the situation where as materials are decomposed and the nutrients released, they are also immediately available for photosynthesis and plant growth continuing to drive ongoing impairments.

Control of the deleterious effects of eutrophication in lakes typically requires reduction in nutrient loads. Both external and internal (recycled) nutrient loads may need to be addressed.

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A.3 Source Assessment

Sources of nutrient loading to a lake may include both point and nonpoint sources. For the purposes of allocating loads among nutrient sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which wasteload allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). This section describes how the loading from point and nonpoint sources were estimated.

A.3.1 POINT SOURCES

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process.

A.3.1.1 MS4 Permittees

In 1990 USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed by stormwater runoff into the Municipal Separate Storm Sewer Systems (MS4), or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

Nitrogen and phosphorus loads from urban stormwater runoff are estimated from event mean concentration (EMC) data and flows predicted from calibrated watershed models (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to an MS4 for each lake. The county of Los Angeles and the Southern California Coastal Water Research Project (SCCWRP) have been collecting pollutant concentration data for storm events in the county of Los Angeles for representative land use classes. These concentrations can be applied to the flow volumes predicted by the LSPC models for each land use to estimate average wet weather nutrient loading to each lake. Appendix D (Wet Weather Loading) describes the datasets, assumptions, and loading results for this analysis.

These systems may also discharge during dry weather as a result of irrigation, car washing, etc. Estimation of nutrient loading from MS4 systems in dry weather is based on SCCWRP regional studies and is described in Appendix F (Dry Weather Loading).

A.3.1.2 Non-MS4 NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). To quantify nutrient loading from non-MS4 NPDES discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the Los Angeles River, San Gabriel River, and Santa Monica Bay Basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the eight nutrient-impaired lakes. Nutrient loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake specific sections of this report.

A.3.1.3 Additional Inputs

Several of the lakes addressed by this TMDL have additional point source inputs that do not currently have NPDES permits. Most are supplemental flows from groundwater wells or potable water that maintain lake levels. Information pertaining to flow volumes from these sources was provided by park staff at each lake (generally based on water usage information from the water suppliers). Where accessible, the Regional Board and USEPA sampled water quality from these inputs during the 2009 sampling events. In some cases, the suppliers were able to provide nutrient concentrations. Nutrient loading was calculated from average nutrient species concentration data and an estimate of annual flow volumes to each lake.

A.3.2 NONPOINT SOURCES

Nutrient loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

A.3.2.1 Internal Loading from Lake Sediments

Lake sediments typically store phosphorus that has sorbed to soil particles or settled to the bottom of the lake following the decomposition of organic matter. When these sediments become hypoxic (i.e., when dissolved oxygen concentrations become low) they may release stored phosphorus into the water column which then becomes available for uptake by plants and algae. In some lakes, internal phosphorus loading may comprise a significant portion of the total load.

Hypoxic conditions also promote release of dissolved ammonia from the sediments. Lake sediments do not typically store and release significant quantities of nitrogen relative to other lake inputs. However, the net nitrogen sedimentation rate calibrated for each lake accounts for internal loading of nitrogen as well.

Intensive monitoring studies are typically required to accurately quantify internal nutrient loading. This level of information was not available for the lakes addressed by this TMDL. Though the internal load may not be quantified for these lakes, it is reflected in the net (settling minus resuspension) nutrient sedimentation rates calibrated for each lake (Section A.4.2).

Internal loading is discussed in more detail in Appendix B (Internal Loading).

A.3.2.2 Wind Resuspension

As wind moves across a lake surface, the resulting wave action may disturb lake sediments in shallow areas and release additional stored phosphorus. Appendix B (Internal Loading) describes the impacts of wind resuspension and defines the critical lake levels where additional internal loading may occur. As wind resuspension impacts internal loading rates, the effects were accounted for in the net sedimentation rates for phosphorus and nitrogen.

A.3.2.3 Bioturbation

Bottom feeding fish and benthic macroinvertebrates can also disturb lake sediments and promote release of stored nutrients. As bioturbation further impacts internal loading rates, the effects were accounted for in the net sedimentation rates for phosphorus and nitrogen.

A.3.2.4 Atmospheric Deposition

The National Atmospheric Deposition Program (NADP) monitors wet nitrogen deposition (as nitrate) at two active and two inactive stations in southern California. Isopleth maps were downloaded from the NADP website and brought into a GIS environment to extract site specific precipitation-weighted annual average nitrate concentrations for grid cells overlaying each lake. NADP has produced these isopleth maps for years 1994 through 2006. The time series was extended to previous years by developing a regression equation for each location based on year and cumulative precipitation (Appendix E, Atmospheric Deposition).

The precipitation-weighted annual average nitrate concentrations were then multiplied by the annual rainfall observed at the nearest weather station (Appendix D, Wet Weather Loading)) and the lake surface area to estimate nitrogen loading from atmospheric deposition to each lake surface. Deposition to land surfaces is accounted for in the loading estimates from the watersheds (Appendices D and F; Wet and Dry Weather Loading, respectively).

Unlike nitrogen, phosphorus does not have a significant gaseous phase, and atmospheric deposition is primarily due to fugitive dust. Phosphorus deposition rates are typically much lower than nitrogen deposition rates and are not included in the NADP monitoring program. At this time, measurements of phosphorus deposition rates are not available for this area. SCCWRP has recently begun a deposition monitoring study that will measure phosphorus, but the results are not expected to be published until 2011.

The datasets, assumptions, and resulting loading from atmospheric deposition are described in detail in Appendix E (Atmospheric Deposition).

A.3.2.5 Wet Weather Loading

Nitrogen and phosphorus loads from areas that do not drain to an MS4 system are estimated from event mean concentration (EMC) data and flows predicted from calibrated watershed models (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate nonpoint source runoff volumes, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4. The county of Los Angeles and SCCWRP have been collecting pollutant concentration data for storm events in the county of Los Angeles for representative land use classes. These concentrations can be applied to the flow volumes predicted by the LSPC models for each land use to estimate average nutrient loading to each lake. Appendix D (Wet Weather Loading) describes the datasets, assumptions, and loading results for this analysis.

A.3.2.6 Dry Weather Loading

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. Nonpoint sources during dry weather include irrigation, fertilization of adjacent parkland, and other miscellaneous urban sources. Estimation of dry weather pollutant loading is discussed in Appendix F (Dry Weather Loading).

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A.4 Linkage Analysis

To simulate the impacts of nutrient loading on each impaired lake, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006). For these TMDLs, target nutrient loads and resulting allocations were determined specifically for each lake based on the secondary target – summer season mean chlorophyll *a* concentration.

Other parameters may be chosen as secondary targets for determining nutrient allocations. Chlorophyll *a*, however, is the best choice for assessing nutrient impacts alone. For example, choosing dissolved oxygen as a secondary target will not only account for fluctuations in concentration caused by algal photosynthesis and respiration but will also include response to loading of organic matter not associated with algal decay (e.g., loading from wastewater treatment plants, organic fertilizers, etc.). The existing dissolved oxygen criteria serves as an additional target in these TMDLs. Light penetration, often measured as Secchi depth, is another indicator of nutrient impairment as greater densities of algae block sunlight penetration and reduce Secchi depth. Light penetration is also impacted by suspended sediment concentrations and low Secchi depth does not always correlate with excessive nutrient loading. This is often the case for waterbodies located in watersheds comprised of silt and clay soils or in areas undergoing land clearing and construction. Thus chlorophyll *a* is the most appropriate parameter for assessing the direct impacts of eutrophication on a waterbody. This section describes how the NNE Tool simulates chlorophyll *a* and its use in developing the nitrogen and phosphorus TMDLs.

A.4.1 Model Description

The USACE developed the BATHTUB model (Walker, 1987) to predict eutrophication in reservoirs across the country. BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. These processes are accounted for implicitly in the model through the calibration of the net sedimentation rates. If an estimate of internal loading of phosphorus is required, the following methodology described by Nürnberg (1984) provides a reasonable estimate:

TPinlake = TPinflow * (1-Rpred) + Lint / Qs , where

Rpred = 15 / (18 + Qs)

TPinlake = mean summer in-lake phosphorus concentration

TPinflow = mean summer tributary phosphorus concentration

Qs = mean depth over hydraulic residence time

Rpred = annual retention due to sedimentation

Lint = internal phosphorus load ($mg/m^2/yr$)

Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality. Algal concentrations are predicted for the summer season when water quality problems are most severe. Annual differences in water quality, or differences resulting from different loading or hydrologic conditions (e.g., wet vs. dry years), can be evaluated by running the model separately for each scenario.

BATHTUB first calculates steady-state phosphorus and nitrogen balances based on nutrient loads, nutrient sedimentation, and transport processes (lake flushing, transport between segments, etc.). Several options are provided to allow first-order, second-order, and other loss rate formulations for nutrient sedimentation that have been proposed from various nutrient loading models in the literature. The resulting nutrient levels are then used in a series of empirical relationships to calculate chlorophyll *a*, oxygen depletion, and turbidity. Phytoplankton concentrations are estimated from mechanistically-based steady-state relationships that include processes such as photosynthesis, settling, respiration, grazing mortality, and flushing. Both nitrogen and phosphorus can be considered as limiting nutrients, at the option of the user. Several options are also provided to account for variations in nutrient availability for phytoplankton growth based on the nutrient speciation in the inflows. The empirical relationships used in BATHTUB were derived from field data from many different lakes, including those in USEPA's National Eutrophication Survey and lakes operated by the Army Corps of Engineers. Default values are provided for most of the model parameters based on extensive statistical analyses of these data.

In 2006, Tetra Tech developed the NNE BATHTUB Tool as a simplified method for predicting summer season chlorophyll *a* lake response to a number of inputs. The NNE BATHTUB Tool is a risk-based approach for estimating site-specific nutrient numeric endpoints (NNE) for California waters (Tetra Tech, 2006). The Tool has been tested for several waterbodies in California as a series of case studies (e.g., Tetra Tech, 2007).

The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations necessary to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth.

For both the nitrogen and phosphorus simulations, the NNE BATHTUB Tool has been set up to incorporate the USACE BATHTUB Model default equations for simulating nutrient sedimentation rates. In accordance with the USACE BATHTUB Model Users Manual (Walker, 1987), the NNE Tool incorporates a calibration factor on each sedimentation rate to improve model fit to observed data.

The NNE BATHTUB Tool simulates phosphorus (P) using the 2nd-order P-sedimentation model (presented as P Model 2 in Walker, 1987):

P Sedimentation Rate (mg/m^3-yr) = $K_P \cdot A1 \cdot P^2$,

where P is the total phosphorus concentration in µg/L.

This yields a solution for P:

$$P = \frac{\sqrt{1 + 4K_P\,A1\,Pi\,T} - 1}{2K_P\,A1\,T}, \text{ where}$$

A1= 0.056 Qs/[Fot · (Qs + 13.3)]

Pi = inflow total P concentration (µg/L)

Qs = overflow rate (m/yr), with a minimum of 4

Fot = ratio of inflow ortho P to inflow total P

K_P = P calibration factor, typically ranging from 0.5 to 2.0

T = hydraulic residence time (yr) = Volume/Inflow-per-yr

The nitrogen (N) simulation is implemented using the 2nd order N-sedimentation (presented as N Model 2 in Walker, 1987):

N Sedimentation Rate (mg/m^3-yr) = $K_N \cdot B1 \cdot N^2$,

where N is the total nitrogen concentration in µg/L. This yields a solution for N:

$$N = \frac{\sqrt{1 + 4 K_N\ B1 Ni T} - 1}{2 K_N\ B1 T}$$, where

B1 = 0.0035 Qs/[Fin$^{0.59} \cdot$ (Qs + 17.3)]

K_N = N calibration factor, typically ranging from 0.3 to 3.0

Ni = inflow total N concentration

Fin = ratio of inflow inorganic N to inflow total N

The USACE BATHTUB Model allows the user to choose from five empirical equations for chlorophyll *a* simulation. The NNE Tool incorporates the equation that considers light, flushing rate, and nutrient concentrations to account for the co-risk factors whose cumulative effect determines algal density (presented as Chl Model 1 in Walker, 1987). A calibration factor on simulated chlorophyll *a* concentration allows the user to improve the model fit based on observed data:

$$Chl - a = \frac{K_C\ Bx}{(1 + 0.025 Bx\, G)(1 + G\, a)}$$, where

Bx = Xpn$^{1.33}$/4.31

Xpn = [P^{-2} + ((N-150)/12)$^{-2}$]$^{-0.5}$

K_C = Chl-a calibration factor

G = Zmix $\cdot$ (0.14 + 0.0039 Fs)

Zmix = mixed depth (m)

Fs = (summer) flushing rate = (inflow – evap)/vol

A = non-algal turbidity (m^{-1}).

The NNE BATHTUB Tool uses Visual Basic’s GoalSeek function to find combinations of N and P loading that result in predicted chlorophyll *a* being equal to the selected target. Because algal growth can be limited by either N or P there is not a unique solution, and the Tool output supplies the user with a curve representing the loading combinations that will result in attainment of the selected chlorophyll *a* target.

Spatial variability in water quality can be simulated with BATHTUB by dividing a lake horizontally into segments, and calculating transport processes such as advection and dispersion between the segments. This is appropriate for large lakes, particularly lakes with multiple sidearms and tributary inflows, that have substantially different water quality in different portions of the lake. However, this was not necessary for the lakes addressed in this TMDL report due to their generally small to moderate sizes, and the lack of detailed data demonstrating significant spatial variations in lake characteristics and water quality. Therefore, the NNE BATHTUB Tool was applied as a whole lake model to each waterbody. In some cases, a chain of multiple lakes was combined into a single lake system for modeling because the multiple lakes had similar characteristics or they functioned essentially as a single lake. The lake-specific chapters describe details associated with each lake model.

A.4.2 MODEL SETUP AND CALIBRATION TO EXISTING CONDITIONS

The NNE BATHTUB Tool was set up individually for each impaired lake or lake system. Bathymetry data for each lake were acquired from various sources to represent the general characteristics of the waterbody, such as surface area, volume, and average depth. The lake specific bathymetry data are discussed in each lake chapter of the TMDL report.

Cumulative nitrogen and phosphorus loads were calculated as a sum of all known, quantifiable sources. Sources of loading resulting from wet weather are discussed in Appendix D; Appendix F summarizes the loading originating during dry weather conditions. Atmospheric deposition to each lake surface is quantified in Appendix E. Internal nutrient loading is discussed in Appendix B, but is not quantified directly due to lack of data (the BATHTUB model accounts for internal loading indirectly by using a net sedimentation rate [sedimentation minus resuspension]). Prior to calibration of the BATHTUB model, the user must determine the appropriate averaging period by calculating the nutrient turnover ratio (Walker, 1987). Average external loading rates are calculated for the summer season (May through September) and for the year. These loads are compared to the mass of nutrients stored in the waterbody (average nutrient concentration times volume) to calculate the mass residence time. Dividing the length of the averaging period (1.0 yr for the annual averaging period or 0.42 yr for the summer season period) by the mass residence time yields the nutrient turnover ratio. The averaging period for the model should be selected such that the nutrient turnover ratio for the limiting nutrient is greater than or equal to 2. The following equations apply:

Mass Residence Time (yr) = Nutrient mass in waterbody (lb) / External nutrient loading (lb/yr)

Nutrient Turnover Ratio = Length of the averaging period (yr) / Mass Residence Time (yr)

Once the bathymetry and loading inputs corresponding to the correct averaging period were input, each model was calibrated to observed conditions. Simulated phosphorus concentrations were compared to the average summer season concentrations based on data collected since the early 1990s (Appendix G, Monitoring Data). The calibration factor, K_P, was adjusted until the simulated concentration approximated those observed. The calibration process was repeated using K_N for nitrogen and K_C for chlorophyll *a*.

For some of the lakes, there are other sources of loading associated with the parkland area for which loading estimates were not available (Appendix F, Dry Weather Loading). Examples include inputs from excessive fertilization relative to product recommendations and runoff of nearby residential areas (through the storm drain system or nonpoint source) where fertilizer application rates were unknown, leaking wastewater infrastructure serving visitors at adjacent parks, natural wildlife populations, and abnormally high wildlife populations caused by feeding and inappropriate trash disposal along the shorelines of park lakes. Loads in this additional parkland loading category were quantified using the NNE BATHTUB model by increasing the inputs until simulated concentrations of total phosphorus and total nitrogen matched those observed. The chlorophyll *a* concentrations were then calibrated using K_C.

A.5 TMDL Development

The TMDL is defined by the loading capacity. A waterbody's loading capacity represents the maximum amount of pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). For nutrients, this is the maximum amount of nitrogen and phosphorus loading consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as an average summer concentration in each impaired lake. Selection of the chlorophyll *a* target is discussed in Section 2.2.3.

A.5.1 Loading Capacity and Allocations

The NNE BATHTUB Tool outputs a matrix of nitrogen and phosphorus loads consistent with achieving the chlorophyll *a* target. For lakes where the calibrated chlorophyll *a* concentration is less than the target, it was assumed that the loading capacity is not exceeded under existing conditions and no reductions in nitrogen or phosphorus are required. For those lakes where the chlorophyll *a* concentration is greater than the target and loading reductions are required, the loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected. This ratio was chosen to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). A ratio of 10 typically limits the growth nuisance species, such as cyanobacteria (blue green algae) (Welch and Jacoby, 2004).

The loading capacity for each nutrient is expressed as pounds per year (lb/yr). The values are further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

Existing loads, loading capacity, WLAs, LAs, and MOS are presented for each individual waterbody or lake system in the respective lake chapters of this TMDL report. As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on simulation of allowable loads with the NNE BATHTUB model and using a ratio close to 10. These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes and, therefore, differ from concentrations associated with various inflows. Each lake chapter also presents nutrient concentrations associated with the WLA and LA inputs. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

A.5.2 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. The nutrient TMDLs for these lakes are based on simulated nitrogen and phosphorous concentrations and include a 10 percent explicit margin of safety when reductions are required. For lakes not currently exceeding the numeric targets, the loading capacity has been set to existing conditions as an antidegradation measure; hence, the MOS is implicitly applied in the TMDL development.

A.5.3 DAILY LOAD EXPRESSION

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River TMDL. The TMDLs developed here each include a daily maximum load estimate consistent with the guidelines provided by USEPA (2007). Because the majority of loads occur during wet weather events, the maximum allowable daily load is calculated from the 99th percentile flow multiplied by the average allowable concentration consistent with achieving the long-term loading targets. In lakes where the majority of loads are associated with supplemental water additions, appropriate flow rates are determined and multiplied by the average allowable concentration to determine the maximum allowable daily load.

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Appendix B. Internal Loading from Lake Sediments

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B.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure B-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). The waterbodies are impaired by combinations of low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

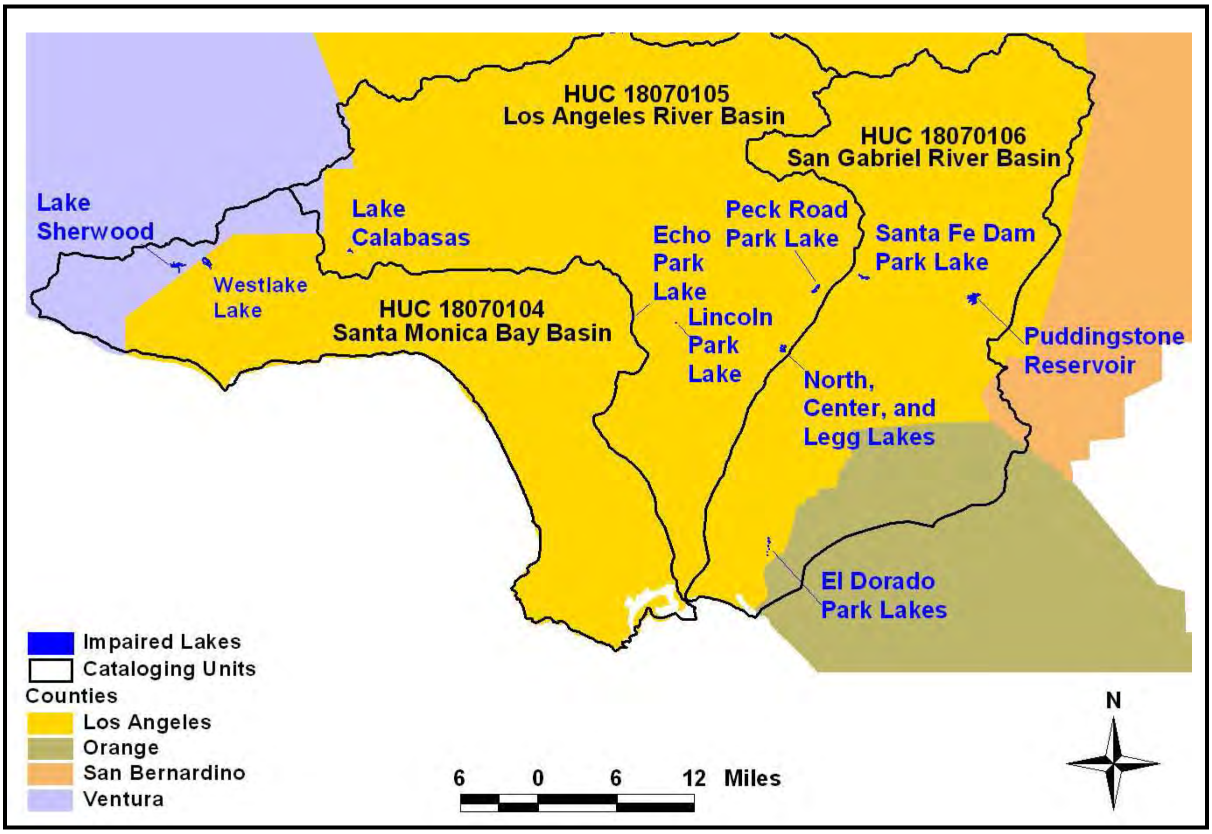

Figure B-1. Location of Impaired Lakes

Internal loading from the lake sediments of impaired waterbodies can be a significant source of pollutant loading, particularly for phosphorus, mercury, and Organochlorine (OC) Pesticides and PCBs. This appendix provides a general overview of the mechanisms that affect rates of internal loading. Although processes affecting internal loads of all pollutants are discussed, internal loads of phosphorus and mercury will not be quantified in the TMDLs because the linkage analyses implicitly account for these mechanisms. For phosphorus, the NNE BATHTUB Tool (Appendix A, Nutrient TMDL Development) accounts for resuspension from internal sediments by applying a net sedimentation rate for phosphorus. For mercury, fish tissue bioaccumulation data reflect both the external and internal loading of methylmercury to the waterbody (Appendix C, Mercury TMDL Development). In addition, loads of phosphorus and mercury continue to enter the impaired waterbodies, although mercury is likely delivered at lower levels than seen previously.

For OC Pesticides and PCBs, the fish tissue bioaccumulation data reflect all sources of loading; however, historic accumulation and internal releases are likely the predominant source of loading to Puddingstone Reservoir, Peck Road Park Lake, and Echo Park Lake as the use of chlordane, DDTs, dieldrin, and PCBs is no longer allowed in the U.S. Thus quantifying internal loading of these pollutants was an important component of TMDL development. Estimation of internal recycling rates of OC Pesticides and PCBs is discussed in Appendix H (Organochlorine Compounds TMDL Development).

This appendix discusses the general process of internal loading from lake sediments and the conditions that tend to increase rates of release of the contaminants addressed by this TMDL report.

B.2 Historic Sediment Stores

External loads of pollutants can enter lakes in surface flow, in groundwater discharge, and by direct atmospheric deposition. Once entering the lake, pollutants may be discharged downstream, degrade, volatilize back to the atmosphere, or settle to the sediment.

Over time nutrients, metals, and OC Pesticides and PCBs that are particle reactive tend to settle and accumulate in sediment on a lake's bottom. The net rate of settling is dependent upon the particular lake's dynamics as well as the compound's chemical characteristics. Sedimentation can also create a concentration gradient in the water column, where water near the water-sediment interface tends to harbor higher concentrations of nutrients, metals, or OC Pesticides and PCBs than the shallower lake levels.

Once material is translocated to the sediment several conceptual pathways may be followed:

- The material may remain in the shallow sediment layers with the potential for continued exchange with the water column and biota.
- The material may degrade or be sequestered in permanently insoluble forms within the sediment, resulting in a net loss of active pollutant mass.
- The material may be buried with clean sediment (either from upland erosion processes or a capping project), sequestering at a depth that minimizes interaction with the water column.
- The material may be released back to the water column.

Release processes such as diffusive exchange, bioturbation, and sediment disturbance by wind mixing or dredging activity can release historical sediment stores, returning pollutants to the water column. For these reasons, sedimentation can act as both a sink and a source of these contaminants in lakes. Refer to Section 2 for a more detailed discussion on the determination of sediment targets.

Most OC Pesticides and PCBs are generally banned from use and no longer manufactured in the US. Despite these efforts, historical loading and sedimentation has often caused a situation in which elevated concentrations continue to be found in lake sediment stores. External loading rates of phosphorus and metals have also often declined over time with better management practices, but historical elevated loading may result in significant stores present in lake sediments. Releases of sediment stores of these compounds may comprise a significant portion of the total load to a lake's water column. For example, internal loading can account for a substantial amount of the total phosphorus within the water column (Moore et al., 1998), creating a situation in which, despite reduction in external loading, phosphorus concentrations in lake water remain high and cause continued impairment (Bachmann, 2005). Authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between methylmercury in the water and fish tissue, when normalized to fish length. Further, elevated concentrations of OC Pesticides and PCBs in fish tissue can occur as a direct result of food chain pathways that lead back to worms and other invertebrates that feed in contaminated sediments, even when water column concentrations meet all applicable criteria (Thomann et al., 1992).

Estimation of the total mass of pollutants stored in sediment is difficult. Concentrations in sediment often vary by orders of magnitude over short distances in both the lateral and vertical dimension, so large amounts of samples are often needed to obtain an accurate characterization of the sediment storage. Historical bathymetric data can assist in determining the net rates of sediment accumulation. This could be used to obtain rough estimates of sediment storage if combined with assumptions about the changes in concentrations on influent sediment over time.

In theory, removal of contaminated sediment could reduce the amount of accumulated pollutants available for exchange into the water column and biota. Unfortunately, the removal process may disturb and release the metals, nutrients, or OC Pesticides and PCBs, returning these constituents to the water column, and thereby increasing the bioavailability of the compounds. Additionally, removal of the top layers of sediment may uncover more contaminated layers deposited in past decades when the use and management of the pollutants was less adequately controlled in the US.

As an alternative to dredging removal, highly contaminated sediments are sometimes sequestered with engineered caps to prevent releases to the water column. Both approaches are very costly, and are thus most often used at highly contaminated Superfund sites. Less expensive techniques attempt to reduce rates of release relative to the processes discussed in the following sections. For example, oxygenating the bottom water can minimize releases that are facilitated by anoxia, while manipulation of lake levels can sometimes reduce resuspension due to wind mixing. For some pollutants, chemical treatments can be useful. For instance, alum is often used to reduce phosphorus recycling in lakes by converting phosphorus to insoluble precipitates.

B.3 Thermal Stratification and Wind Mixing

Lakes located in the Los Angeles region are exposed to extreme heat during the summer months. Cycles of warming and cooling due to seasonal variations impact the release of suspended sediments and associated constituents into the water column. Several of the lakes addressed by this TMDL report are also relatively shallow and are subject to wind mixing which may disturb lake sediments and associated pollutants and further impair water quality.

Thermal stratification refers to the process in which a warm layer of water develops in the epilimnion (the upper level of a stratified lake) due to the transfer of solar energy, while deeper waters remain cooler and, sometimes, anoxic, particularly in the summer (see Section B.4 for more details on anoxic conditions). The difference in temperature causes a density gradient and increased resistance to mixing between the upper and lower lake depths (cooler water being more dense), which limits the exchange of water and compounds between the layers and typically results in epilimnotic concentrations of sediment-associated pollutants being less than those found in deeper waters. The greater the temperature differential, the more resistant the water column is to vertical mixing. Stratified conditions remain until the thermal density gradient disappears due to cooling of the surface water or wind energy is able to overcome the remaining density gradient, allowing the water to mix; this process is referred to as lake turnover. As deep waters rise to the surface, they may transport significant amounts of sediment-associated pollutants (e.g., metals, nutrients, and OC Pesticides and PCBs) that were released during periods of stratification into surface waters where they may exacerbate algal growth or contaminate fish tissue.

Wind mixing also has the potential to increase resuspension of bed sediments and associated pollutants in shallow waters. The wind-mixed depth, referred to as the “critical depth,” is directly related to the fetch (the distance wind travels across the surface of the lake), the lake depth, and the wind speed. Longer lake fetches tend to allow for a greater critical depth, and lakes unprotected from the wind are more susceptible to increased wind mixing. In most shallow lakes, the critical depth is approximately equal to the average depth of the lake; this allows for areas prone to resuspension. The degree to which wind mixing impacts pollutant resuspension is also related to the lake’s water-level, as there is considerably less potential for sediment resuspension under deep waters; sediments underlying shallow waters have an increased potential for resuspension due to wind action.

The degree to which wind mixing and lake turnover impact water column pollutant concentrations also depends on the physical characteristics of sediment present at the bottom of the lake, the presence or absence of a lake liner, and the presence or absence of benthic algae and macrophytic (rooted plant) communities. Locations with loose organic sediment and sparse plant coverage are more prone to increased rates of resuspension due to wind mixing. Lakes with coarse sediments (sands and gravels), low amounts of settled organic material, or those with artificial liners have less potential for resuspension. Refer to the lake-specific TMDL sections (Sections 4 through 13) for information regarding soil types, lake liners, and bathymetric data.

As described, sediment resuspension has been predicted in studies drawing relationships between resuspension and wind speed, wind direction, fetch and depth to sediment (Carper and Bachmann, 1984). As wind blows over the surface, a deep water wave will be generated when the depth of the water is greater than one half of the wave-length (Wetzel, 2001). The transition of the wave from the deep water to shallow water creates a situation prone to resuspension. The wavelength (L) of a deepwater wave is related to its period (T), in the following relationship, where g is the gravitational constant (Martin and McCutcheon, 1999):

$$L = \frac{gT^2}{2\pi}$$

The period of a wave can be estimated by using the equation derived by the US Army Coastal Engineering Research Center (Carper and Bachmann, 1984). Where U is the wind speed and F is the fetch:

$$T = \frac{2.4\pi U \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right]}{g}$$

Although the pollutant loads, due to lake turnover and wind mixing, are not explicitly quantified for these TMDLs, they are included inherently in the eutrophication, mercury bioaccumulation, and OC Pesticides and PCBs models developed for each lake (see Appendices A, C, and H, respectively).

B.4 Internal Loading and Anoxic Conditions

The dissolved oxygen concentration at the sediment-water interface plays an important role in the internal loading of various ionic compounds. The condition in which oxygen is fully depleted is called anoxia; partial depletion (below 2 mg/L) is referred to as hypoxia. Deeper lakes will often become thermally stratified in summer months, resulting in anoxic or hypoxic conditions within the lower metalimnion (the middle layer of a stratified lake) and hypolimnion (the bottom layer of a stratified lake). To a certain degree, this is a natural process within deeper lakes; however, it is more common for lakes with small surface areas to become anoxic due to stagnation or limited water exchange. Additionally, the decomposition of the phytoplankton associated with eutrophication requires oxygen, thus decreasing the available dissolved oxygen within the water column, particularly near the sediment-water interface where decaying organic matter tends to settle and accumulate.

The oxidation-reduction (redox) potential of an aquatic system is used to describe the process or degree to which ions are exchanged within a system. Compounds gaining electrons are said to be reduced, while those losing electrons are oxidized. Important biological processes used to create energy (i.e., respiration and photosynthesis) involve the exchange of electrons. The most energetically favorable reaction occurs with the oxidation of organic material (oxic respiration). However, in the absence of oxygen, bacterial processes shift to denitrification, manganese reduction, iron reduction, sulfate reduction, and methanogenesis (releasing compounds such as NH_3/NH_4^+, Mn^{2+}, Fe^{2+}, S^{2-}).

In oxygenated environments, free electrons are readily bound by oxygen and associated compounds are partitioned to sediments. In anoxic environments, particularly at the sediment-water interface or the oxic-anoxic boundary within the water column, electrons and compounds are released into the water column via redox reactions. This release can dramatically increase the concentration of reduced species (NH_3/NH_4^+, Mn^{2+}, Fe^{2+}, S^{2-}) within the waterbody. Artificial aeration of bottom waters impedes these reactions and the release of pollutants.

For example, with limited oxygen, bacterial decomposition of organic material in lake sediments results in the release of inorganic phosphorus into the water column. The iron cycle has a dramatic effect on the rates of recycling of phosphorus: under oxidizing conditions, iron and phosphorus form insoluble ferric hydroxy complexes; under reducing conditions these complexes dissolve, releasing both iron and phosphorus to the water column. In fact, one study found that sediment phosphorus flux was fourfold greater under anoxic conditions (Haggard, 2005) than aerobic. Increased levels of phosphorus resulting from sediment release add to the available nutrient pool and continue the cycle of eutrophication.

Denitrification also occurs under anoxic conditions where nitrates are first reduced to ammonia (NH_4/NH_3) and then to nitrogen gas (N_2). Conversion to ammonia may occur in environments with low oxygen levels; reduction to nitrogen gas requires anoxic conditions.

Under anoxic conditions, sulfates (SO_4^+) are reduced to bisulfide or sulfide (HS^- or S^{2-}). The presence of sulfidic compounds produces a strong sulfur odor, which can lead to an odor impairment in a waterbody. Methylation of mercury is an additional microbial process that occurs under low-oxygen, or reducing conditions. Research shows that sulfur-reducing bacteria may play an important role in the methylation process (Compeau, 1985). The transformation of mercury into methylmercury is of concern as the methylated form, methylmercury, bioaccumulates within the food chain and may accumulate to levels that are unsafe for human or wildlife consumption. A more detailed description of mercury methylation is presented in Appendix C (Mercury TMDL Development).

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B.5 Bioturbation

Bioturbation is the mixing and resuspension of sediment and benthic material by fish and macroinvertebrates. This disturbance of the sediments can have an impact on nutrient cycling and the availability of sediment-associated pollutants. In particular, bioturbation by bottom feeding fish can stir up the sediment and increase the movement of nitrogen and phosphorus into the water. Organic contaminants are typically hydrophobic and prefer to sorb to organic matter that may have settled to the lake bottom where bioturbation may cause resuspension and loading to the water column. Of great concern is the release of historical stores of OC Pesticides and PCBs.

For example, a positive relationship was observed between carp biomass and total suspended sediments within the water column and, more specifically, bream (a benthivorous fish) was shown to cause a 0.03 mg/L increase in total phosphorus per 100 kg of bream per hectare (Breukelaar, 1994). An additional study by Persson and Svensson (2006) showed increased concentrations of nitrogen and phosphorus in the water column of enclosures with benthivorus fish relative to controls with no fish.

Fish are stocked at the El Dorado Park lakes, Santa Fe Dam Park Lake, Echo Park Lake, North, Center and Legg lakes, Puddingstone Reservoir, Lincoln Lake, Peck Road Park Lake, and Westlake (California Department of Fish and Game, 2009). Fish have also been observed in Lake Calabasas and Sherwood Lake during recent monitoring events. Despite the confirmed presence of fish, available data do not include a comprehensive fish population assessment.

Another type of bioturbation is caused by macroinvertebrates that feed in the sediment. This first causes vertical mixing in the sediment. Some macroinvertebrates – particularly tubificid oligochaete worms – maintain burrows that enable them to feed at depth but defecate on the surface of the sediment. Such worms, which often occur at very high densities in organic sediments, can effectively pump significant amounts of both sediment-sorbed and porewater dissolved pollutants from depths of up to 10 inches or more into the water column (e.g., Reible et al., 1996).

Without comprehensive population assessments (species and population size), it is difficult to quantify the amount of pollutants in the water column that are directly related to bioturbation. Although bioturbation may not be precisely calculated without complete population assessments, it is assumed that samples collected at locations containing fish include water column concentrations impacted by bioturbation. In addition, impacts of bioturbation are included inherently in the eutrophication, mercury bioaccumulation, and OC Pesticides and PCBs models developed for each lake (see Appendices A, C, and H, respectively).

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B.6 Impacts of Sedimentation

Under certain conditions, lake sediments behave as significant sinks, removing pollutants from contact with the water column by allowing for deep burial and sequestration. In general, deep burial depends on the net sedimentation rate, which is the external sediment supply less resuspension. Rates of burial loss of specific compounds depend on the extent to which the compound is adsorbed to sediment, and lake dynamics (stratification, internal concentration, wind mixing, and depth) that determine rates of recycling of deposited material. Burial rates are often high for lakes in arid climates due to the sparse vegetative ground cover compared to areas receiving higher amounts of rainfall.

B.6.1 PHOSPHORUS

Inorganic phosphorus is particle-reactive. The burial and sequestration of phosphorus is an important mechanism that can reduce the mass of bioavailable phosphorus within the water column. Sedimentation rates depend on the specific lake dynamics as well as the size and settling velocity of the particulate matter to which the phosphorus is bound (Welch and Jacoby, 2004).

As explained in Sections B.4 through B.5, sediment stores of phosphorus can be released into the water column through multiple mechanisms. Thus, sedimentation may act as a sink under certain conditions and as a source under other conditions. First, anoxic environments, often present at the sediment-water interface, increase the reduction and release of phosphorus (Section B.4). Second, resuspension of sediment by wind mixing (Section B.3), and bioturbation (Section B.5) can result in additional recycling from the sediment to the water column.

B.6.2 MERCURY

In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of methylmercury (MeHg) in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of MeHg from lake sediments. For instance, in McPhee Reservoir in Colorado (Tetra Tech, 2001), 71 percent of the MeHg present in the water column was estimated to derive from watershed inflows, while much of the MeHg created in lake sediment was apparently buried. Lakes with high sedimentation rates are therefore likely to respond approximately linearly to reductions in the watershed MeHg and total Hg load – although there may well be a delay in the response to load reductions, as found for McPhee Reservoir (Tetra Tech, 2001).

B.6.3 ORGANOCHLORINE PESTICIDES AND PCBS

Many OC Pesticides and PCBs have a high propensity to partition to sediment. For example, chlordane, DDT, dieldrin, and PCBs are hydrophobic and have low water solubilities. These characteristics increase the partitioning and, therefore, these OC Pesticides and PCBs are more likely to bind to sediment. The majority of the pollutant loads for such compounds will be stored in the lake sediments and further concentrated in aquatic organisms through bioaccumulation in the food chain. It is important to note that despite sediment contamination, water column concentrations of OC Pesticides and PCBs are frequently below detectable concentrations.

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Appendix C.   Methodology for Mercury TMDL Development

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C.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure C-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, dieldrin, DDT, PCBs, and trash.

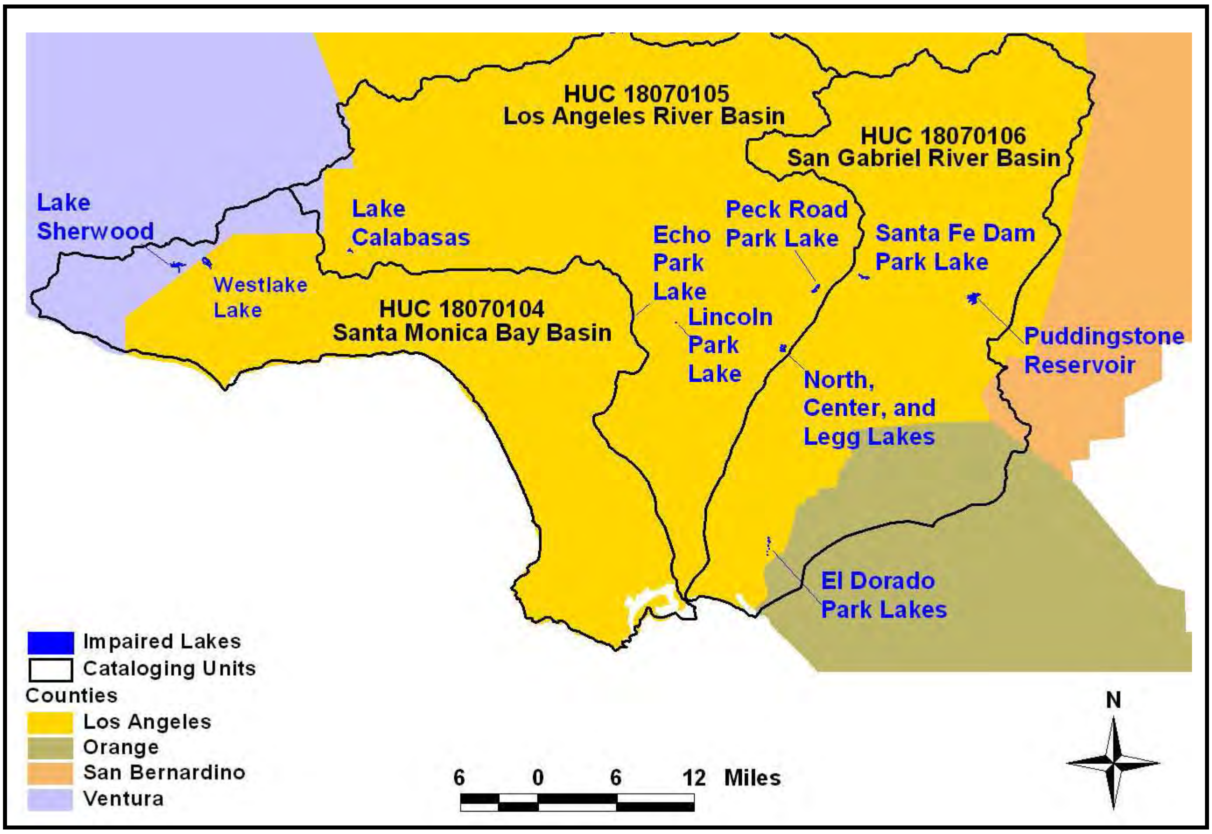

Figure C-1. Location of Impaired Lakes

Three of these waterbodies are listed as impaired by mercury due to elevated fish tissue concentrations: the El Dorado Park lakes, Puddingstone Reservoir, and Lake Sherwood. This appendix discusses the lake specific load allocations based on the measured tissue concentrations observed in each waterbody.

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C.2 Description of the Mercury Cycle

Selected aspects of the lake and watershed mercury cycle are summarized schematically in Figure C-2, based on the representations discussed in Hudson et al. (1994) and Tetra Tech (1999). The boxes represent stores of mercury, and the arrows represent fluxes. The top of the diagram summarizes the various forms of mercury that may be loaded to a lake.

It is important to recognize that mercury exists in a variety of forms, including elemental mercury (Hg(0)), ionic mercury (Hg(I) and Hg(II)), and compounds in which mercury is joined to an organic molecule.

In the figure, Hg(I) is ignored because Hg(II) species generally predominate in aquatic systems. Mercuric sulfide (HgS or cinnabar) is a compound formed from Hg(II) but is shown separately because it is the predominant natural ore. Organic forms of mercury include methylmercury (CH_3Hg or “MeHg”), and other natural forms such as dimethylmercury and manmade compounds such as organic mercury pesticides. (Where sorption and desorption are indicated in Figure C-2, “Hg(II)” and “MeHg” refer to the same common pools of water column Hg(II) and MeHg shown in the compartments at the top of the diagram.)

Dimethylmercury (CH_3-Hg-CH_3) is also ignored in the conceptual model shown in Figure C-2, because this mercury species seems to occur in measurable quantities only in marine waters. Organic mercury pesticides also have been ignored in this TMDL study, because such pesticides are not currently used in this country. Loads delivered to the lake historically have likely been buried under years of accumulated sediment. If contaminated upland sediments continue to contribute loading to the impaired waterbodies, recent tributary monitoring data will include this component of loading.

Ionic mercury and methylmercury form strong complexes with organic substances (including humic acids) and strongly sorb onto soils and sediments. Once sorbed to organic matter, mercury can be ingested by invertebrates, thus entering the food chain. Some of the sorbed mercury will settle to the lake bottom; if buried deeply enough, mercury in bottom sediments will become unavailable to the lake mercury cycle. Burial in bottom sediments can be an important route of removal of mercury from the aquatic environment.

Methylation and demethylation play an important role in determining how mercury will accumulate through the food web. Hg(II) is methylated by a biological process that appears to involve sulfate-reducing bacteria. Rates of biological methylation of mercury can be affected by a number of factors. Methylation can occur in water, sediment, and soil solutions under anaerobic conditions, and to a lesser extent under aerobic conditions. In lakes, methylation occurs mainly at the sediment-water interface and at the oxic-anoxic boundary within the water column. The rate of methylation is affected by the concentration of available Hg(II) (which can be affected by the concentration of certain ions and ligands), the microbial concentration, pH, temperature, redox potential, and the presence of other chemical processes. Methylation rates appear to increase at lower pH. Demethylation of mercury is also mediated by bacteria.

Both Hg(II) and methylmercury (MeHg) sorb to algae and detritus, but only the methylmercury is readily passed up to the next trophic level (inorganic mercury is relatively easily egested). Invertebrates eat both algae and detritus, thereby accumulating any MeHg that has sorbed to these. Fish eat the invertebrates and either grow into larger fish (which continue to accumulate body burdens of mercury), are eaten by larger fish or other piscivores, or die and decay.

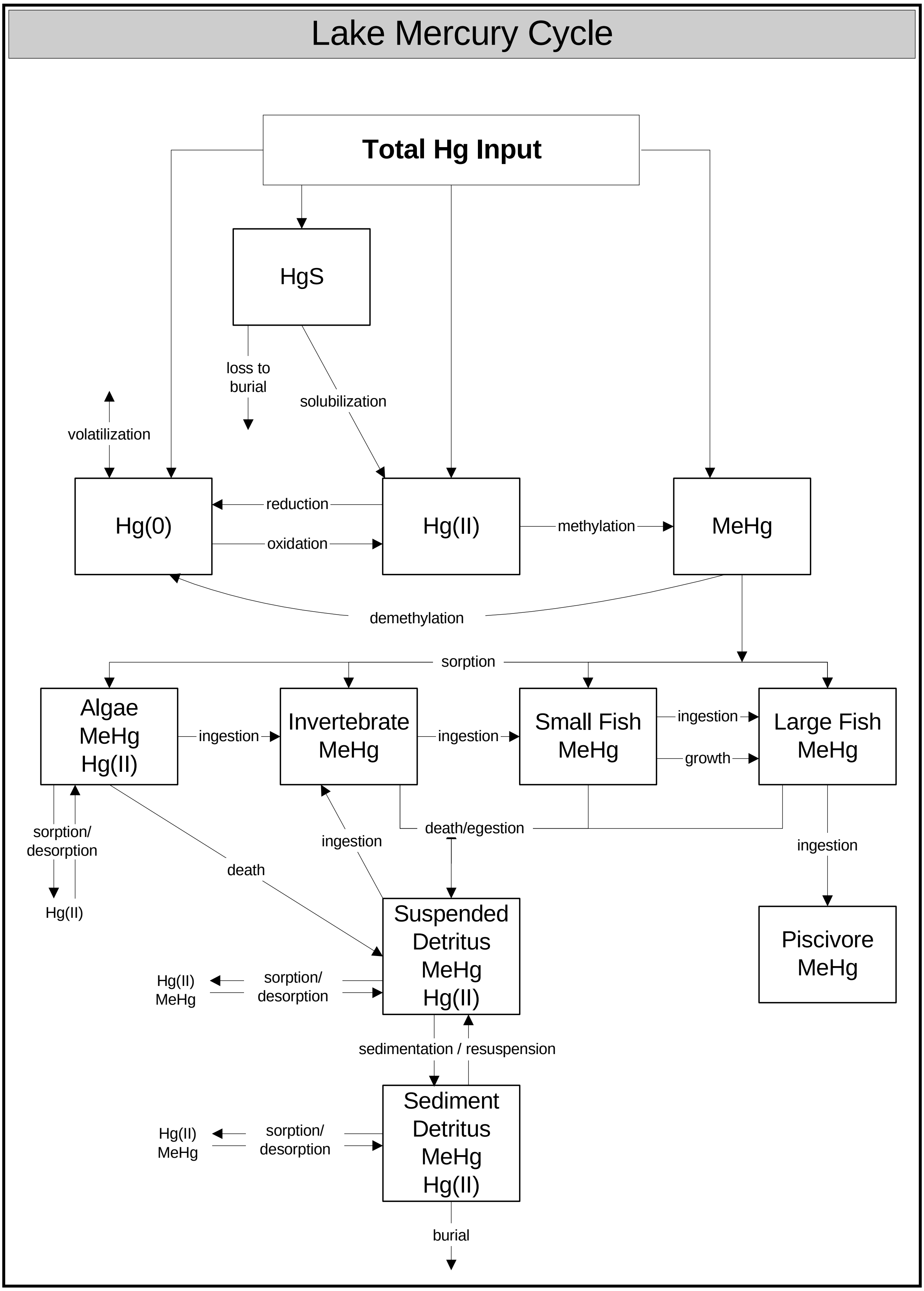

Figure C-2. Conceptual Diagram of Lake Mercury Cycle

Typically, almost all of the mercury found in fish (greater than 95 percent) is in methylmercury form. Studies have shown that fish body burdens of mercury tend to increase concurrently with increasing size or age of the fish, under conditions of constant exposure.

Although it is important to identify external sources of mercury to the reservoir, there may be fluxes of mercury within the reservoir that would continue for some time even if all external sources of mercury load were eliminated. The most important store of mercury within the reservoir is the bed sediment. Mercury in the bed sediment may cause exposure to biota by being:

- Resuspended into the water column, where it is ingested or it adsorbs to organisms that are later ingested.
- Methylated by bacteria. The methylmercury tends to attach to organic matter, which may be ingested by invertebrates and thereby introduced to the lake food web.

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C.3 Source Assessment

Sources of mercury loading to a lake may include both point and nonpoint sources. For purposes of allocations among mercury sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which waste load allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). The most significant source of mercury in point source discharges is wastewater associated with the installation or removal of mercury amalgam dental fillings. Sources in the watershed include junkyards housing automobiles where mercury-containing switches have not been removed prior to crushing, and landfills where fluorescent light bulbs have not been properly disposed. Significant releases to the atmosphere may occur from coal-power plants, cement manufacturing facilities, oil refineries, and chlor-alkali plants. This section describes how loading from point and nonpoint sources were estimated for the mercury-impaired watersheds.

C.3.1 POINT SOURCES

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process.

C.3.1.1 MS4 Permittees

In 1990 USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed by stormwater runoff into the Municipal Separate Storm Sewer Systems (MS4), or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

Mercury loads from urban stormwater runoff and associated sediment are estimated from monitoring data collected at the mouth of each major tributary that discharges to a mercury impaired lake (Appendix G, Monitoring Data) and simulated flows and sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated watershed models (using the Loading Simulation Program in C++ [LSPC]) models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate stormwater runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to a MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output, summarizes the mercury monitoring data, and presents the resulting mercury loading from MS4 systems.

These systems may also discharge during dry weather as a result of irrigation, car washing, etc. Estimation of mercury loading from MS4 systems in dry weather is based on SCCWRP regional flow estimates and local monitoring data as described in Appendix F (Dry Weather Loading).

C.3.1.2 Non-MS4 NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). . To quantify mercury loading from non-MS4 NPDES discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the San Gabriel River and Santa Monica Bay basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the three mercury impaired lakes. Mercury loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake specific sections of this report.

C.3.1.3 Additional Inputs

Several of the lakes addressed by this TMDL have additional point source inputs that do not currently have NPDES permits. Most are supplemental flows from groundwater wells, or potable water that maintain lake levels. Information pertaining to flow volumes from these sources was provided by park staff at each lake (generally based on water usage information from the water suppliers). Where accessible, the Regional Board and USEPA sampled water quality from these inputs during the 2009 sampling events. Mercury loading was calculated from observed concentration data and an estimate of annual flow volumes to each lake.

C.3.2 NONPOINT SOURCES

Mercury loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

C.3.2.1 Atmospheric Deposition

Mercury deposition from the atmosphere to the earth's surface may occur in several forms: gaseous elemental mercury (Hg(0)), divalent ionic mercury (Hg(II)), reactive gaseous mercury (RGM), and aerosol particulate mercury (Hg-P). Atmospheric deposition can be divided into short-range or near-field deposition, which includes deposition from sources located near the watershed, and long-range or far-field deposition, which includes mercury deposition from regional and global sources. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport.

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). There is one MDN site in Southern California, but it has only been active since May of 2006. The rates of wet mercury deposition to each lake water surface were estimated with a regression approach that utilized nitrate and sulfate wet deposition data collected by the National Atmospheric Deposition Program (NADP), along with mercury wet deposition data collected by the MDN (see Appendix E, Atmospheric Deposition).

Dry deposition is more difficult to monitor and less localized data are available to estimate this component. To estimate loading from this component, grid-cell output from regional deposition models developed by USEPA were obtained for each lake impaired by mercury (see Appendix E, Atmospheric Deposition).

To evaluate potential nearfield sources at each impaired lake, the USEPA Toxics Release Inventory (TRI) was used to determine the proximity of point sources that may contribute to airborne mercury loads including coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories.

Precipitation events following recent forest fires also result in increased loads of total and methylmercury from the watershed and release of elemental mercury to the atmosphere which is then available for deposition.

C.3.2.2 Watershed Loading

Mercury loads from areas that do not drain to an MS4 system are estimated from monitoring data collected at the mouth of each major tributary that discharges to a mercury impaired lake (Appendix G, Monitoring Data) and simulated flows and sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output, summarizes the mercury monitoring data, and presents the resulting wet weather mercury loading areas that do not discharge to an MS4.

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. Nonpoint sources during dry weather include groundwater discharges, irrigation (reclaimed water is used for irrigation of parklands adjacent to two of the waterbodies), fertilization of adjacent parkland, and other miscellaneous urban sources. Estimation of dry weather pollutant loading is discussed in Appendix F (Dry Weather Loading).

C.3.2.3 Methylation

Accumulation of mercury in biota is determined by methylmercury concentrations, not total mercury. These concentrations reflect both methylation within the lake and external loading of methylmercury. Methylation of mercury occurs under oxygen-poor, reducing conditions. Wetland areas are particularly likely sites for methylation in the watershed. Other likely sites include shallow riparian groundwater, the bottom waters and sediment of impoundments that stratify and go anoxic, and beaver ponds and their associated wetlands. Sampling for methylmercury concentrations in the water column and sediment was performed at each tributary or input to the impaired lakes. One of the tributaries at Lake Sherwood exhibited characteristics associated with high methylation (see Section 12). The implementation section for each lake will address how to best manage these loads.

Dredging activities to remove accumulated sediment from lakes and sedimentation basins may have significant impacts on total and methylmercury loading to lake waters. In theory, removal of accumulated sediment should reduce the amount of total and methylated mercury stored in the sediments. Unfortunately, the removal process may disturb and release methylated mercury into the water column and increase the bioavailability of the metal. Additionally, removal of the top layers of sediment may uncover layers deposited during the 1960s through 1980s when air emissions of mercury were less adequately controlled. Proper testing and planning is required to ensure that removal activities do not add to the overall mercury burden.

C.3.2.4 Direct Geologic Sources

Geological formations containing significant mercury concentrations have a higher probability of occurrence in mineralized areas along fault lines, intrusive dikes in igneous formations, or resulting from natural springs. Volcanic activity has the potential to release mercury into the air, so areas with large ash deposits may contain higher concentrations of mercury. Mercury is also more likely to occur in shale and slate deposits as they are derived from clays, which have high affinities for adsorbing metals such as mercury (this affinity explains why coal burning power plants emit mercury). Sediment mercury concentrations measured at the mouth of each major tributary include the geologic component as well as anthropogenic sources of mercury.

The California Geological Survey has posted a map online of the earthquake hazard across the state (http://www.consrv.ca.gov/cgs/rghm/psha/Pages/index.aspx). This map indicates that fault line activity in these three watersheds is moderate.

The U.S. Geological Survey conducted a geochemical survey of stream sediments and generated estimates of mercury concentrations in soil by county in their Open-File Report 2004-1001 accessible via their website (http://tin.er.usgs.gov/geochem/doc/home.htm). The mean concentration estimated for Ventura County (where Lake Sherwood is located) is 0.064 ppm with a standard deviation of 0.034 ppm (minimum of 0.022 and maximum of 0.232 ppm). The mean concentration estimated for the County of Los Angeles (where El Dorado Park lakes and Puddingstone Reservoir are located) is 0.149 ppm with a standard deviation of 0.217 ppm (minimum of 0.010 and maximum of 1.849 ppm). The nearest sample to Lake Sherwood was NURE record ID RA000197, which had a mercury concentration of 0.02 ppm whereas the nearest sample to El Dorado Park lakes was NURE record ID RA000163 with a mercury concentration of 0.07 ppm. At Puddingstone Reservoir the nearest sampling location, which was analyzed for mercury was NURE record ID RA000425, had a mercury concentration of 0.08 ppm.

C.3.2.5 Indirect Geologic Sources

Geological formations containing deposits of precious metals (e.g., gold, silver, and copper) have been targets of historic and current mining activities. In cases where the desired metals are contained in ore as opposed to veins, extraction of the desired metal commonly occurs through the process of amalgamation, in which mercury is used as the amalgam. Amalgamation is an easy and inexpensive process of removing fine metal particles from ore, but when poorly implemented, it can lead to spillage of mercury, contaminated mine tailings, and localized atmospheric deposition.

Oil production may also release mercury into the environment, particularly in California. Mercury often causes corrosion and fouling problems in pipelines and equipment and is easily transferred from oil to water during refinement processes. Researchers at the University of North Dakota found that typical mercury concentrations in crude oil across the globe are less than 20 ppb, but that some measurements taken in California have been as high as 24,000 ppb (http://www.undeerc.org/catm/pdf/area3/MJH_Crude_2002.pdf). Wilhelm et al. (2004) also report that average concentrations measured from crude oil samples in California were higher than those measured from other states in the US (11.3 ppb compared to 4.3 ppb).

Thus, in relation to mining potential and oil production, the geological formations in a watershed can indirectly influence mercury loadings. No precious metal mines or oil refineries are known to have operated within the watersheds of the three mercury impaired lakes. However, the presence of oil refineries in the general region indicates that high mercury sediment concentrations may exist.

C.4 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of each impaired lake. This in turn allows estimation of the Total Maximum Daily Load (TMDL), and allocation of that load to urban sources (wasteload allocations) and rural sources (load allocations). The TMDL also contains a Margin of Safety, which is described in detail below.

Neither data nor resources are available to create and calibrate detailed lake response models for mercury cycling in the El Dorado Park lakes, Puddingstone Reservoir, and Lake Sherwood. The key to the TMDL target is achieving acceptable concentrations in fish. The mercury TMDLs for these three lakes are being developed in a similar fashion to the Big Bear Lake TMDL, which applies watershed specific mercury concentrations to simulated sediment loads and water volumes (Tetra Tech, 2008; note: as of the writing of these TMDLs, the Big Bear Lake TMDL for mercury has not been finalized and approved). For these three lakes, previously developed LSPC models provide a mechanism for incorporating wet, normal, and dry simulation years into the TMDL.

In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of MeHg in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of MeHg from lake sediments. For instance, in McPhee Reservoir in Colorado (Tetra Tech, 2001), 71 percent of the MeHg present in the water column was estimated to derive from watershed inflows, while much of the MeHg created in lake sediment was apparently buried. Lakes with high sedimentation rates are therefore likely to respond approximately linearly to reductions in the watershed MeHg and total Hg load – although there may well be a delay in the response to load reductions, as found for McPhee Reservoir (Tetra Tech, 2001).

Lakes in arid climates are predisposed to high rates of sedimentation given the lower density of vegetative ground cover compared to areas receiving higher amounts of rainfall. Each of the three mercury impaired systems addressed by this TMDL likely experience average to high rates of sedimentation. In fact, Lake Sherwood is listed as impaired by sedimentation, as was Big Bear Lake (Tetra Tech, 2008). Two studies have summarized sedimentation rates for Puddingstone Reservoir. According to the Reservoir Sedimentation Database (accessed 6/5/2009), the average annual historical sedimentation rate measured from 1915 to 1941 for Puddingstone Reservoir was 16 ac-ft per year (approximately 0.76 inches per year). The Department of Boating and Waterways and State Coastal Conservancy (2002) reports that the average annual sedimentation rate measured in Puddingstone Reservoir from 1925 to 1980 was 31 ac-ft per year (approximately 1.5 inches per year). For Lake Sherwood, the reported average annual sedimentation rate measured from 1905 to 1938 ranged from 2.5 to 10 acre-feet per year (0.22 to 0.88 inches per year); this rate has likely increased with development around the perimeter of the lake. Site specific data for the El Dorado Park lakes are not available. However, watershed loading at El Dorado Park is less significant than loads associated with the groundwater source used for lake filling (see Appendix F, Dry Weather Loading).

The available evidence suggests that sedimentation rates are likely to diminish the relative importance of MeHg recycling from lake sediments if coupled with reductions in mercury. This, in turn, suggests that MeHg exposure concentrations in each lake should respond approximately linearly to reductions in mercury load, particularly if conditions favoring methylation are discouraged (i.e., anoxic conditions near the sediment-water interface). While this is the best assumption that can be made with the current data, two caveats should be mentioned. First, the burial and sequestration of MeHg due to sedimentation may be counteracted by dredging activities that may occur periodically as part of an overall lake management plan. Second, the potential role of peripheral wetlands or forebays as a locus of mercury methylation and subsequent loading to each lake is currently unknown. It is clear that reductions in external mercury loads to each waterbody will be beneficial, although a program of adaptive implementation may need to be pursued if elevated fish tissue concentrations persist.

Nationally, authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between MeHg in water and MeHg in fish tissue normalized to length. However, this relationship is well-approximated by a linear relationship for the ranges of fish tissue concentration of concern for these impaired lakes. Until such time as lake response models for mercury are constructed for these waterbodies, and sufficient calibration data collected to develop them, an assumption of an approximately linear response of fish tissue concentrations to changes in external loads is sufficient for the development of these TMDLs.

Each of the three lakes shows exceedances of the fish tissue mercury concentration in largemouth bass. Exceedances of the total mercury water quality standard were not observed in any of the impaired waterbodies; however, two lakes had exceedances of the dissolved methylmercury water quality standard (Lake Sherwood and Puddingstone Reservoir; Note: the observed data were based on the total fraction, while the water column target is for the dissolved fraction, resulting in more conservative assessments). Because limited samples were available to compare to the dissolved methylmercury target and the long-term average fish tissue concentrations are more predictive of exposure pathways for humans and wildlife, the TMDLs were based on the reduction required to meet the fish tissue guideline. In addition, the mercury reductions required by the fish tissue data were consistently higher than the reductions required to meet the methylmercury water column target; therefore, meeting the reductions for fish tissue should also result in attainment of the water column target for methylmercury.

C.5 TMDL Development

The TMDL is defined by the loading capacity. A waterbody's loading capacity represents the maximum amount pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). For mercury, this is the maximum amount of mercury loading and methylation uptake consistent with meeting the numeric target of 0.22 ppm for mercury in 350mm largemouth bass.

C.5.1 Loading Capacity and Allocations

A model of lake response and fish bioaccumulation has not been created at this time for these impaired lakes. Rather, it is assumed that, in the long term, fish tissue concentrations will respond approximately linearly to reductions in mercury loads. This assumption has been found to be a reasonable first-order approximation in other systems with high burial rates, such as McPhee and Narraguinnep reservoirs in Colorado (Tetra Tech, 2001). For McPhee in particular, a detailed model of lake mercury cycling and bioaccumulation was created using the D-MCM model (Tetra Tech, 1999). The calibrated model yielded predictions that were well-approximated by the assumption of a linear response of fish tissue concentration to reductions in external mercury loads.

Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass, the predominant trophic level 4 fish in each waterbody. To do this, a linear regression analysis was performed on tissue concentrations versus length for each lake, which was then used to predict the existing concentration associated with the target size fish (see Appendix G [Monitoring Data] for details regarding fish tissue monitoring data). The resulting linear regression equations are presented as

$$Hg(fish) = Y\text{-}intercept + Slope \cdot Len$$

where *Hg(fish)* is the total mercury concentration in largemouth bass (ppm), *Len* is length in mm, and *Y-intercept* and *Slope* are constants representing the point at which the line crosses the y-axis and the slope or gradient of the line, respectively. In addition, the one-sided 95 percent upper confidence limits on mean predictions about the regression line (95 percent UCL) and the 95 percent upper prediction intervals on individual predicted concentrations (95 percent UPI) were calculated. The UPI gives the confidence limit on the individual predictions for a given length while the UCL gives the confidence limit on the average of the predictions for a given length. These regressions have non-zero intercepts and should not be considered valid for lengths less than the representative dataset (150 to 200 mm depending on the lake).

For mercury, long-term cumulative exposure is the primary concern. Therefore, it is appropriate to use the 95 percent UCL rather than the UPI to provide a Margin of Safety on the appropriate age class. Use of the UCL provides a Margin of Safety because it represents an upper confidence bound on the long-term exposure concentration.

The one-sided 95 percent UCL is given by

$$UCL_{0.95} = \mu_{y|x_0} + t_{0.05,n-2} \cdot s_{\mu_y|x_0}$$

where $\mu_{y|x}$ is the predicted value of *y* given $x{=}x_0$, *t* is the Student's *t*-statistic with *n*-1 degrees of freedom, and *n* is the number of observations used in the regression. The variance on the prediction at $x{=}x_0$, $s^2_{\mu_y|x_0}$, is given by

$$s^2_{\mu_y|x_0} = s^2_{y|x} \cdot \left[\frac{1}{n} + \frac{(x_0 - \bar{x})^2}{\sum (x_i - \bar{x})^2} \right],$$

where x_0 is the value of the independent variable (*Len*) at which the prediction is made, $\bar{x}$ is the mean of the observed independent variables, x_i, and $s^2_{y|x}$ is the standard error of the model estimates. For example, for the El Dorado Park lakes data (see Section 8), this yields:

$$UCL_{0.95}[Hg(fish)] = -0.15316 + 0.001461 \cdot Len + 2.0796 \cdot 0.14285 \cdot \sqrt{\frac{1}{23} + \frac{(Len - 365.9)^2}{106245}}$$

This equation expresses the upper 95 percent confidence limit on predicted fish tissue mercury concentrations for any length (*Len*). The first two terms alone would generate the prediction line; the addition of the last term results in the UCL line.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in each lake. The target length for assessing compliance with this tissue concentration is 325-375 mm for largemouth bass. A range is provided for compliance; however, an average of 350 mm largemouth bass is used for TMDL calculations. The predicted mercury concentration based on a one-sided 95 percent upper confidence limit on mean predictions about the regression line (95 percent UCL) for this length is compared to the target fish concentration to determine the required reduction in mercury loading, which includes a Margin of Safety as described above.

For each lake, the fraction of existing load consistent with attaining the target (the loading capacity) is the ratio of the target (0.22 ppm) to the best estimate of current average concentrations in the target fish population. The difference between the direct regression estimate and the 95 percent UCL provides the Margin of Safety. Therefore, the allocatable fraction of the existing load (the loading capacity less the Margin of Safety) is the ratio of the target to the 95 percent UCL. The resulting loading capacities and allocatable loads are expressed as fractions of the existing load in the lake-specific chapters. For example, at Lake Sherwood the predicted total mercury concentration for a 350 mm largemouth bass is 0.607 ppm, and the 95th percent UCL is 0.744 ppm. The following calculations apply:

Loading capacity as fraction of existing load = 0.22 ppm / 0.607 ppm = 0.362

Allocatable load as fraction of existing load = 0.22 ppm / 0.744 ppm = 0.296

Margin of safety as fraction of the existing load = 0.362 – 0.296 = 0.067

The loading capacity can also be expressed as grams per year (g/yr) using the existing load from the source assessments and the calculated fractions of the existing load. Estimates of the existing mercury load to each lake are discussed in Appendices D, E, and F. Specifically, the loading capacity is presented as a percentage of the existing load (in grams per year). This value can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

For division of WLAs and LAs, the percent reduction in mercury loading was applied equally to all sources of mercury in each watershed based on the results of the lake-specific source assessments.

C.5.2 MARGIN OF SAFETY

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. An implicit MOS is included based on comparison of total mercury concentrations in fish tissue to the methylmercury guideline (most, but not all, of the total mercury in fish is in the methyl form) (Note: additional lake-specific conditions or assumptions may also have been included in the implicit MOS). An explicit MOS is provided by the use of the 95 percent UCL to determine the allocatable load.

C.5.3 DAILY LOAD EXPRESSION

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. Although it is long-term cumulative load rather than daily loads of mercury that are driving the bioaccumulation of mercury in fish in, these TMDLs do present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of loads occur during wet weather events, the daily maximum allowable load of mercury is calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) multiplied by the allowable concentration for mercury consistent with achieving the long-term loading target. For lakes with significant loading from other sources, such as supplemental water additions, appropriate daily flow rates were identified and multiplied by the allowable concentration for mercury to determine the daily maximum allowable load.

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Appendix D. Estimation of Wet Weather Loading from Runoff and Sediment Transport

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D.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure D-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). The waterbodies are impaired by low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

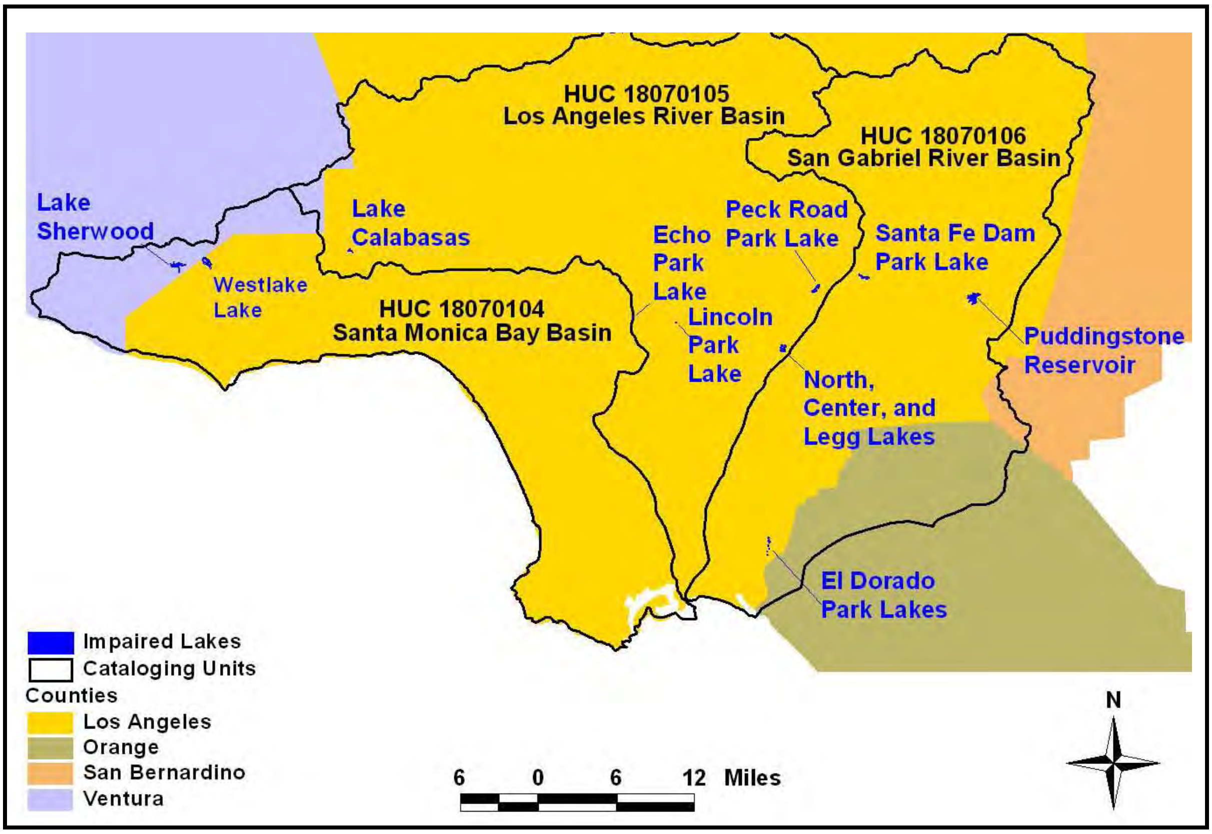

Figure D-1. Location of Impaired Lakes

Estimation of watershed loading as a result of wet weather events is based on calibrated watershed models developed for the Los Angeles and San Gabriel river basins. Each model was previously calibrated for flow and metals loading. For the purposes of developing nutrient and mercury TMDLs, the simulated flows predicted for land uses in spatially relevant modeling subbasins were used along with regional event mean concentrations (EMCs) of total suspended sediment, nitrogen, and phosphorus to estimate loading to each impaired waterbody. Mercury concentrations were based on monitoring data collected at the mouth of each major storm drain or tributary to the mercury-impaired lakes.

Each of the impaired lakes, with the exception of Lake Sherwood, is in either the Los Angeles or San Gabriel River Basin. Lake Sherwood, however, is in close proximity to the Los Angeles River Basin, and the land use coverage compiled for this model covers the Lake Sherwood drainage area.

Each of the river basin watershed models (using the Loading Simulation Program in C++ [LSPC] model) was calibrated for flow. Model output available for the years 1983 to 2006 was used to estimate average monthly runoff depths by land use for each LSPC modeling subbasin that contains one of the impaired lakes addressed by this TMDL document (Note: all references to runoff in this appendix are associated with both the storm drain system and nonpoint sources). These years represent dry, normal, and wet conditions for the Los Angeles area and provide a reasonable estimate of average runoff conditions for these waterbodies.

The TMDLs are allocated based on subwatershed and MS4 stormwater permittee. A GIS environment was used to overlay the subwatersheds, jurisdictions, and the LSPC land use coverages to estimate the area of each modeled land use within a subwatershed/jurisdiction area. Monthly runoff volumes were then calculated for each combination of land use/subwatershed/jurisdiction based on land use area and simulated runoff depth.

To estimate loading of nutrients, metals, and Organochlorine (OC) Pesticides and PCBs to each waterbody from upland areas, event mean concentrations (EMCs) based on the Southern California Coastal Water Research Project (SCCWRP) and the county of Los Angeles monitoring studies were applied to the average monthly runoff volumes calculated for each land use/subwatershed/jurisdiction area (i.e., water quality EMCs and runoff volume were used to calculate loadings for nutrients, metals, and OC Pesticides and PCBs). Mercury loads are estimated from simulated runoff volumes, predicted sediment loads (based on EMCs), and watershed monitoring data. Specifically, mercury loading is associated with both sediment and runoff from upland areas. To determine sediment loading of mercury, the sediment EMCs and runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data were then applied to these sediment loads. Similar to the nutrients, metals, and OC Pesticides and PCBs, mercury loading associated with runoff from upland areas was calculated using the water quality monitoring data and simulated runoff volumes. Section D.3 provides more details on these calculations.

These calculated loads represent a portion of the existing pollutant load to each impaired waterbody. Estimates of loading from other sources are described in other sections or appendices of the TMDL report. The summation of loads from all sources will then be used to estimate existing loading to each lake.

D.2 Simulation of Urban Runoff

D.2.1 MODEL OVERVIEW

The U.S. Environmental Protection Agency's (USEPA) Loading Simulation Program C++ (LSPC) has been used to represent the hydrological and water quality conditions in the Los Angeles River and San Gabriel River watersheds (Tetra Tech, 2004; Tetra Tech, 2005). LSPC is a component of the USEPA's TMDL Modeling Toolbox, which has been developed through a joint effort between USEPA and Tetra Tech. It integrates a geographical information system (GIS), comprehensive data storage and management capabilities, a dynamic watershed model (a re-coded version of USEPA's Hydrological Simulation Program – FORTRAN [HSPF] [Bicknell et al., 2001]), and a data analysis/post-processing system into a convenient PC-based Windows interface that dictates no software requirements. LSPC is capable of representing loading, both flow and water quality, from nonpoint and point sources, and simulating in-stream processes. LSPC can simulate flow, sediment, metals, nutrients, pesticides, and other conventional pollutants, for pervious and impervious lands and waterbodies. Each river basin LSPC model was configured to simulate the respective watershed as a series of hydrologically connected subwatersheds.

Each watershed model represented the variability of nonpoint source contributions through dynamic representation of hydrology and land practices. Each model also included all point and nonpoint source contributions. Key components of the watershed modeling included:

- Watershed segmentation
- Meteorological data
- Land use representation
- Soils
- Reach characteristics
- Point source discharges
- Hydrology representation
- Pollutant representation
- Flow data

D.2.2 WATERSHED SEGMENTATION

In order to evaluate sources contributing to an impaired waterbody and to represent the spatial variability of these sources, the contributing drainage area was represented by a series of subwatersheds. This subdivision was primarily based on the stream networks and topographic variability, and secondarily on the locations of flow and water quality monitoring stations, consistency of hydrologic factors, land use consistency, and existing watershed boundaries.

The subwatersheds for the Los Angeles River basin were delineated after dividing the watershed into two general components: headwaters and lower-elevation urban areas. The headwaters were generally more mountainous and had steeper slopes than the downstream portion of the watershed. In this mountainous region, Digital Elevation Models (DEMs) were utilized for delineating subwatersheds. Specifically, subwatershed boundaries were based upon slopes, ridges, and projected drainage patterns. Alternatively, in the downstream flatter areas of the watershed, maps illustrating the catchment network and drainage pipes were used to isolate sewer-sheds. The Los Angeles River watershed was ultimately delineated into 35 subwatersheds for appropriate hydrologic connectivity and representation (Figure D-2).

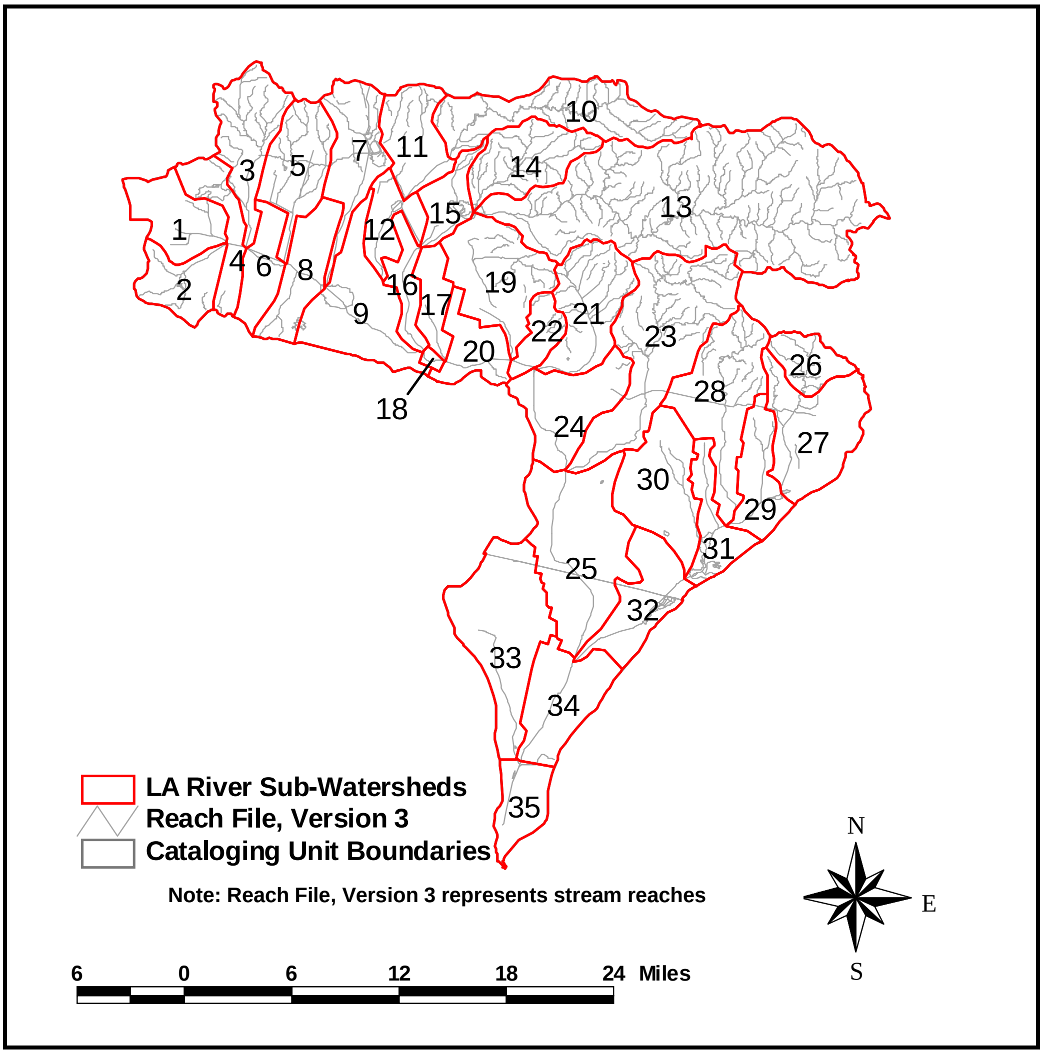

Figure D-2. Subwatershed Delineation for the Los Angeles River Watershed

For the San Gabriel River LSPC model, watershed segmentation was primarily based on the stream networks and topographic variability, and secondarily on the locations of flow and water quality monitoring stations, consistency of hydrologic factors, land use consistency, and existing watershed boundaries (based on CALWTR 2.2 watershed boundaries and municipal storm sewer-sheds). The San Gabriel River watershed was divided into 139 subwatersheds for appropriate hydrologic connectivity and representation (Figure D-3).

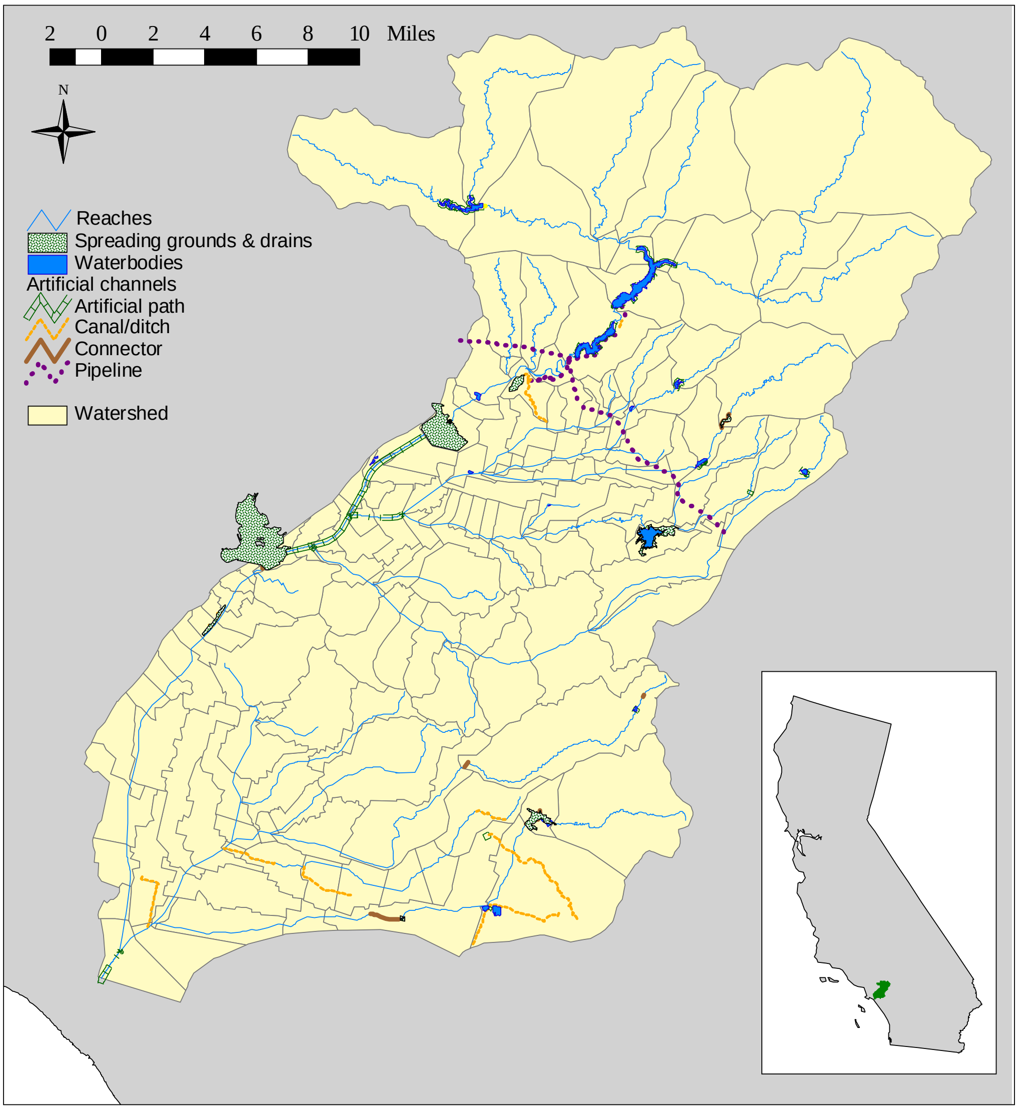

Figure D-3. Subwatershed Delineation for the San Gabriel River Watershed

D.2.3 METEOROLOGICAL DATA

Meteorological data are a critical component of the watershed model. LSPC requires appropriate representation of precipitation and potential evapotranspiration. In general, hourly precipitation (or finer resolution) data are recommended for nonpoint source modeling. Therefore, only weather stations with hourly-recorded data were considered in the precipitation data selection process (note: stations with daily evapotranspiration data were also used and the data were disaggregated to hourly, as describe below). Rainfall-runoff processes for each subwatershed were driven by precipitation data from the most representative station. These data provide necessary input to LSPC algorithms for hydrologic and water quality representation.

Precipitation data available from the National Climatic Data Center (NCDC) were reviewed based on geographic location, period of record, and missing data to determine the most appropriate meteorological stations. Ultimately, hourly rainfall data were obtained from 11 weather stations located in and around the Los Angeles River watershed (see Table D-1 and Figure D-4).

Long-term hourly wind speed, cloud cover, temperature, and dew point data were available for the Los Angeles International Airport (WBAN #23174). These data were obtained from NCDC for the characterization of meteorology of the modeled watersheds. Using these data, hourly potential evapotranspiration was calculated for the Los Angeles River LSPC model.

Table D-1. Precipitation and Meteorological Stations Used in the Los Angeles River LSPC Watershed Model

Station #	Description	Elevation (ft)	Latitude	Longitude
CA1194	BURBANK VALLEY PUMP PLA	655	34.183	-118.333
CA1682	CHATSWORTH RESERVOIR	910	34.225	-118.618
CA3751	HANSEN DAM	1087	34.261	-118.385
CA5085	LONG BEACH AP	31	33.812	-118.146
CA5114	LOS ANGELES WSO ARPT	100	33.938	-118.406
CA5115	LOS ANGELES DOWNTOWN	185	34.028	-118.296
CA5637	MILL CREEK SUMMIT R S	4990	34.387	-118.075
CA7762	SAN FERNANDO PH 3	1250	34.317	-118.500
CA7926	SANTA FE DAM	425	34.113	-117.969
CA8092	SEPULVEDA DAM	680	34.166	-118.473
CA9666	WHITTIER NARROWS DAM	200	34.020	-118.086

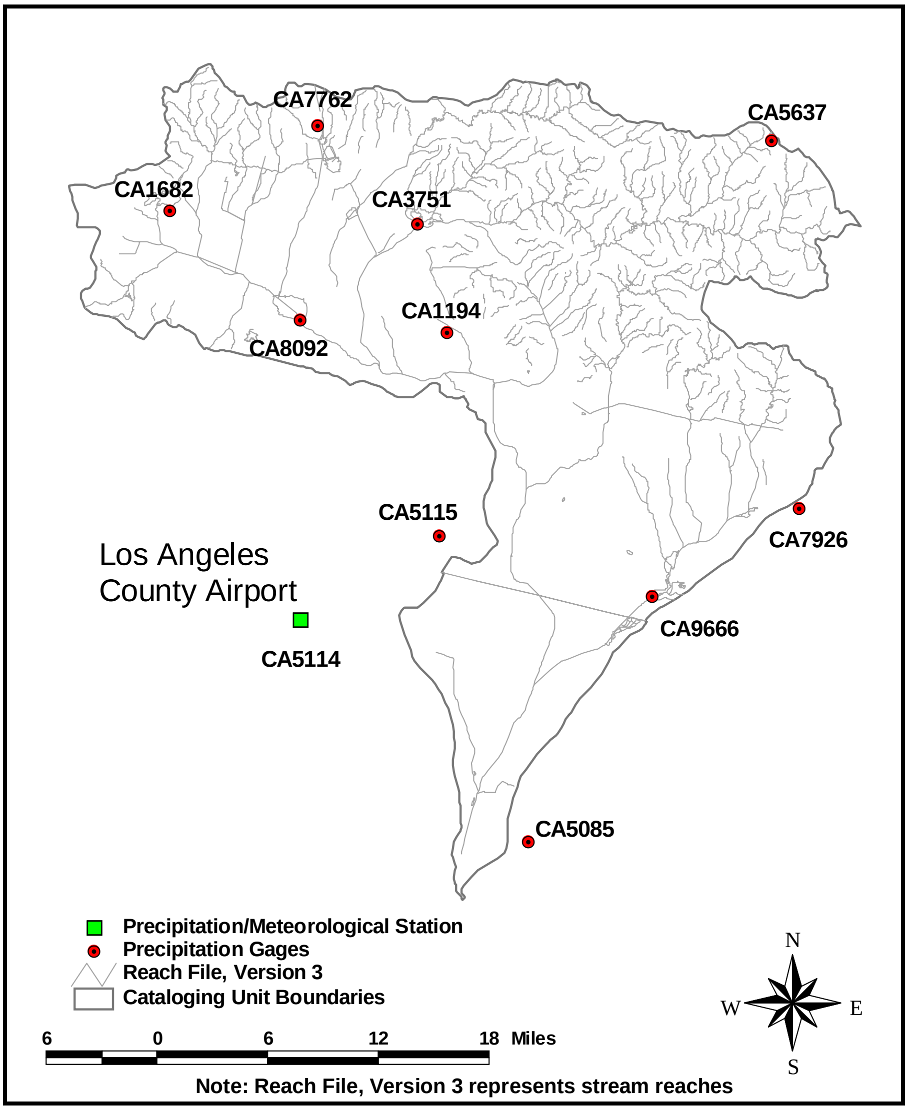

Figure D-4. Location of Precipitation and Meteorological Stations

For the San Gabriel model, hourly rainfall data were obtained from nine weather stations located in and around the watershed (Table D-2 and Figure D-5).

Table D-2. Precipitation Datasets Used for the San Gabriel River Model

Station #	Description	Elevation (ft)
CA1057	Brea Dam	275
CA1272	Cajon West Summit	4,780
CA1520	Carbon Canyon - Workman	1,180
CA5085	Long Beach	31

Station #	Description	Elevation (ft)
CA6473	Orange County Reservoir	660
CA7779	San Gabriel Dam	1,481
CA7926	Santa Fe Dam	425
CA8436	Spadra Lanterman Hospital	676
CA9666	Whittier Narrows Dam	200

Figure D-5. Location of Precipitation Stations for the San Gabriel LSPC Model

Because rainfall gages are not always in operation and accurately recording data, the resulting dataset may contain various intervals of accumulated, missing, or deleted data. Missing or deleted intervals are periods over which either the rainfall gage malfunctioned or the data records were somehow lost.

Accumulated intervals represent cumulative precipitation over several hours, but the exact hourly distribution of the data is unknown. To address the incomplete portions of each dataset, it was necessary to patch the rainfall data with information from nearby gages (see Tetra Tech, 2005 for more information).

Evapotranspiration (ET) data are also required by the LSPC model and were obtained for 10 weather stations from the Los Angeles County Department of Public Works (LADPW) and the California Irrigation Management Information System (CIMIS) (Table D-3 and Figure D-6). The six LADPW stations provided daily ET data while the four CIMIS stations recorded hourly ET. For model input, the daily values were averaged and then disaggregated to hourly increments using hourly data. Specifically, the average hourly percent of total ET from the CIMIS stations was applied to the daily LADPW data, resulting in hourly LADPW ET values. The hourly averages for all 10 stations were then averaged and incorporated into the model weather files.

Table D-3. Evapotranspiration Datasets Used for the San Gabriel River Model

Station #	Description	Elevation (ft)	Source
78	Brea Dam	730	CIMIS
82	Cajon West Summit	1,620	CIMIS
159	Carbon Canyon - Workman	595	CIMIS
174	Long Beach Airport	17	CIMIS
89B	San Dimas Dam	1,350	LADPW
96C	Puddingstone Dam	1,030	LADPW
223B	Big Dalton Dam	1,587	LADPW
334B	Cogswell Dam	2,300	LADPW
390B	Morris Dam	1,210	LADPW
425B	San Gabriel Dam	1,481	LADPW

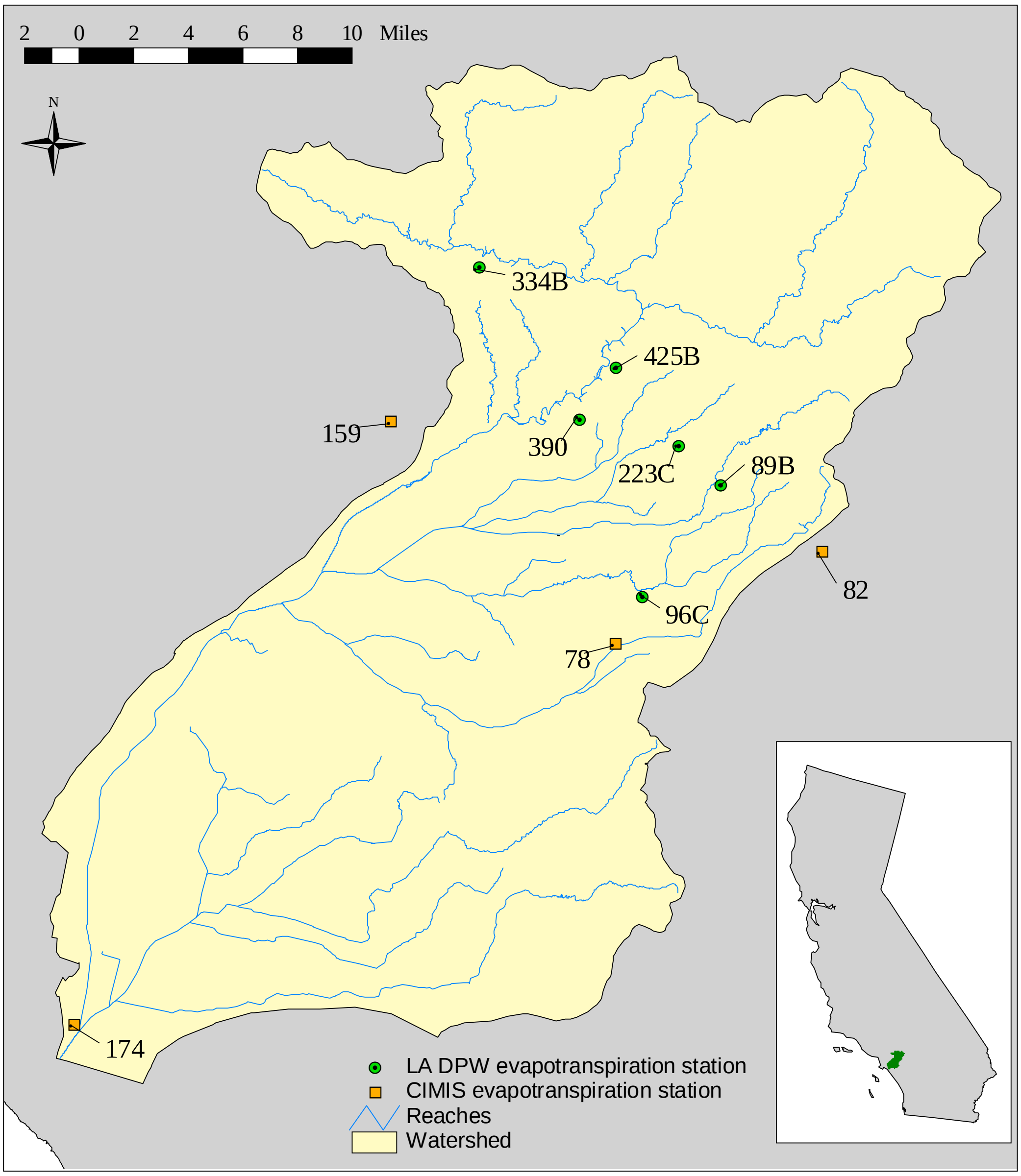

Figure D-6. Location of Evapotranspiration Stations

D.2.4 LAND USE REPRESENTATION

A watershed model requires a basis for distributing hydrologic and pollutant loading parameters. This is necessary to appropriately represent hydrologic variability throughout the basin, which is influenced by land surface and subsurface characteristics. It is also necessary to represent variability in pollutant loading, which is highly correlated to land practices. The basis for this distribution was provided by land use coverages developed for each watershed.

Two sources of land use data were used for the original Los Angeles River LSPC Model. The primary source of data was the County of Los Angeles Department of Public Works (LADPW) 1994 land use dataset that covers the county of Los Angeles. This dataset was supplemented with land use data from the

1993 USGS Multi-Resolution Land Characteristic (MRLC) dataset. For the original San Gabriel Model, the primary source of data was the Southern California Association of Governments (SCAG) 2000 land use dataset that covers the county of Los Angeles. This dataset was supplemented with land use data from the 1993 USGS Multi-Resolution Land Characteristic (MRLC) dataset. More recent land use data (SCAG, 2005) are currently available that did not exist during configuration of the original Los Angeles River and San Gabriel River LSPC models.

For development of these TMDLs, Tetra Tech verified the accuracy, to the extent practicable, of the land use coverages provided with the LSPC models relative to SCAG 2005 land use data. When discrepancies were observed, the land use categorization was updated. Current satellite imagery was used when necessary. Special attention was given to areas classified in the LSPC models as agriculture or strip mines due to the prevalence with which these areas are developed and their relatively high pollutant loading rates (details associated with these modifications are discussed in the lake-specific sections below).

Although the multiple categories in the land use coverages provide much detail regarding spatial representation of land practices in the watershed, such resolution is unnecessary for watershed modeling if many of the categories share hydrologic or pollutant loading characteristics. Therefore, many land use categories were grouped into similar classifications, resulting in a subset of 7 categories for the Los Angeles River model and 12 categories for the San Gabriel River model. Selection of the land use categories was based on the availability of monitoring data and literature values that could be used to characterize individual land use contributions and critical metals-contributing practices associated with different land uses. Land use areas by modeling subbasin are presented in the modeling reports (Tetra Tech, 2004 and 2005).

LSPC algorithms require that land use categories be divided into separate pervious and impervious land units for modeling. This division was made for the appropriate land uses to represent impervious and pervious areas separately. The division was based on typical impervious percentages associated with different land use types defined by LADPW (DePoto et al., 1991).

D.2.5 SOILS

Soil data for each watershed were obtained from the State Soil Geographic Data Base (STATSGO). There are four main Hydrologic Soil Groups (Groups A, B, C, and D). These groups, which are described below, range from soils with low runoff potential to soils with high runoff potential (USDA, 1986).

- Group A Soils have low runoff potential and high infiltration rates even when wet. They consist chiefly of sand and gravel and are well drained to excessively-drained.
- Group B Soils have moderate infiltration rates when wet and consist chiefly of soils that are moderately-deep to deep, moderately- to well-drained, and moderately coarse textured.
- Group C Soils have low infiltration rates when wet and consist chiefly of soils having a layer that impedes downward movement of water with moderately-fine to fine texture.
- Group D Soils have high runoff potential, very low infiltration rates and consist chiefly of clay soils. These soils also include urban areas.

The total area associated with each specific soil type was determined for each model subbasin. The representative soil group for each model subbasin was based on the dominant soil type found in that subwatershed.

D.2.6 REACH CHARACTERISTICS

Each delineated subbasin was represented with a single stream assumed to be a completely mixed, one-dimensional segment with a trapezoidal cross-section. The National Hydrography Dataset (NHD) stream reach network for USGS hydrologic unit 18070105 was used to determine the representative stream reach for each Los Angeles River subbasin. The NHD stream reach network for USGS hydrologic unit 18070106 was used to determine the representative stream reach for each San Gabriel River subbasin. Once the representative reach was identified, slopes were calculated based on DEM data and stream lengths measured from the original NHD stream coverage. In addition to stream slope and length, mean depths and channel widths are required to route flow and pollutants through the hydrologically connected subwatersheds. Mean stream depth and channel width were estimated for the Los Angeles River LSPC model from as-built channel construction drawings provided by the LADPW and were supplemented or verified through field reconnaissance. For the San Gabriel model, mean stream depth and channel width were estimated using regression curves that relate upstream drainage area to stream dimensions.

D.2.7 POINT SOURCE DISCHARGES

Both LSPC models incorporate flows and pollutant loads from major NPDES dischargers in the basin. However, none of these facilities impact the impaired lakes addressed by this TMDL.

D.2.8 HYDROLOGY REPRESENTATION

Watershed hydrology plays an important role in the determination of nonpoint source flow and ultimately nonpoint source loadings to a waterbody. The watershed model must appropriately represent the spatial and temporal variability of hydrological characteristics within a watershed. Key hydrological characteristics include interception storage capacities, infiltration properties, evaporation and transpiration rates, and watershed slope and roughness. LSPC's algorithms are identical to those in the Hydrologic Simulation Program – FORTRAN (HSPF). The LSPC/HSPF modules used to represent watershed hydrology for TMDL development included PWATER (water budget simulation for pervious land units) and IWATER (water budget simulation for impervious land units). A detailed description of relevant hydrological algorithms is presented in the HSPF User's Manual (Bicknell et al., 2001).

Key hydrologic parameters in the PWATER and IWATER modules are infiltration, groundwater flow, and overland flow. USDA's STATSGO Soils Database served as a starting point for designation of infiltration and groundwater flow parameters. For parameter values not easily derived from these sources, documentation on past HSPF applications was accessed. Starting values were refined through the hydrologic calibration process (Tetra Tech, 2004; Tetra Tech, 2005).

D.2.9 FLOW DATA

Flow gaging stations representing relatively diverse hydrologic regions were used for calibration and validation of each LSPC model. Eight stations were selected for calibration of the Los Angeles River LSPC Model because they either had a robust historical record or they were in a strategic location (i.e., along a 303(d)-listed waterbody). The selected flow stations are maintained by the LADPW. Information about each flow station, including location and use in model calibration or validation, is presented in Table D-4 and illustrated in Figure D-7.

Table D-4. Calibration and Validation Stations used in the Los Angeles River LSPC Model

Number	Station Description	Latitude	Longitude	Comment
F45B-R	Rio Hondo above Stuart and Gray Road	33.946	-118.164	Calibration
F300-R	Los Angeles River at Tujunga Ave.	34.141	-118.379	Calibration
F285-R	Burbank Western Storm Drain at Riverside Dr.	34.161	-118.304	Validation
F37B-R	Compton Creek near Greenleaf Drive	33.882	-118.224	Validation
F252-R	Verdugo Wash at Estelle Avenue	34.156	-118.273	Validation
F57C-R	Los Angeles River above Arroyo Seco	34.082	-118.226	Validation
F34D-R	Los Angeles River below Firestone Blvd.	33.949	-118.174	Validation
F319-R	Los Angeles River below Wardlow River Rd.	33.815	-118.205	Validation

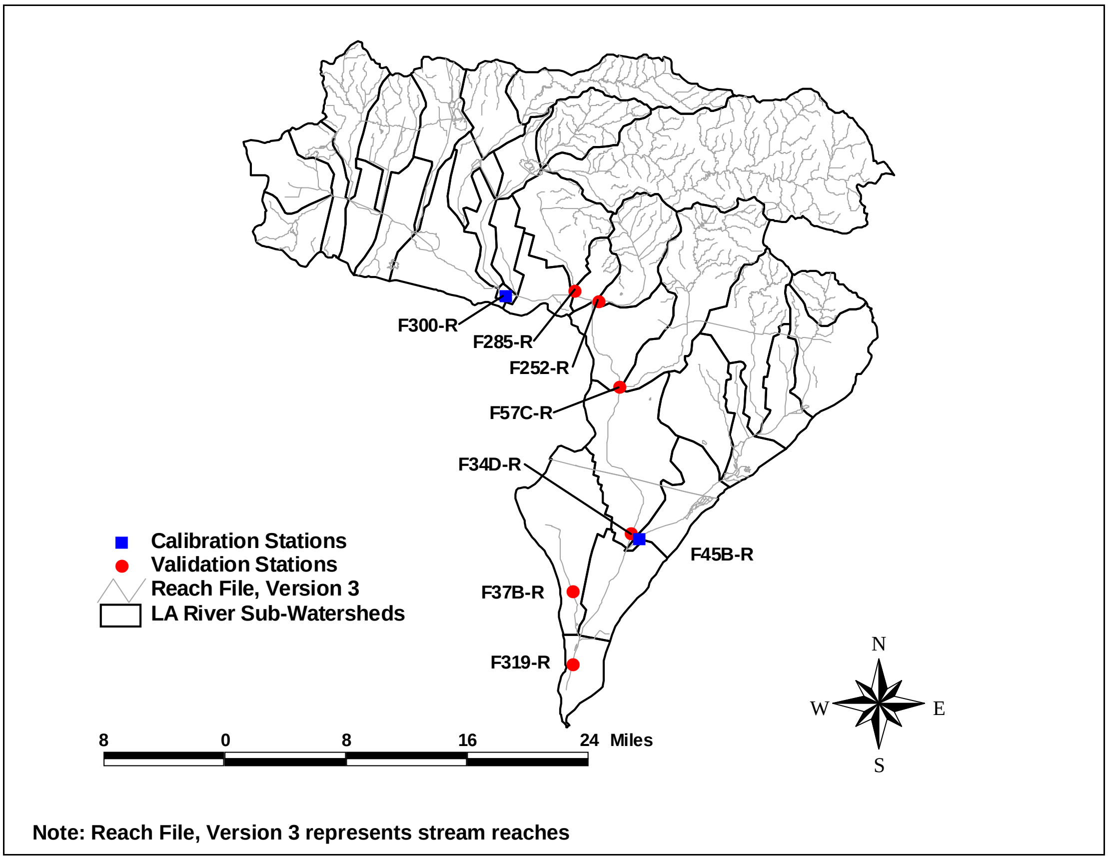

Figure D-7. Location of Hydrology Calibration and Validation Stations for the Los Angeles River LSPC Model

For the San Gabriel Model, 12 flow gaging stations containing full or partial records of flow for the simulation period were identified. These flow stations are maintained by LADPW or the United Stated Geological Survey (USGS). Information about each flow station, including outflow subwatershed, the station identification number (which also indicates the responsible agency) and period(s) used for model calibration and validation, is presented in Table D-5, and their locations are illustrated in Figure D-8.

Table D-5. Flow Data Used for San Gabriel River LSPC Model Calibration and Validation

Gaging Station	Station Description	Outflow Subwatershed	Calibration Dates	Validation Dates
USGS 11089500	Fullerton Creek	56	7/01/94 – 9/30/97	10/01/97 – 9/30/02
USGS 11088500	Brea Creek	59	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F304-R[a]	Walnut Creek	83	1/01/98 – 12/30/02	none
LADPW F274B-R	Dalton Wash	99	10/01/92 – 9/30/95	none
LADPW F312B-R[a]	San Jose Channel	67	10/01/92 – 9/30/94	1/01/98 – 9/30/02
USGS 11087020	San Gabriel River	18	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F262C-R[a]	San Gabriel River	8	1/01/98 – 12/30/02	none
LADPW F42B-R[a]	San Gabriel River	2	1/01/98 – 12/30/02	none
USGS 11085000	San Gabriel River	24	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F190-R	San Gabriel River	26	7/01/94 – 9/30/95	none
LADPW U8-R	San Gabriel River	29	7/01/94 – 9/30/95	none
LADPW F354-R[a]	Coyote Creek	37	12/01/01 – 12/30/02	none

[a] There are various periods of missing data from this gage station.

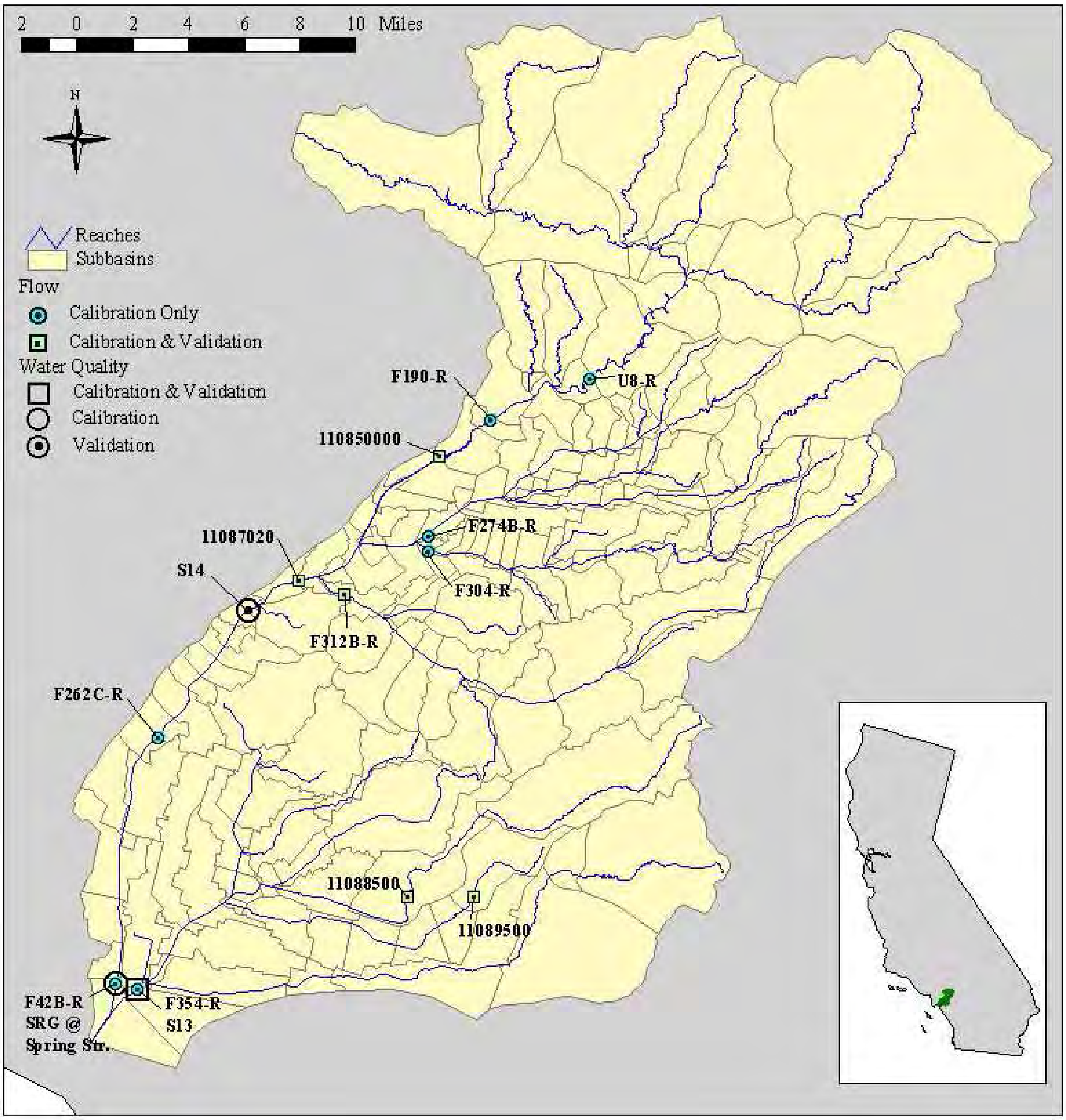

Figure D-8. Locations of Monitoring Stations Used for Model Calibration and Validation of the San Gabriel LSPC Model

The process and results of the flow calibration for each LSPC model is discussed in detail in the respective modeling reports (Tetra Tech, 2004 and 2005).

D.2.10 MODEL OUTPUT

Both of the LSPC models were used to estimate the average monthly runoff depths from land uses present in the watersheds of the impaired waterbodies addressed by this TMDL. Runoff depths by land use were extracted from the LSPC modeling subbasin that contains the watershed of each impaired lake. Table D-6 lists the impaired lakes and associated LSPC model, modeling subbasin, weather station, and dominant hydrologic soil group used to drive the runoff simulation.

Table D-6. Impaired Waterbodies and Associated LSPC Modeling Subbasins and Meteorological Stations

Impaired Lake/Reservoir	LSPC Model	Subbasin #	Meteorological Station	Dominant Soil Group
Peck Road Park	Los Angeles River	27	CA7926	C
Lincoln Park	Los Angeles River	25	CA5115	D
Echo Park	Los Angeles River	25	CA5115	D
Calabasas	Los Angeles River	2	CA1682	D
El Dorado Park	San Gabriel River	46	CA5085	NL[1]
Legg	Los Angeles River	31	CA9666	D
Puddingstone	San Gabriel River	93, 94, 95	CA8436	NL[1]
Santa Fe Dam Park	San Gabriel River	24	CA7926	NL[1]
Sherwood	Los Angeles River	2	CA1682	D

[1] NL: dominant soil hydrologic group is not listed by subbasin in the San Gabriel River Basin LSPC Modeling Report.

Average monthly runoff depths by land use for each impaired waterbody are listed in the respective lake sections of this appendix.

D.3 Event Mean Concentrations

Event mean concentrations (EMCs) represent flow-weighted average concentrations delivered during storm events. Because the LSPC models have not been calibrated to generate loading estimates of key parameters of concern, pollutant EMCs applied to calibrated flow volumes for representative land uses are the best approximation of wet weather loading to the impaired lakes at this time. For these TMDLs, EMCs for nutrients are used to estimate loading associated with runoff volumes. For the mercury and OC Pesticides and PCBs TMDLs, sediment EMCs are used along with simulated runoff volumes and watershed specific water column and sediment concentrations of pollutants to estimate wet weather loading.

For sediment and sediment-associated parameters, the observed instream concentration can be significantly affected by channel scour and deposition processes. The LSPC models are not fully calibrated for such channel processes, which tend to be location-specific. The magnitude of this component for the TMDL watersheds is significantly reduced by the fact that many of the channels are either piped or hardened with concrete and thus not subject to channel degradation. There are, however, portions of the channel network that are not hardened, and even within concrete-lined channels it is expected that there were cycles of deposition and scour of sediment.

Because sufficient data are not available to calibrate detailed models of sediment scour and deposition in reaches, the TMDL analysis is based on an assumption of long-term dynamic equilibrium in the stream network. This approach makes the approximate assumption that the amount of sediment moving through the streams is equivalent (as a long-term average) to the rate of sediment loading to those streams, as estimated from the reported EMCs. Such an assumption was clearly not valid during the earlier period of land use change, construction, and rapid development in the study watersheds, but is believed to provide a reasonable approximation for current conditions.

EMC data for several monitoring years were provided by SCCWRP (Ackerman and Schiff, 2003) and Los Angeles County Department of Public Works (LACDPW) (LACDPW, 2000) for various land uses. Though an EMC may be the same for two seemingly different land uses, loading rates will vary due to differences in runoff volumes. Table D-7 summarizes the EMCs for modeled land uses in the Los Angeles River and San Gabriel River LSPC models.

Table D-7. EMCs for Modeled Land Uses in the Los Angeles and San Gabriel LSPC Models

Los Angeles Model	San Gabriel Model	Sediment (mg/L)	Nitrogen (mg/L)	Phosphorus (mg/L)
Agriculture	Cropland and Pasture	1,520	8.6	0.56
Commercial Other Urban	Commercial and Services Other Urban or Built Up	56.5	4.41	0.67
Industrial	Industrial Transportation, Communication, Utilities	84.7	4.55	0.58
Open	Evergreen Forest Land Herbaceous Rangeland Mixed Rangeland Shrub and Brush Rangeland	28.83	3.2	0.11
Residential	Residential Transitional Areas	55.2	4.51	0.73
NA	Strip Mines	1,520	4.55	0.58

The 12 land uses simulated by the San Gabriel River Basin LSPC Model were aggregated to modeled land use categories as presented in Table D-8. The table also lists the impervious fractions of each urban land use simulated by the model (rural land uses are assumed 100 percent pervious). The impervious area is the major determinant of total runoff volume for urban land uses. At the basin-wide scale, areas classified as “other urban or built-up” were simulated as commercial areas with 65 percent imperviousness. Comparison of the LSPC land use coverage to SCAG 2005 data and recent satellite imagery indicate that areas adjacent to the impaired waterbodies classified as “other urban or built-up” are actually parkland. To predict runoff volumes and sediment loading from these areas, model output for the pervious fraction of commercial areas was assumed representative of parkland.

Table D-8. Aggregation of Land Use Classes in the LSPC Model

Original Land Use	Modeled Land Use
Commercial and services	Commercial (80 percent impervious)
Cropland and pasture	Cropland
Evergreen forest land	Forest
Herbaceous rangeland	Pasture
Industrial	Industrial (80 percent impervious)
Mixed rangeland	Pasture
Other urban or built-up[1]	Commercial (65 percent impervious)
Residential	Residential (19 percent impervious)
Shrub & brush rangeland	Pasture
Trans, comm, util	Transportation (80 percent impervious)
Transitional areas	Residential (10 percent impervious)

[1] Other urban or built-up areas surrounding impaired waterbodies are parkland and are simulated as commercial with zero percent imperviousness.

Runoff depths for the simulated land uses vary by LSPC modeling subbasin. The subsequent sections of this report summarize the monthly average runoff depths for land uses draining to each of the impaired lakes. EMCs are applied to runoff depths for a corresponding area to estimate pollutant loading associated with a particular area. For example, the average runoff depth in January for agricultural lands in the Peck Road Park Lake watershed is 0.5361 inches (Table D-12). There are 4.19 acres of agriculture present in the Western Subwatershed (Table D-9).

The following calculation estimates the total nitrogen load delivered during January from this area:

$$\frac{8.6mg-N}{L}\cdot\frac{0.536in}{mo}\cdot\frac{1ft}{12in}\cdot 4.19ac\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=4.38lb-N$$

For the waterbodies impaired by mercury and OC Pesticides and PCBs, watershed specific monitoring data were available to estimate loading to each lake. The dissolved portion of the load can be represented as flow multiplied by the observed water column concentration, and the sediment-associated portion of the load can be represented as average sediment movement times the observed concentration in stream sediment.

For example, the near lake undeveloped subwatershed draining to Lake Sherwood is comprised of 197 acres of open space (Table D-64). The monthly average runoff depth for open space in January is 0.3808 inches (Table D-66), and the sediment EMC is 28.83 mg/L (Table D-7). The winter season water column and sediment-associated concentrations of total mercury are 2.96 ng/L (Table D-68) and 129 µg/kg (Table D-69), respectively. The total mercury delivered from runoff generated in January from this area is

$$197ac \cdot \frac{0.3808in}{mo} \cdot \frac{1ft}{12in} \cdot \frac{43{,}560\,ft^2}{ac} \cdot \frac{28.32L}{ft^3} \cdot \frac{2.96ng - Hg}{L} \cdot \frac{1g}{1 \cdot 10^9\,ng} = 0.0228g - Hg$$

The total mercury (Hg) associated with the delivered sediment (sed) in January is

$$197ac \cdot \frac{0.3808in}{mo} \cdot \frac{1ft}{12in} \cdot \frac{43{,}560\,ft^2}{ac} \cdot \frac{28.32L}{ft^3} \cdot \frac{28.83mg - sed}{L} \cdot \frac{1g - sed}{1{,}000mg - sed} \cdot \frac{129ng - Hg}{g - sed} \cdot \frac{1g - Hg}{1 \cdot 10^9\,ng - Hg} = 0.0287g - Hg$$

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D.4 Peck Road Park Lake

Peck Road Park Lake is located in the Los Angeles River Basin. However, the LACDPW diverts flows from the San Gabriel River to Peck Road Park Lake via the Santa Fe Diversion Channel.

Impairments of this lake include low dissolved oxygen/organic enrichment, eutrophication (originally on the consent decree, but currently delisted), odor, lead, chlordane, DDT, dieldrin, PCBs, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads from upland areas of OC Pesticides and PCBs and nutrients, which may be contributing to the low dissolved oxygen/organic enrichment, eutrophication, and odor impairments. Loads from the diversion are estimated from measured flow volumes and area-weighted event mean concentrations for the land use classes upstream of the diversion channel.

Three subwatersheds comprise the drainage area to Peck Road Park Lake. The subwatershed draining the western part of the watershed via Santa Anita Wash is 12,686 acres, and the eastern subwatershed draining to Saw Pit Wash is 10,557 acres. There is an inwardly draining mining operation in the southern part of the eastern watershed that has been removed from the loading analysis. The area surrounding the lake is 321 acres. Each subwatershed drains to a storm sewer system so all allocations for the TMDLs are wasteload allocations (except for the trash TMDL which also has a load allocation).

Figure D-9 shows the MS4 stormwater permittees in the Peck Road Park Lake watershed. The western subwatershed is comprised of the county of Los Angeles, Sierra Madre, Arcadia, Monrovia, Angeles National Forest, and Caltrans areas. The eastern subwatershed is comprised of the county of Los Angeles, Monrovia, Duarte, Bradbury, Arcadia, Irwindale, Angeles National Forest, and Caltrans areas. The county of Los Angeles, Monrovia, Irwindale, Arcadia, and El Monte comprise the drainage around the lake. The park area is comprised of 152 acres adjacent to the lake.

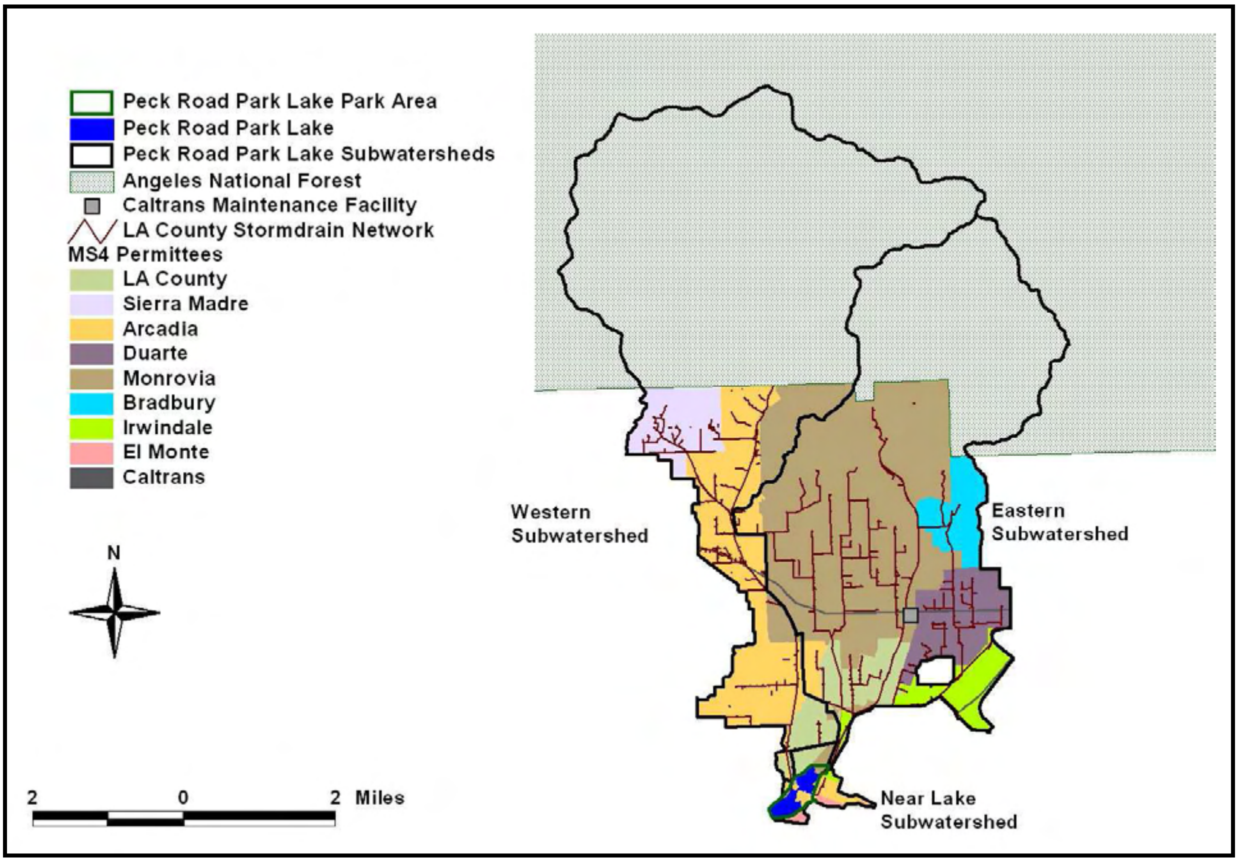

Figure D-9. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Peck Road Park Lake Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-10. Upon review of the SCAG 2005 database, as well as current satellite imagery, it was evident that a portion of the areas classified by the LSPC model as agriculture were inaccurate. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Approximately 82 acres classified by LSPC as agriculture corresponded to orchards, vineyards, and horse farms and were not altered. However, approximately 27 acres of agriculture was reclassified as open space and 28 acres were reclassified as residential. Areas classified as industrial or commercial in the Angeles National Forest were also inaccurate and were reclassified as open. Inaccuracies in land use assignment were corrected for each subwatershed and jurisdiction to reflect the more recent SCAG 2005 dataset and current satellite imagery. All areas within the Caltrans jurisdiction were simulated as industrial since the Los Angeles River Basin LSPC model lumped transportation uses into the industrial category. Table D-9, Table D-10, and Table D-11 summarize the post-processed land use areas used to estimate pollutant loading from upland areas draining to Peck Road Park Lake.

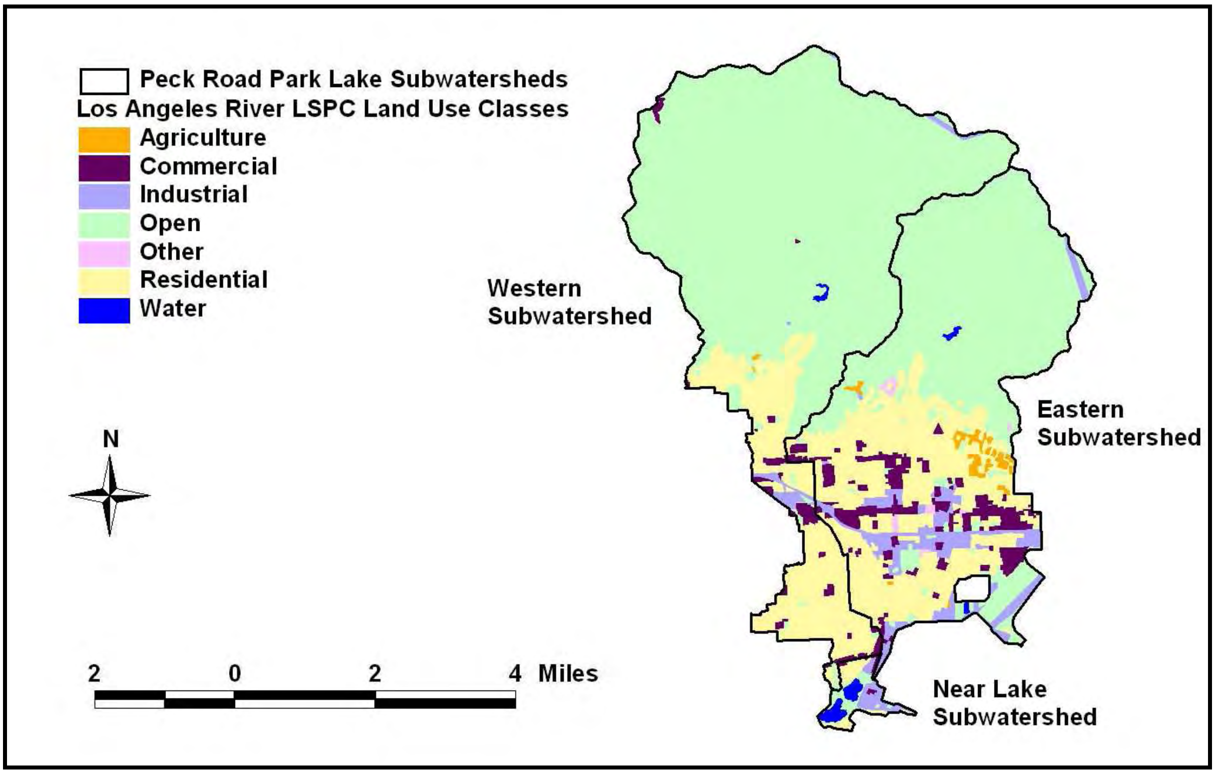

Figure D-10. LSPC Land Use Classes for the Peck Road Park Lake Subwatersheds

Table D-9. Land Use Areas (ac) Draining from the Western Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Sierra Madre	Arcadia	Monrovia	Caltrans	Angeles National Forest	Total
Agriculture	0	4.19	0	0	0	0	**4.19**
Commercial	34.8	2.62	124	13.0	0	0	**175**
Industrial	0	0	70.4	0.319	16.9	0	**87.6**
Open	3.50	377	319	483	0	9,104	**10,286**
Other Urban	0	0	0.053	0	0	0	**0.053**
Residential	207	296	1,516	114	0	0	**2,133**
Total	**245**	**679**	**2,030**	**611**	**16.9**	**9,104**	**12,686**

Table D-10. Land Use Areas (ac) Draining from the Eastern Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Duarte	Bradbury	Arcadia	Irwindale	Caltrans	Angeles National Forest	Total
Agriculture	0	0	0	78.1	0	0	0	0	**78.1**
Commercial	24.8	430	232	0	33.9	12.7	0	0	**733**
Industrial	1.27	407	107	0	0	180	78.4	0	**774**
Open	5.29	1,419	53.5	229	16.0	274	0	3,511	**5,508**
Other Urban	0	51.0	1.74	2.90	1.71	0	0	0	**57.3**
Residential	467	2,149	424	193	158	15.5	0	0	**3,406**
Total	**499**	**4,456**	**818**	**503**	**209**	**483**	**78.4**	**3,511**	**10,557**

Table D-11. Land Use Areas (ac) Draining from the Near Lake Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Irwindale	Arcadia	El Monte	Total
Agriculture	0	0	0	0	0	**0**
Commercial	7.10	7.90	0	3.86	0	**18.9**
Industrial	0.0003	14.4	13.9	69.7	10.2	**108**
Open	0.233	24.6	0.187	61.6	0.984	**87.5**
Other Urban	0	0	0	0	0	**0**
Residential	60.4	1.30	0	4.18	40.9	**107**
Total	**67.7**	**48.1**	**14.1**	**139**	**52.1**	**321**

The land use composition upstream of the San Gabriel River at the diversion to Peck Road Park Lake is primarily rangeland (56 percent) and forest (40 percent). The remaining 4 percent is comprised of other types of open areas and urban development. To estimate the pollutant concentrations associated with the diverted flows, EMCs (Section D.3) were area-weighted based on the land use composition upstream of the diversion.

D.4.1 RUNOFF AND DIVERTED FLOWS

LSPC-predicted runoff from the Peck Road Park Lake subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-11 shows the simulated annual rainfall for the Peck Road Park Lake subwatersheds. The annual average rainfall is 19.1 inches.

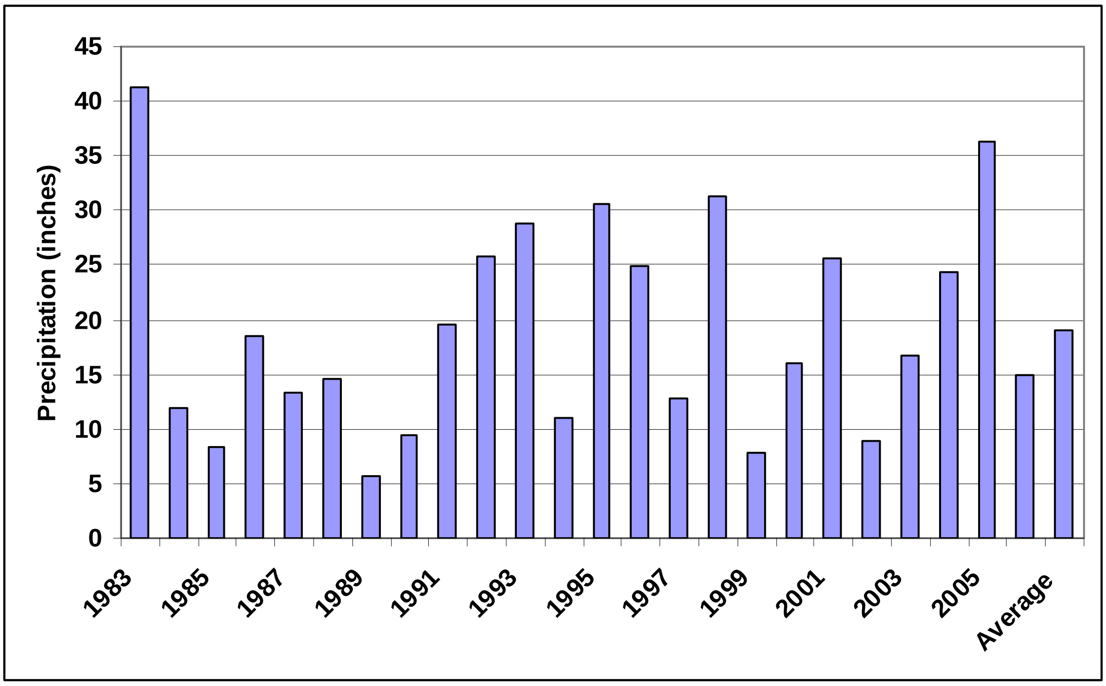

Figure D-11. Annual Rainfall for the Peck Road Park Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the Peck Road Park Lake subwatersheds are shown in Table D-12.

Table D-12. Monthly Average Runoff Depths (inches/month) for Land Uses in the Peck Road Park Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.5361	3.0291	2.7414	0.1966	2.0223	1.9645
February	0.8942	3.9665	3.6105	0.4150	2.7206	2.6491
March	0.5614	2.4735	2.2559	0.2416	1.7120	1.6683
April	0.1153	0.8499	0.7608	0.0548	0.5381	0.5202
May	0.0531	0.2477	0.2250	0.0216	0.1682	0.1636
June	0.0097	0.1020	0.0904	0.0053	0.0614	0.0591
July	0.0010	0.0090	0.0080	0.0006	0.0054	0.0052
August	0.0047	0.0632	0.0558	0.0024	0.0373	0.0358
September	0.0163	0.2219	0.1959	0.0080	0.1312	0.1260
October	0.0407	0.5706	0.5037	0.0202	0.3364	0.3230
November	0.0684	0.9569	0.8447	0.0339	0.5641	0.5416
December	0.1226	1.5882	1.4051	0.0575	0.9475	0.9108

The LACDPW provided Tetra Tech with mean daily flows measured over the past 15 years (October 1994 through May 2009) in the diversion channel that directs flow from the San Gabriel River to Peck Road Park Lake. The average monthly flows from this diversion are summarized in Table D-13.

Table D-13. Average Monthly Flow Volumes Diverted to Peck Road Park Lake

Month	Diverted Flow (ac-ft)
January	223
February	229
March	981
April	717
May	1,028
June	2,039
July	1,134
August	343
September	854
October	718
November	76.8
December	395
Total	**8,737**

Figure D-12 summarizes the monthly average runoff and diversion volumes delivered to Peck Road Park Lake. The total annual volume delivered to the lake is 16,529 ac-ft, and approximately half the flow is from the San Gabriel diversion. Flows during the months May through October are primarily from the diversion channel.

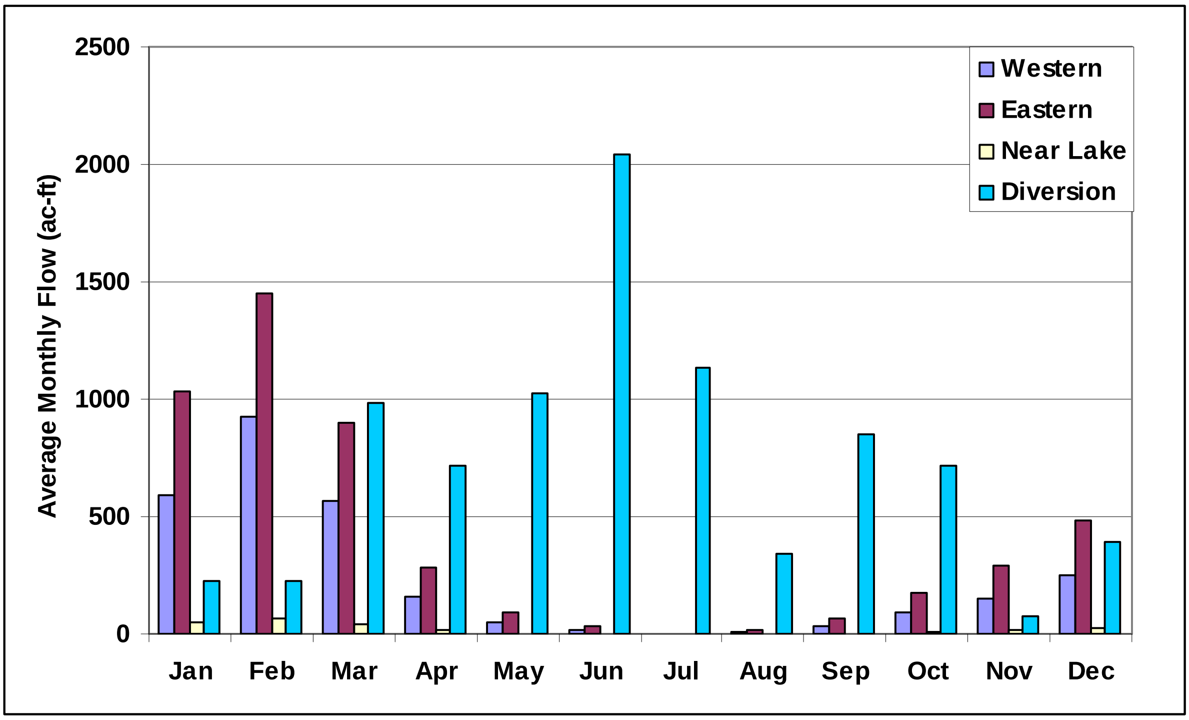

Figure D-12. Monthly Average Runoff Volumes to Peck Road Park Lake

D.4.2 Sediment Loads

Sediment loads are calculated from simulated volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-14 summarizes the average annual sediment loads for each jurisdiction by subwatershed. Sediment loads estimated for the diversion are included as well. See example calculations in Section D.3.

Table D-14. Average Annual Sediment Loads to Peck Road Park Lake

Subwatershed	Jurisdiction	Sediment (tons/yr)
Eastern	Arcadia	12.1
Eastern	Bradbury	44.4
Eastern	Caltrans	9.55
Eastern	Duarte	58.0
Eastern	Irwindale	24.9
Eastern	County of Los Angeles	28.6
Eastern	Monrovia	217
Eastern	Angeles National Forest	12.1
Near Lake	Arcadia	9.29
Near Lake	El Monte	3.55
Near Lake	Irwindale	1.70
Near Lake	County of Los Angeles	4.03

Subwatershed	Jurisdiction	Sediment (tons/yr)
Near Lake	Monrovia	2.62
Western	Arcadia	106
Western	Caltrans	2.06
Western	County of Los Angeles	14.7
Western	Monrovia	9.27
Western	Sierra Madre	19.9
Western	Angeles National Forest	31.4
Diversion		379
Total		**990**

D.4.3 NUTRIENT LOADS

Nutrient loads are estimated from simulated volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-15 summarizes the total nitrogen and total phosphorus loads delivered to Peck Road Park Lake from each jurisdiction and subwatershed or from the diversion. See example calculations in Section D.3.

The loads presented in the table are existing loads, not allocated loads.

Table D-15. Average Annual Nutrient Loads to Peck Road Park Lake

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Eastern	Arcadia	1,951	309
Eastern	Bradbury	2,337	320
Eastern	Caltrans	1,027	131
Eastern	Duarte	8,606	1,307
Eastern	Irwindale	2,891	358
Eastern	County of Los Angeles	4,653	749
Eastern	Monrovia	32,627	4,894
Eastern	Angeles National Forest	2,692	92.5
Near Lake	Arcadia	1,053	132
Near Lake	El Monte	510	77.8
Near Lake	Irwindale	183	23.3
Near Lake	County of Los Angeles	653	105
Near Lake	Monrovia	330	43.4
Western	Arcadia	16,812	2,641
Western	Caltrans	221	28.2
Western	County of Los Angeles	2,386	381
Western	Monrovia	1,601	210
Western	Sierra Madre	3,056	456
Western	Angeles National Forest	6,981	240

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Diversion		76,970	2,960
Total		**167,539**	**15,458**

D.4.4 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant of concern using monitoring data collected by USEPA, the Regional Board, and UCLA, between 2008 and 2009. Only data from sites representing inflows are used; these sites include locations in an inflow, or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Peck Road Park Lake, this included PRPL-6, PRPL-7, PRPL-12 and PRPL-13 (Figure D-13).

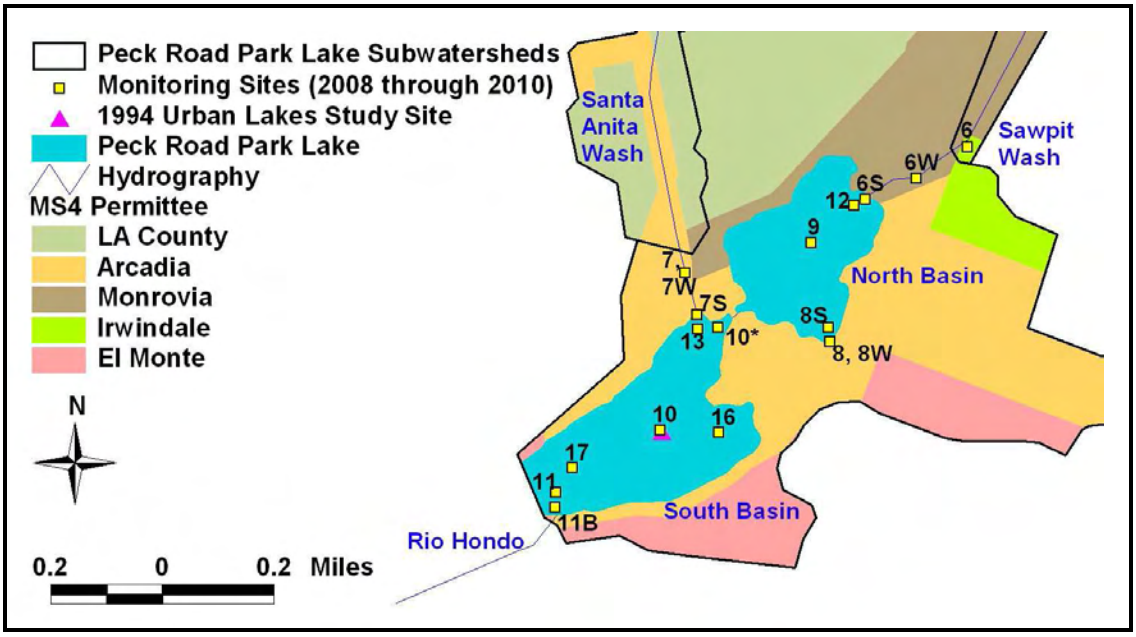

Figure D-13. Monitoring Stations at Peck Road Park Lake

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads will be associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For OC Pesticides and PCBs where some of the samples had detectable quantities of a pollutant, the average inflow concentration was calculated assuming samples analyzed below detection limits were equal to one-half the detection limit. For all of the sediment samples, dieldrin was below detection and reporting levels . Instead, an upper-bound analysis was performed using the detection limit as the incoming concentration associated with the sediment. The average concentration of total chlordane in sediments associated with inputs was 3.15 µg/kg dry weight and the average level of PCBs was 15.38 µg/kg dry weight. The average concentration of DDT was 5.57 µg/kg dry weight. The inflow sediment data are summarized in Table D-16 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-16. Summary of Sediment Data near Inflow Locations at Peck Road Park Lake

Parameter	Number of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit (µg/kg dry weight)
Chlordane	6	2	3.15	0.34-1
DDT	6	3	5.57	0.69-1.18
Dieldrin	6	0	(0.91)[2]	0.69-1.18
Total PCBs	6	5	15.38	0.34-1

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

[2] All sample results were below detection limits. An upper-bound analysis was performed using the highest reported detection limit for dieldrin.

These input sediment concentrations were applied to the calculated sediment loads (Section D.4.2) to estimate sediment-associated OC Pesticides and PCBs loads entering the lake. Specifically, to determine sediment loading of OC Pesticides and PCBs, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads (Table D-17), and the sediment OC Pesticides and PCBs concentrations from monitoring data (Table D-16) were then applied to these sediment loads.

Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the three Peck Road Park Lake subwatersheds in Table D-17 along with the predicted sediment loads for each land use. Dissolved concentrations in inflows are assumed insignificant.

Table D-17. Annual Sediment Load to Peck Road Park Lake

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Eastern	Arcadia	209	12.1	1.22%
Eastern	Bradbury	503	44.4	4.48%
Eastern	Caltrans	78.4	9.6	0.96%
Eastern	Duarte	785	57.2	5.78%
Eastern	General Industrial Stormwater Permittees* (in the city of Duarte)	33	0.8	0.08%
Eastern	Irwindale	463	23.3	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	19.9	1.6	0.16%
Eastern	County of Los Angeles	499	28.6	2.89%
Eastern	Monrovia	4,323	200	20.2%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	134	16.3	1.65%

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Eastern	Angeles National Forest	3,511	12.1	1.22%
Diversion	Los Angeles County Department of Public Works	-	379	38.3%
Near Lake	Arcadia	125	7.59	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	14	1.70	0.17%
Near Lake	El Monte	52.1	3.55	0.36%
Near Lake	Irwindale	14.1	1.70	0.17%
Near Lake	County of Los Angeles	67.7	4.03	0.41%
Near Lake	Monrovia	48.1	2.62	0.26%
Western	Arcadia	1,720	68.1	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	310	37.8	3.82%
Western	Caltrans	16.9	2.06	0.21%
Western	County of Los Angeles	245	14.7	1.49%
Western	Monrovia	611	9.27	0.94%
Western	Sierra Madre	679	19.9	2.01%
Western	Angeles National Forest	9,104	31.4	3.18%
Total		**23,564**	**990.3**	**100%**

*The disturbed area associated with general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

The chlordane, PCB, DDT, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-16) to the calculated sediment load of 900.3 tons per year (Table D-17). See example calculations in Section D.3. Loads for each jurisdiction are shown by subwatershed in Table D-18.

Table D-18. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Peck Road Park Lake Watershed (g/yr)

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual DDT Load[1]	Annual Dieldrin Load[1]	Percent of Total Load
Eastern	Arcadia	0.17	0.034	0.061	<0.010	1.22%
Eastern	Bradbury	0.62	0.127	0.224	<0.037	4.48%
Eastern	Caltrans	0.13	0.027	0.048	<0.008	0.96%
Eastern	Duarte	0.80	0.163	0.289	<0.047	5.78%

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual DDT Load[1]	Annual Dieldrin Load[1]	Percent of Total Load
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	0.01	0.002	0.004	<0.001	0.08%
Eastern	Irwindale	0.33	0.067	0.118	<0.019	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	0.02	0.005	0.008	<0.001	0.16%
Eastern	County of Los Angeles	0.40	0.082	0.145	<0.024	2.89%
Eastern	Monrovia	2.80	0.573	1.013	<0.165	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	0.23	0.047	0.0821.65%0.061	<0.013	1.65%
Eastern	Angeles National Forest	0.17	0.035	1.917	<0.010	1.22%
Diversion	Los Angeles County Department of Public Works	5.29	1.084	0.038	<0.313	38.3%
Near Lake	Arcadia	0.11	0.022	0.009	<0.006	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	0.02	0.005	0.018	<0.001	0.17%
Near Lake	El Monte	0.05	0.010	0.009	<0.003	0.36%
Near Lake	Irwindale	0.02	0.005	0.020	<0.001	0.17%
Near Lake	County of Los Angeles	0.06	0.012	0.013	<0.003	0.41%
Near Lake	Monrovia	0.04	0.007	0.344	<0.002	0.26%
Western	Arcadia	0.95	0.195	0.191	<0.056	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	0.53	0.108	0.010	<0.031	3.82%
Western	Caltrans	0.03	0.006	0.074	<0.002	0.21%
Western	County of Los Angeles	0.21	0.042	0.047	<0.012	1.49%
Western	Monrovia	0.13	0.026	0.100	<0.008	0.94%
Western	Sierra Madre	0.28	0.057	0.159	<0.016	2.01%
Western	Angeles National Forest	0.44	0.090	0.061	<0.026	3.18%
Total		**13.7**	**2.83**	**5.00**	**0.818**	**100%**

[1] Results from upper-bound analysis representing the maximum possible dieldrin load.

[2] The disturbed area associated with general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

D.5 Lincoln Park Lake

Lincoln Park Lake is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, lead, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations has been used to estimate loads from upland areas of nutrients, which may be contributing to the low dissolved oxygen/organic enrichment, odor, eutrophication, and ammonia impairments.

Figure D-14 shows the MS4 stormwater permittee in the Lincoln Park Lake watershed (the city of Los Angeles). Though the lake appears to be connected to the county of Los Angeles storm drain network, this system actually passes under Lincoln Park Lake and does not discharge stormwater to the lake. The subwatershed for Lincoln Park Lake (37.1 acres) is comprised only of the surrounding parklands. All loads generated from this area are assigned load allocations for TMDL development.

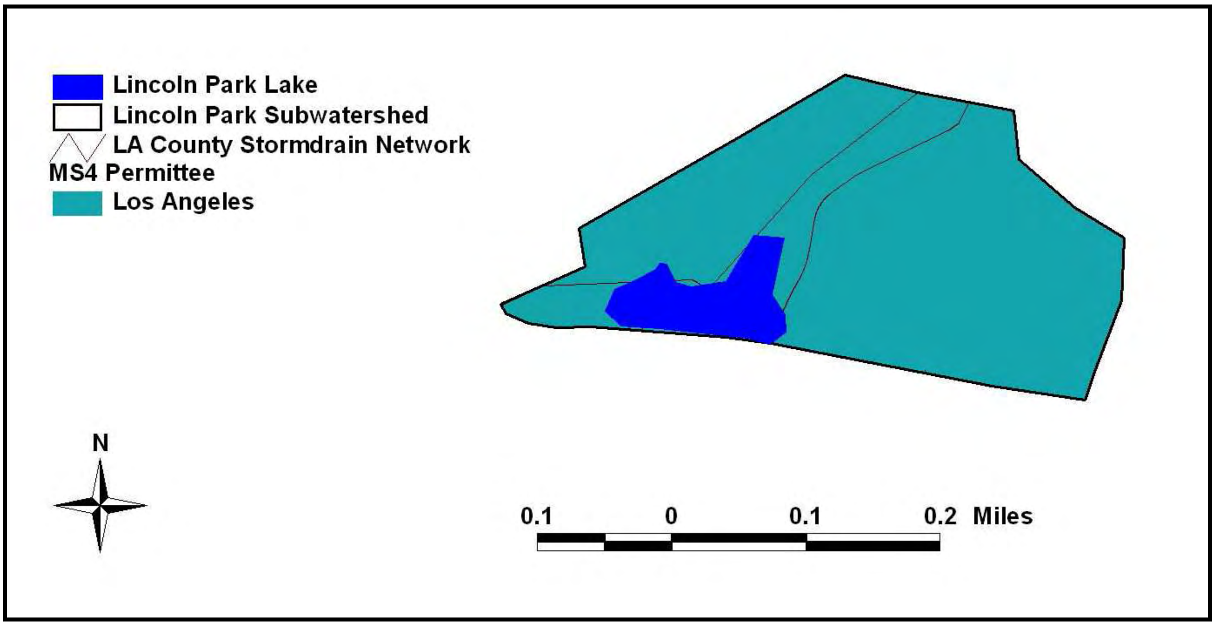

Figure D-14. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lincoln Park Lake Watershed

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-15. The watershed is comprised of open space and industrial areas. Table D-19 summarizes the land use areas used to estimate pollutant loading from upland areas draining to Lincoln Park Lake.

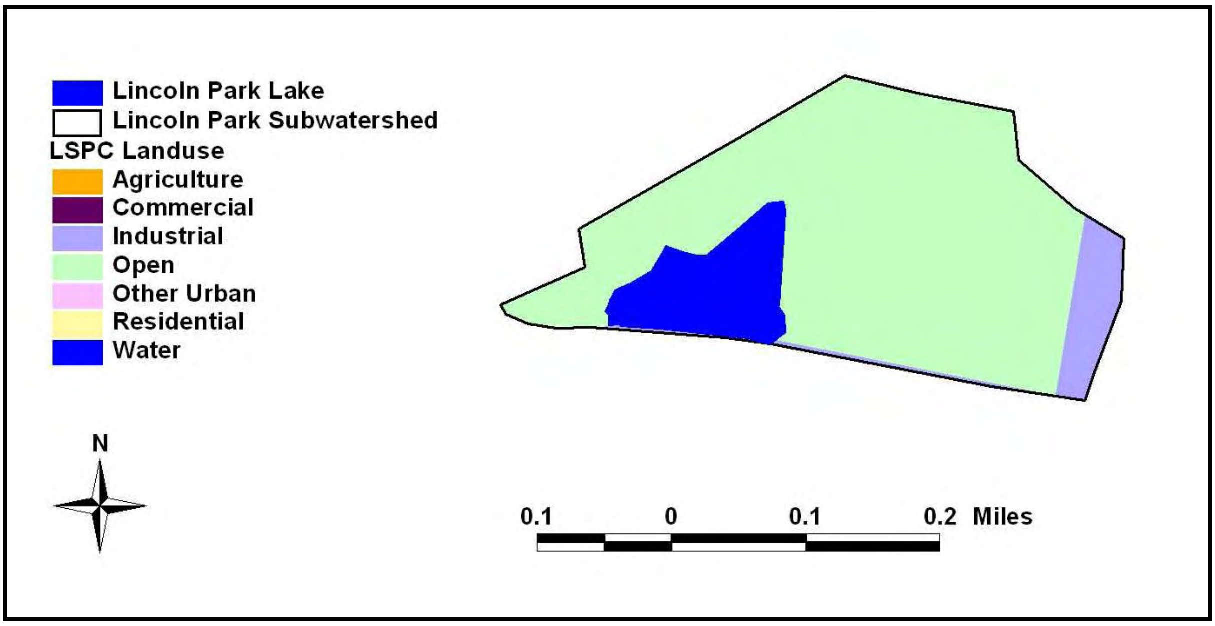

Figure D-15. LSPC Land Use Classes for the Lincoln Park Lake Watershed

Table D-19. Land Use Areas (ac) Draining to Lincoln Park Lake

Land Use	Los Angeles
Agriculture	0
Commercial	0
Industrial	3.40
Open	33.7
Other Urban	0
Residential	0
Total	**37.1**

D.5.1 RUNOFF

LSPC-predicted runoff from the Lincoln Park Lake watershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-16 shows the simulated annual rainfall for the Lincoln Park Lake watershed. The annual average rainfall is 15.2 inches.

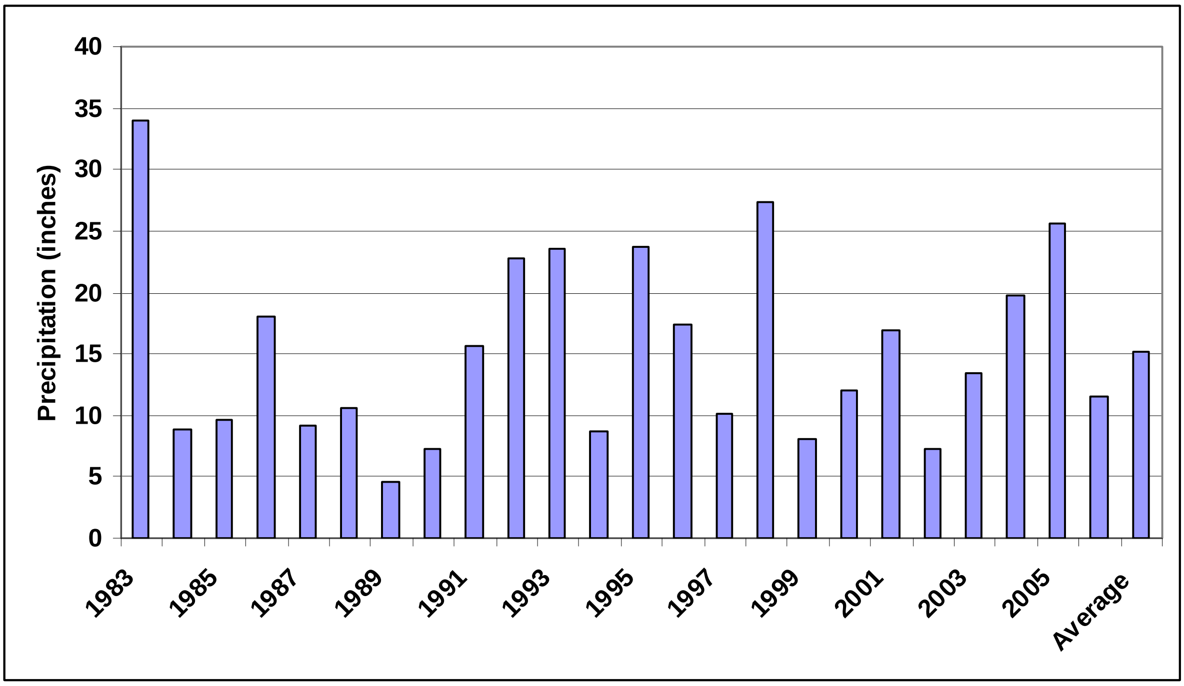

Figure D-16. Annual Rainfall for the Lincoln Park Lake Watershed

The simulated monthly average runoff depths for land uses in the Lincoln Park Lake watershed are shown in Table D-20.

Table D-20. Monthly Average Runoff Depths (inches/month) for Land Uses in the Lincoln Park Lake Watershed, 1983 - 2006

Month	Industrial	Open
January	2.0170	0.0963
February	2.7225	0.1613
March	1.7918	0.1136
April	0.5372	0.0334
May	0.1602	0.0094
June	0.0475	0.0024
July	0.0024	0.0002
August	0.0232	0.0010
September	0.1352	0.0055
October	0.3393	0.0136
November	0.6098	0.0244
December	1.2099	0.0487

Figure D-17 summarizes the monthly average runoff volumes delivered to Lincoln Park Lake. The total annual volume delivered to the lake is 4.15 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

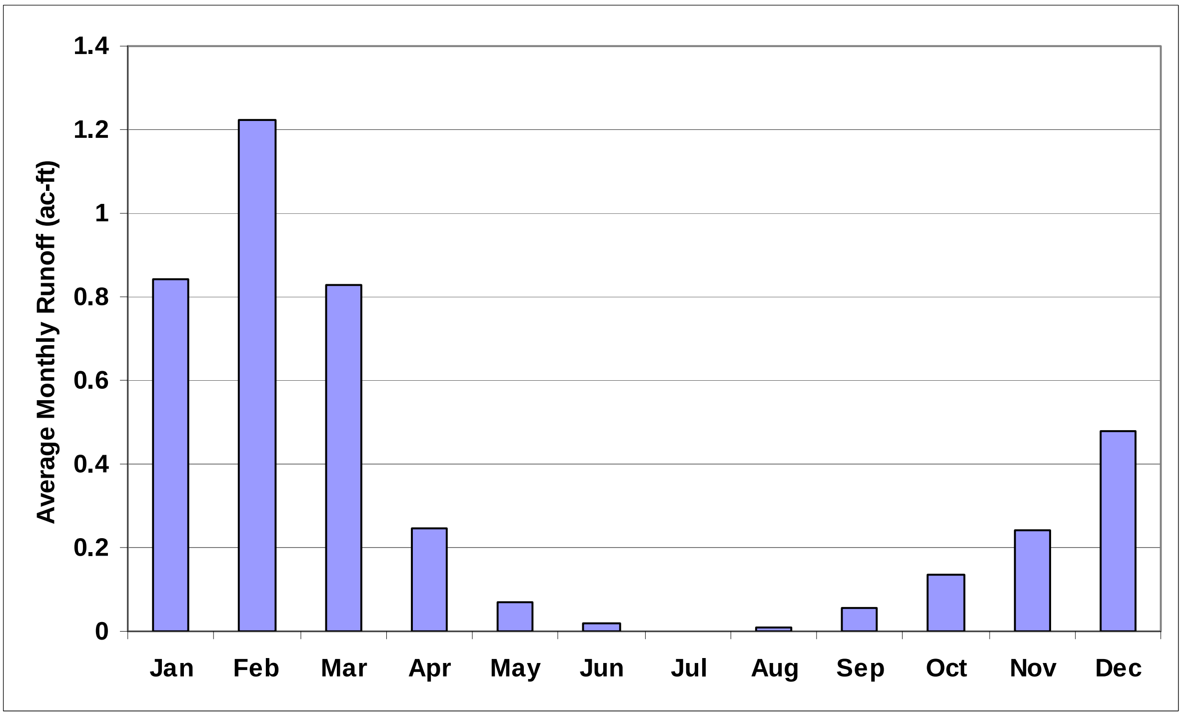

Figure D-17. Monthly Average Runoff Volumes to Lincoln Park Lake

D.5.2 NUTRIENT LOADS

Nutrient loads are estimated from simulated volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-21 summarizes the total nitrogen and total phosphorus loads estimated for Lincoln Park Lake. See example calculations in Section D.3.

Table D-21. Average Annual Nutrient Loads to Lincoln Park Lake

Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Los Angeles	46.1	4.72

D.6 Echo Park Lake

Echo Park Lake is located in the Los Angeles River Basin. Impairments of this lake include odor, ammonia, eutrophication, algae, pH, copper, lead, PCBs, dieldrin, chlordane, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads from upland areas of OC Pesticides and PCBs and nutrients, which may be contributing to the odor, ammonia, eutrophication, algae, and pH impairments.

Two subwatersheds comprise the drainage area to Echo Park Lake. The subwatershed draining the northern part of the watershed is 614 acres and the southern subwatershed drains 170 acres. Both subwatersheds drain to a storm drain system, so all allocations for the TMDLs are wasteload allocations (except for the trash TMDL which also has a load allocation). Dry weather flows from the storm drain system are diverted downstream of Echo Park Lake. Figure D-18 shows the MS4 stormwater permittees in the Echo Park Lake watershed. Both subwatersheds are located entirely within the city of Los Angeles with a small portion of Caltrans area. The park is comprised of 15.5 acres of land adjacent to the lake.

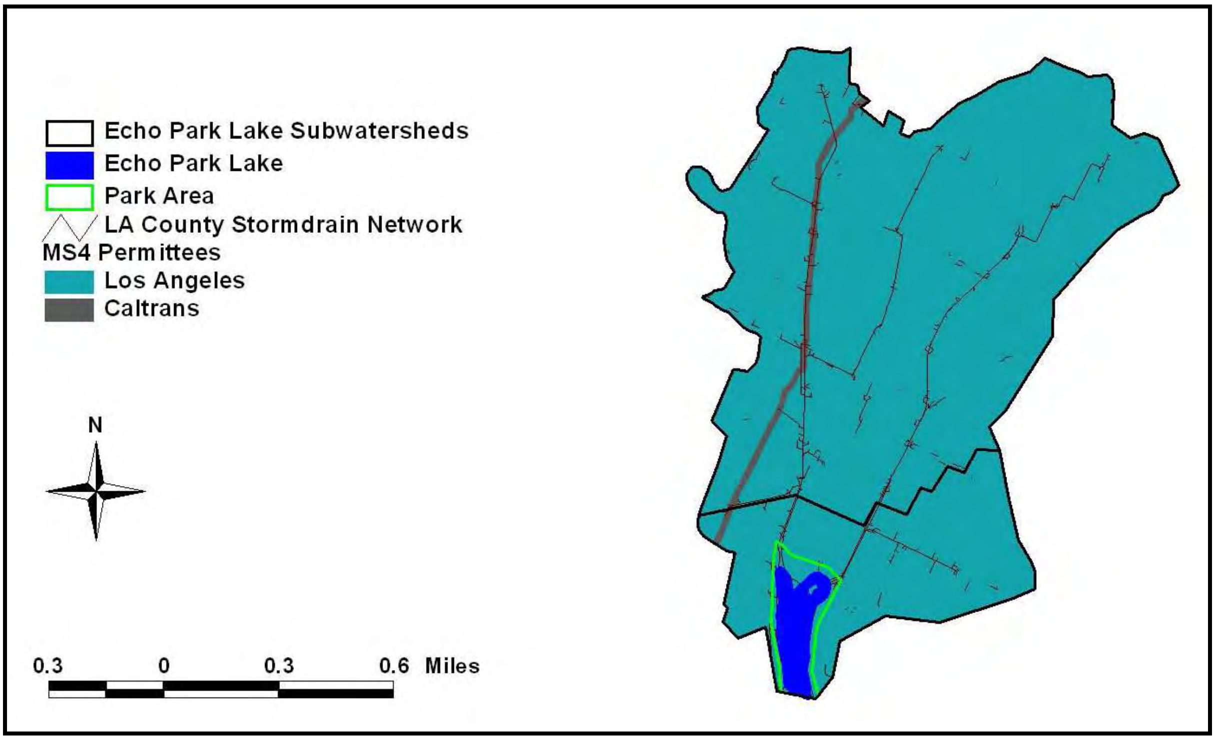

Figure D-18. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Echo Park Lake Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-19. The watershed is comprised primarily of residential development as well as commercial, other urban, industrial, and open space areas. Table D-22 and Table D-23 summarize the land use areas used to estimate pollutant loading from the Northern and Southern subwatersheds, respectively.

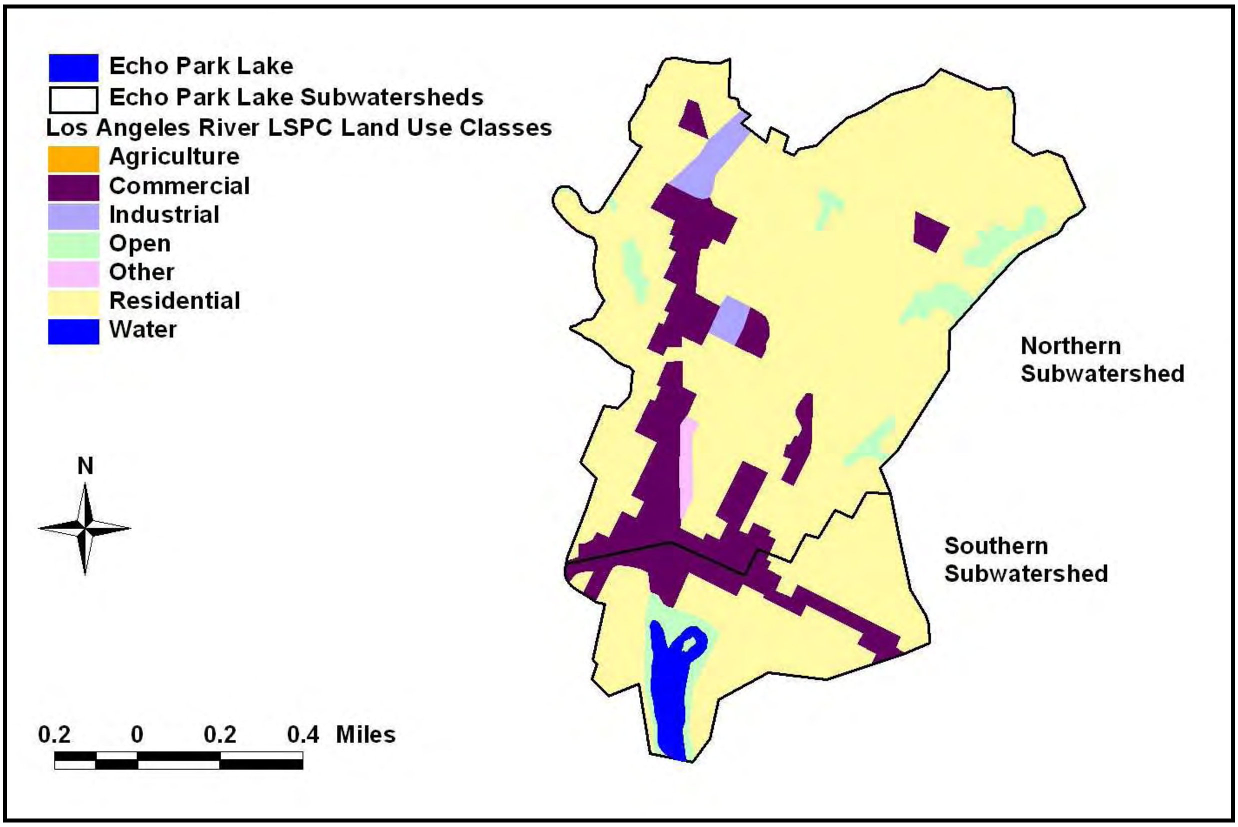

Figure D-19. LSPC Land Use Classes for the Echo Park Lake Subwatersheds

Table D-22. Land Use Areas (ac) Draining to Echo Park Lake from the Northern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	78.4	0	**78.4**
Industrial	12.2	13.0	**25.2**
Open	27.5	0	**27.5**
Other Urban	4.67	0	**4.67**
Residential	479	0	**479**
Total	**601**	**13.0**	**614**

Table D-23. Land Use Areas (ac) Draining to Echo Park Lake from the Southern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	31.6	0	**31.6**
Industrial	0	1.10	**1.10**
Open	15.5	0	**15.5**
Other Urban	0	0	**0**
Residential	122	0	**122**
Total	**169**	**1.10**	**170**

D.6.1 RUNOFF

LSPC-predicted runoff from the Echo Park Lake subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-20 shows the simulated annual rainfall for the Echo Park Lake subwatersheds. The annual average rainfall is 15.2 inches.

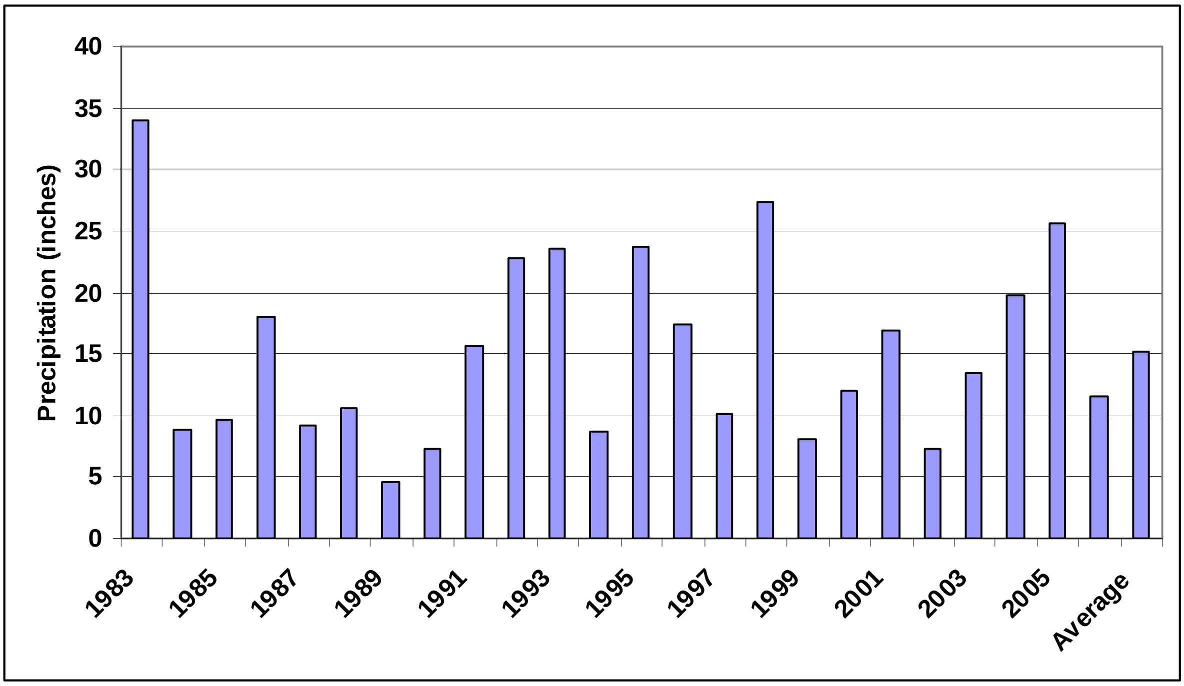

Figure D-20. Annual Rainfall for the Echo Park Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the Echo Park Lake subwatersheds are shown in Table D-24.

Table D-24. Monthly Average Runoff Depths (inches/month) for Land Uses in the Echo Park Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.2843	2.2493	2.0170	0.0963	1.4365	1.3899
February	0.4635	3.0258	2.7225	0.1613	1.9644	1.9036
March	0.3191	1.9875	1.7918	0.1136	1.3028	1.2636
April	0.0826	0.6010	0.5372	0.0334	0.3779	0.3651
May	0.0264	0.1783	0.1602	0.0094	0.1147	0.1111
June	0.0047	0.0537	0.0475	0.0024	0.0322	0.0309
July	0.0004	0.0027	0.0024	0.0002	0.0017	0.0016
August	0.0020	0.0263	0.0232	0.0010	0.0155	0.0149
September	0.0110	0.1532	0.1352	0.0055	0.0903	0.0867
October	0.0272	0.3845	0.3393	0.0136	0.2263	0.2172
November	0.0489	0.6911	0.6098	0.0244	0.4067	0.3904
December	0.0995	1.3697	1.2099	0.0487	0.8103	0.7783

The majority of the runoff from the Echo Park Lake watershed is diverted downstream of the lake and on average, only 16.7 ac-ft/yr are delivered through the storm drain network (personal communication, Charlie Yu, City of Los Angeles, 3/4/2010). The simulated runoff volumes and associated pollutant loading were scaled down by the ratio of delivered flow (16.7 ac-ft/yr) to simulated flow (452 ac-ft/yr) to estimate the amount of loading reaching Echo Park Lake. It was assumed that all runoff (0.6 ac-ft/yr) and associated pollutant loading from the 15.5 acres of park adjacent to the lake were not diverted downstream. Figure D-12 summarizes the monthly average runoff volumes delivered to Echo Park Lake. The total annual volume delivered to the lake is 17.3 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

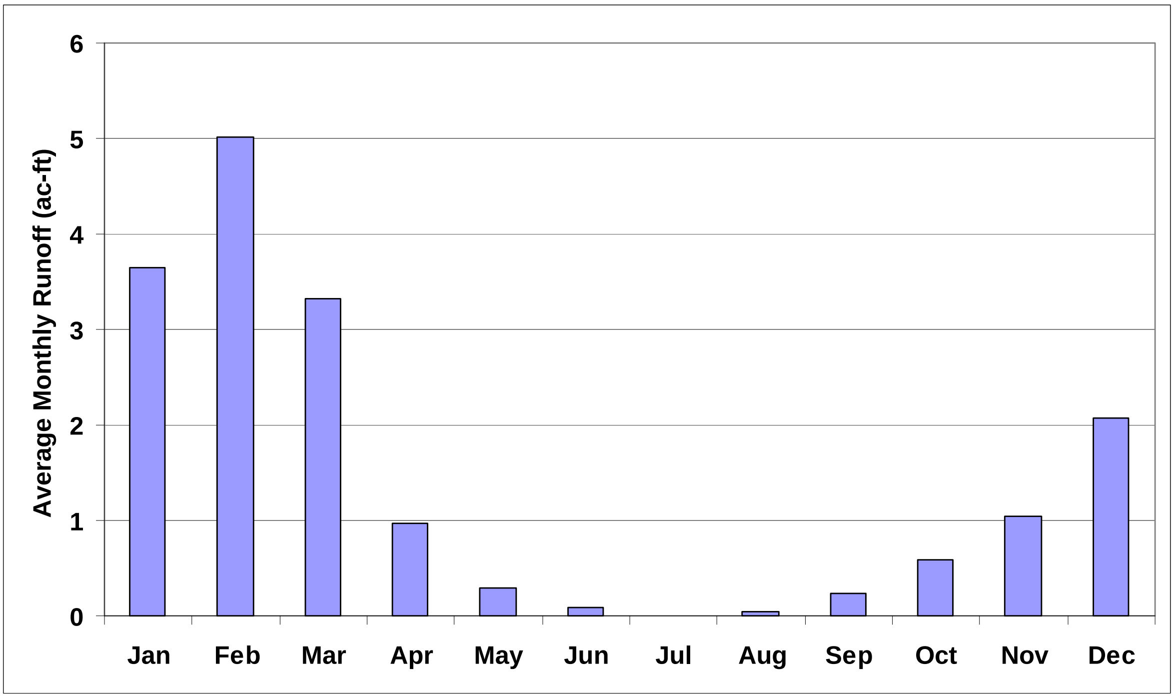

Figure D-21. Monthly Average Runoff Volumes to Echo Park Lake

D.6.2 Sediment Loads

Sediment loads are calculated from delivered runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-25 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-25. Average Annual Sediment Loads to Echo Park Lake

Subwatershed	Jurisdiction	Sediment (tons/yr)
Northern	City of Los Angeles	0.976
Northern	Caltrans	0.044
Southern	City of Los Angeles	0.291
Southern	Caltrans	0.0037
Total		**1.32**

D.6.3 Nutrient Loads

Nutrient loads are estimated from delivered volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-26 summarizes the total nitrogen and total phosphorus loads estimated for Echo Park Lake from each jurisdiction and subwatershed. See example calculations in Section D.3.

Table D-26. Average Annual Nutrient Loads to Echo Park Lake

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Northern	City of Los Angeles	155	24.7
Northern	Caltrans	4.80	0.608
Southern	City of Los Angeles	169	6.99
Southern	Caltrans	1.10	0.051
Total		**209**	**32.3**

D.6.4 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant of concern using monitoring data collected by USEPA, the Regional Board, and UCLA, between 2008 and 2009. Only data from sites representing inflows are used; these sites include locations in an inflow, or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Echo Park Lake, data from the following stations was included: EPL-1, EPL-2, and EPL-12 (Figure D-22).

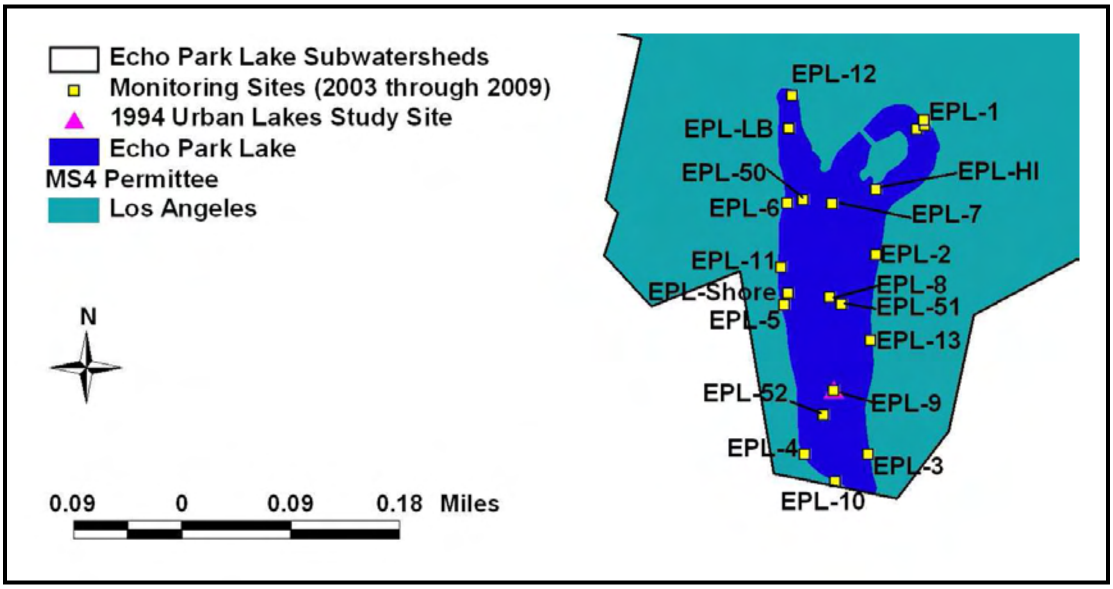

Figure D-22. Echo Park Monitoring Stations

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads were associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For chlordane and PCBs, samples below detection limits were assumed to be one-half of the detection limits. For all of the sediment samples, dieldrin was below detection levels; therefore an inflow concentration could not be determined. Instead, an upper-bound analysis was performed using the detection limit as the incoming concentration associated with the sediment. The inflow sediment data are summarized in Table D-27 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-27. Summary of Sediment Data near Inflow Locations at Echo Park Lake

Parameter	Number of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit Range (µg/kg dry weight)
Chlordane	6	2	8.31	0.44-1.23
Dieldrin	6	0	(1.32)[2]	0.83- 3.00
Total PCBs	6	5	24.16	0.44-1.23

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

[2] All sample results were below detection limits. An upper-bound analysis was performed using the highest reported detection limit for dieldrin.

These input sediment concentrations were applied to the calculated sediment loads (Section D.6.2) to estimate the sediment-associated OC Pesticides and PCBs loads entering the lake. Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the two Echo Park Lake subwatersheds in Table D-28. Dissolved concentrations in inflows are assumed insignificant.

Table D-28. Annual Sediment Load to Echo Park Lake

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Northern	Caltrans	13.0	0.044	3.44%
Northern	City of Los Angeles	601	0.98	75.66%
Southern	Caltrans	1.10	0.0037	0.29%
Southern	City of Los Angeles	169	0.29	20.61%
Total		**784**	**1.32**	**100%**

The chlordane, PCB, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-27) to the calculated sediment load of 1.32 tons per year (Table D-28). See example calculations in Section D.3. Loads for each jurisdiction are shown by subwatershed in Table D-29.

Table D-29. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Echo Park Watershed (g/yr)

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual Dieldrin Load[1]	Percent of Total Load
Northern	Caltrans	0.0010	0.0003	<0.00005	3.44%
Northern	Los Angeles	0.021	0.0074	<0.00117	75.66%
Southern	Caltrans	0.0001	0.00003	<0.00000	0.29%
Southern	Los Angeles	0.0064	0.0022	<0.00035	20.61%
Total		**0.029**	**0.0099**	**<0.0016**	**100%**

[1] Results from upper-bound analysis representing the maximum possible dieldrin load.

D.7 Lake Calabasas

Lake Calabasas is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH. A DDT impairment was previously reported for this lake, but was delisted by the Regional Board in 2009. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate nutrient loads from upland areas, which may be contributing to the low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH impairments.

One subwatershed draining 86.5 acres comprises the drainage area to Lake Calabasas. Figure D-23 shows the MS4 stormwater permittee in the Lake Calabasas watershed. The entire subwatershed is comprised of the city of Calabasas. This subwatershed drains to a storm drain system, so all allocations for the TMDLs are wasteload allocations.

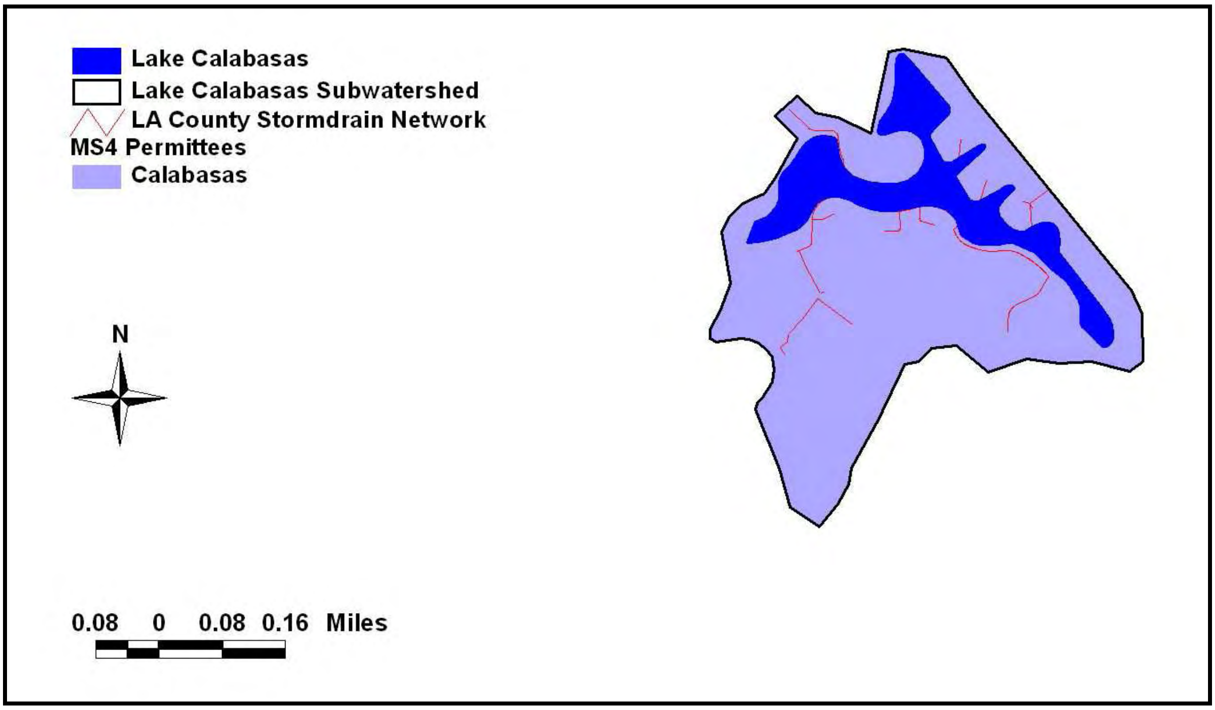

Figure D-23. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lake Calabasas Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-24. The watershed is comprised of residential development and open space. Table D-30 summarizes the land use areas used to estimate pollutant loading from upland areas draining to Lake Calabasas.

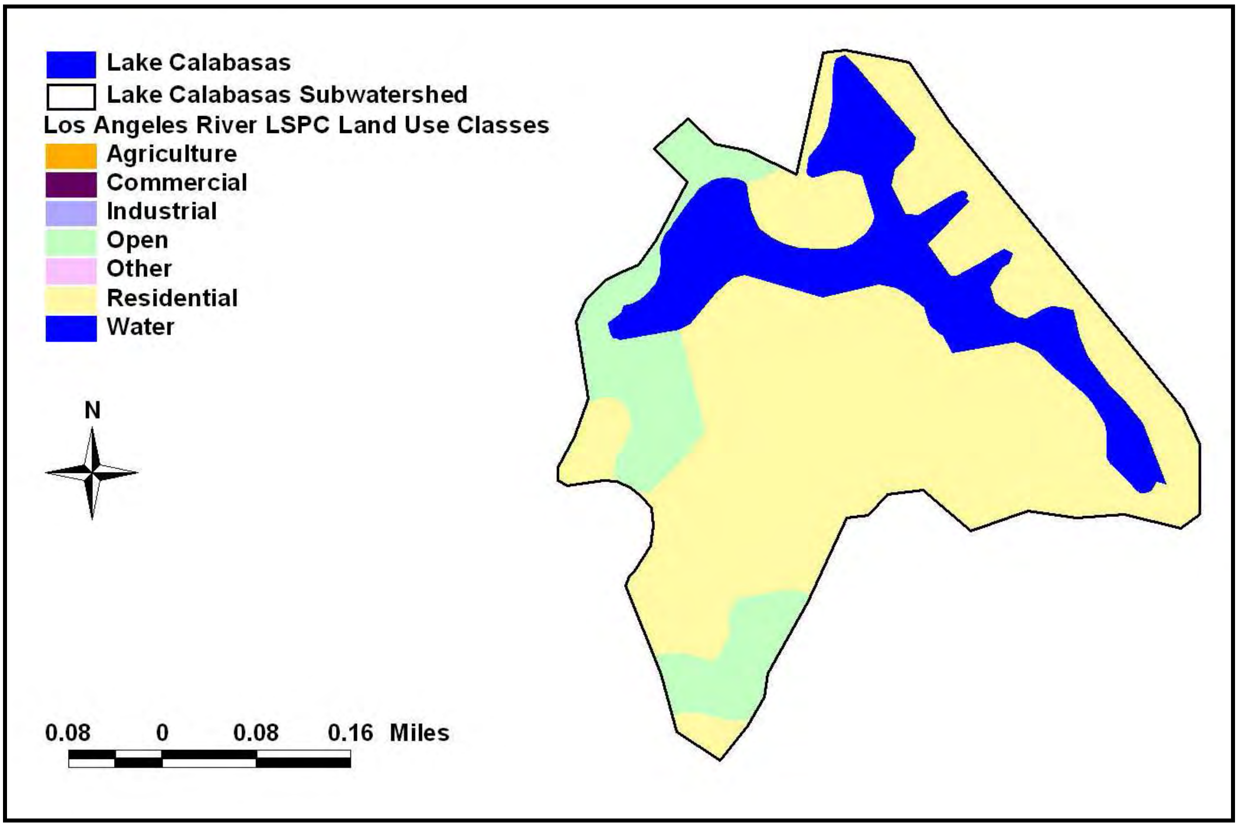

Figure D-24. LSPC Land Use Classes for the Lake Calabasas Subwatershed

Table D-30. Land Use Areas (ac) Draining to Lake Calabasas

Land Use	City of Calabasas
Agriculture	0
Commercial	0
Industrial	0
Open	14.2
Other Urban	0.0
Residential	72.3
Total	**86.5**

D.7.1 RUNOFF

LSPC-predicted runoff from the Lake Calabasas subwatershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-25 shows the simulated annual rainfall for the Lake Calabasas subwatershed. The annual average rainfall is 17.5 inches.

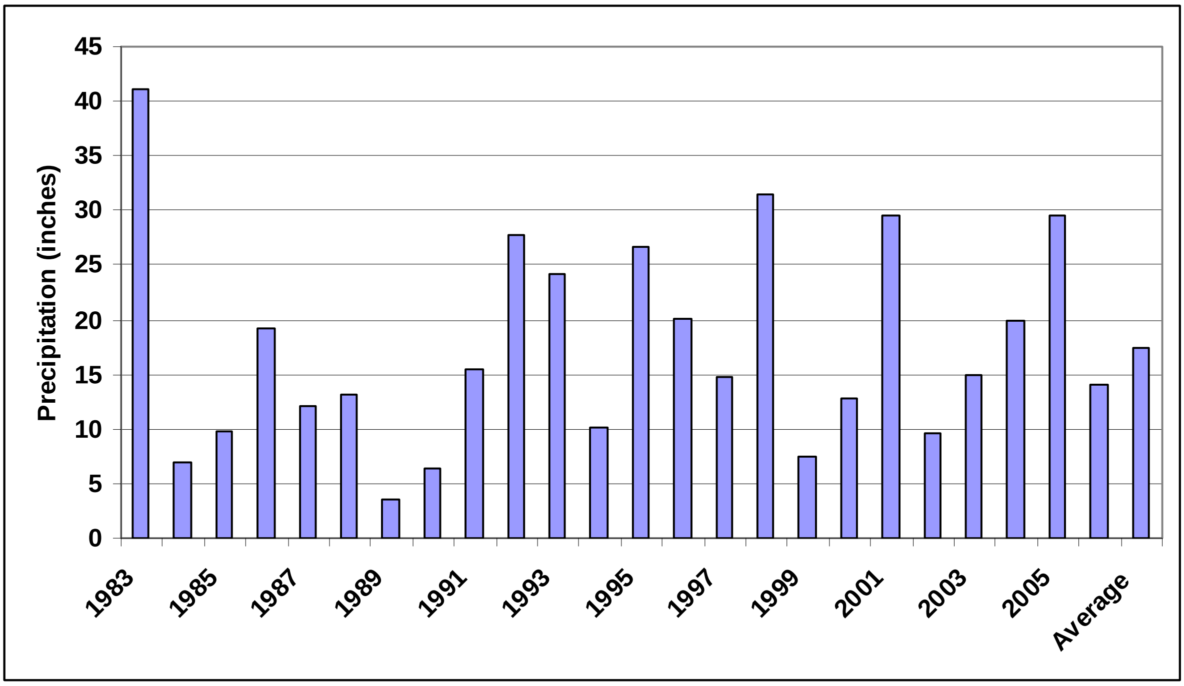

Figure D-25. Annual Rainfall for the Lake Calabasas Subwatershed

The simulated monthly average runoff depths for land uses in the Lake Calabasas subwatershed are shown in Table D-31.

Table D-31. Monthly Average Runoff Depths (inches/month) for Land Uses in the Lake Calabasas Subwatershed, 1983 - 2006

Month	Open	Residential
January	0.1271	1.6687
February	0.3202	2.5495
March	0.2219	1.5042
April	0.0473	0.4536
May	0.0174	0.1452
June	0.0020	0.0134
July	0.0005	0.0023
August	0.0009	0.0116
September	0.0056	0.0878
October	0.0192	0.3065
November	0.0342	0.5464
December	0.0585	0.9309

Figure D-26 summarizes the monthly average runoff volumes delivered to Lake Calabasas. The total annual volume delivered to the lake is 50.6 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

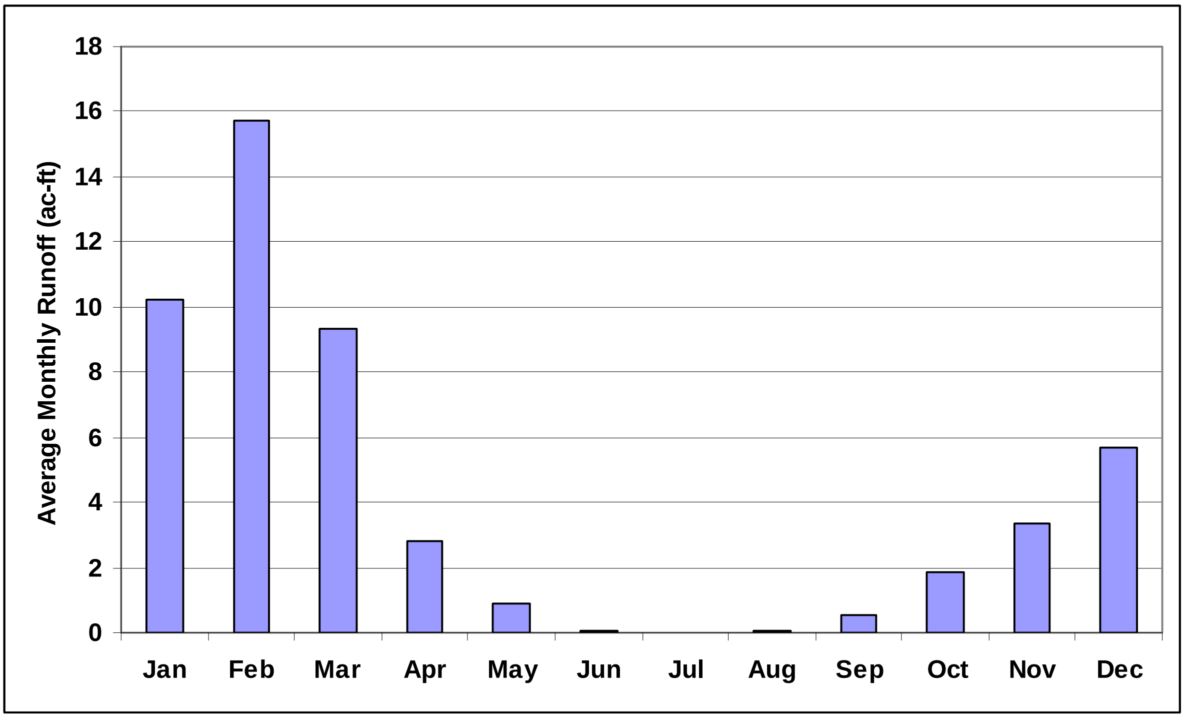

Figure D-26. Monthly Average Runoff Volumes to Lake Calabasas

D.7.2 Nutrient Loads

Nutrient loads are estimated from runoff volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-32 summarizes the total nitrogen and total phosphorus loads delivered to Lake Calabasas. See example calculations in Section D.3.

Table D-32. Average Annual Nutrient Loads to Lake Calabasas

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Calabasas	Calabasas	616	98.7

D.8 El Dorado Park Lakes

The El Dorado Park lakes are located in the San Gabriel River Basin. Six lakes are located in the park. The northern four lakes are hydraulically connected and separate from the system comprised by the two southern lakes, also hydraulically connected. These lakes are listed as impaired by algae, ammonia, eutrophication, pH, copper, lead, and mercury. Output from the San Gabriel River LSPC model, coupled with regional pollutant event mean concentrations, has been used to estimate loads of nutrients, which may be contributing to the algae, ammonia, eutrophication, and pH impairments. LSPC model output and monitoring data collected in 2009 are used to estimate mercury loading.

Two separate watersheds have been delineated for these separate lake systems. The subwatershed draining to the northern four lakes is comprised of 185 acres, and the subwatershed draining to the southern two lakes is comprised of 33.8 acres.

Figure D-27 shows the MS4 stormwater permittee that comprises both the northern and southern drainages of the El Dorado Park lake systems as well as the Los Angeles County storm drain network. Though both watersheds are in the city of Long Beach incorporated area, there are no major drains that divert runoff directly to the lake: a few small culverts pass water beneath walking paths and park roads. Because both watersheds are comprised solely of parklands that do not drain to a major storm drain system, the watershed loads to the El Dorado Park lakes are assigned load allocations in the TMDLs.

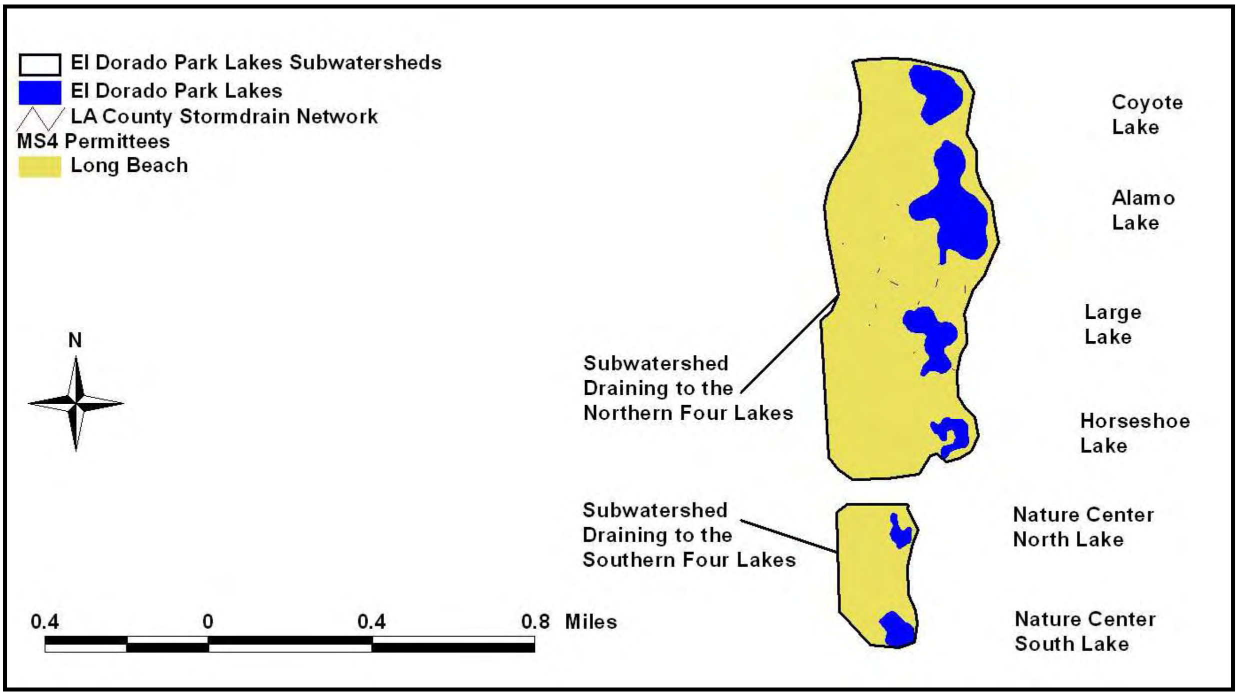

Figure D-27. MS4 Permittee and the County of Los Angeles Storm Drain Network in the El Dorado Park Lake Subwatersheds

Both subwatersheds are comprised of land classified by the San Gabriel LSPC model as "other urban or built-up" except for the two polygons classified as water (Figure D-28). To improve accuracy in land use areas, the SCAG 2005 database was used to estimate the area of the lakes in each subwatershed. Runoff loads from the lakes are assumed zero. All remaining areas in each subwatershed were assumed other urban or built-up (185 acres of the northern subwatershed and 33.8 acres in the southern subwatershed).

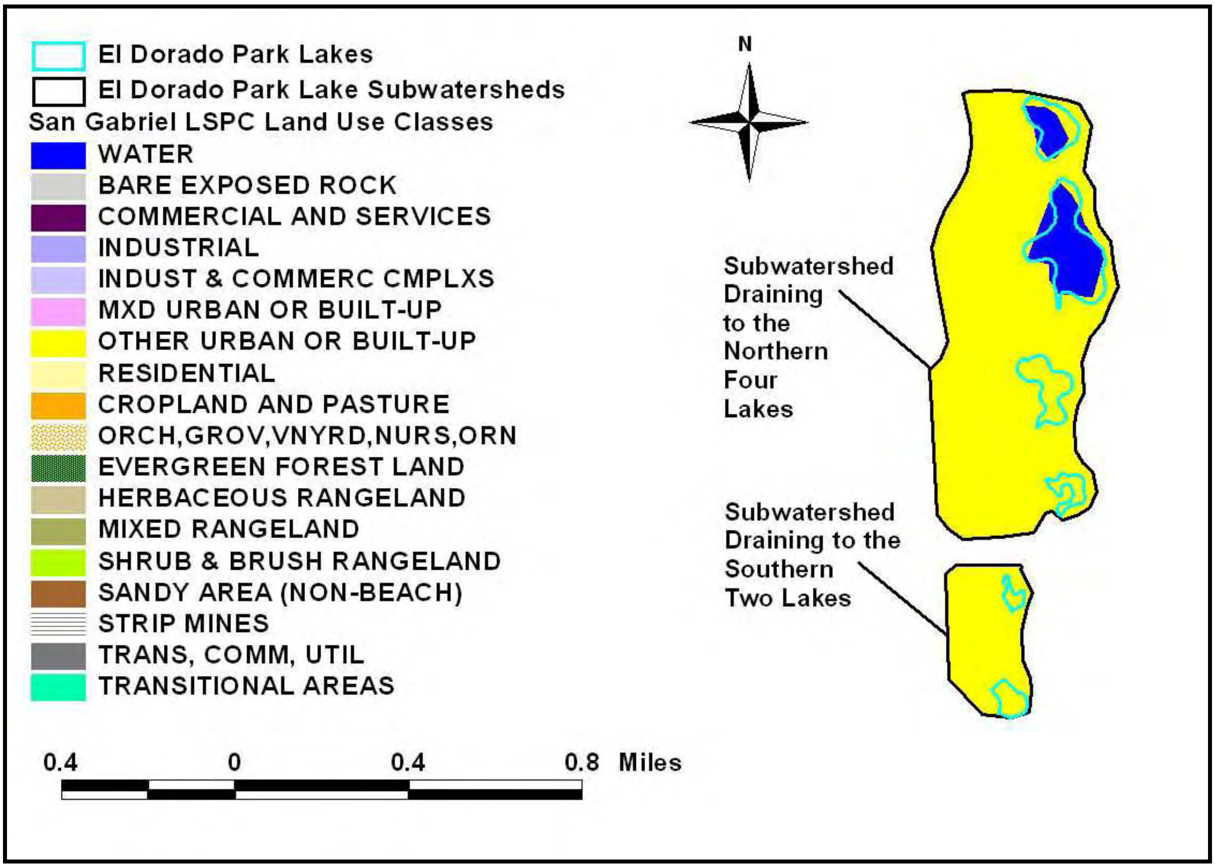

Figure D-28. LSPC Land Use Classes for the El Dorado Park Lakes Subwatersheds

The San Gabriel LSPC Model aggregated the identified land uses into modeled land uses. In the original model, lands classified as “other urban or built-up” were modeled as commercial areas, which is reasonable at the larger basin scale for which the model was developed. Comparison to the SCAG 2005 dataset and current satellite imagery indicate that these areas around the El Dorado Park lakes are actually parkland. To simulate pollutant loading from these areas, LSPC output for pervious commercial areas was assumed representative of park areas. Table D-33 summarizes the areas draining to the El Dorado Park lakes.

Table D-33. Areas Draining to the El Dorado Park Lakes

Land Use	Long Beach – Northern Lake System	Long Beach – Southern Lake System
Other urban or built-up (parkland)	185	33.8

D.8.1 RUNOFF

LSPC-predicted runoff from the El Dorado Park lakes subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-29 shows the simulated annual rainfall for the El Dorado Park lakes subwatersheds. The annual average rainfall is 11.7 inches.

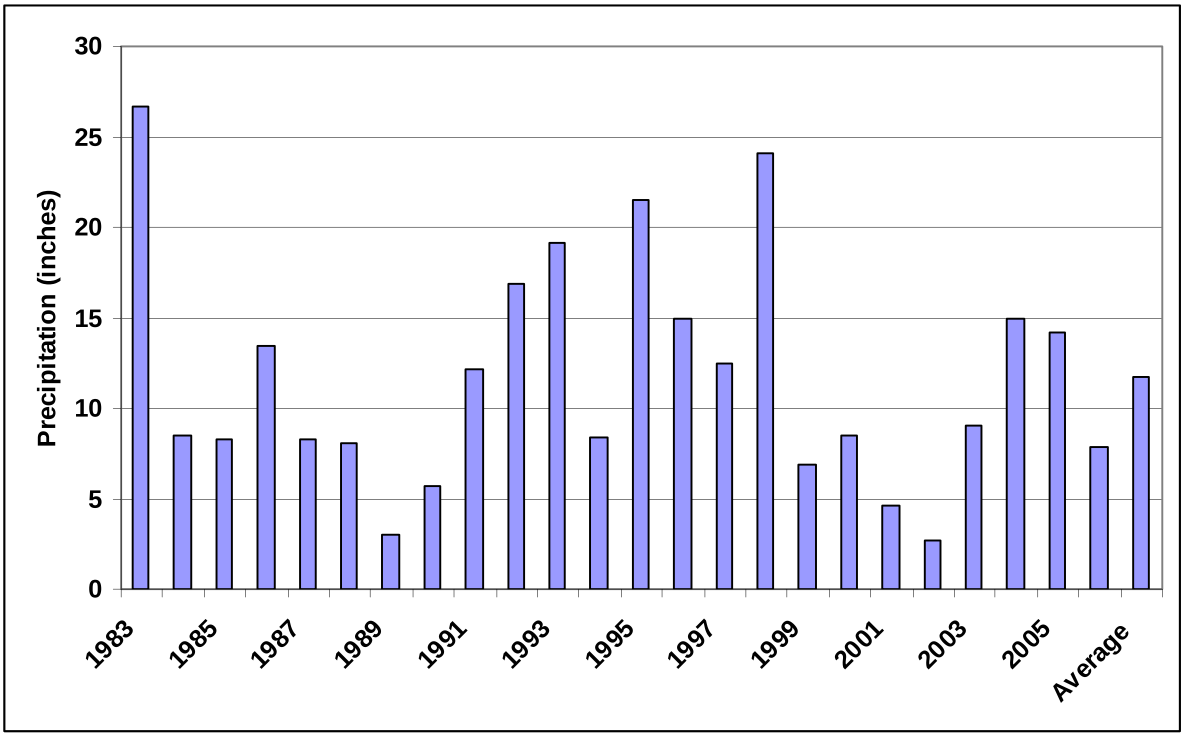

Figure D-29. Annual Rainfall for the El Dorado Park Lakes Subwatersheds

The simulated monthly average runoff for parkland (commercial pervious) areas in the El Dorado Park lakes subwatersheds is shown in Table D-34.

Table D-34. Monthly Average Runoff Depths (inches/month) for Land Uses in the El Dorado Park Lakes Subwatersheds, 1983 - 2006

Month	Runoff from Parkland
January	0.0154
February	0.0268
March	0.0345
April	0.0206
May	0.0069
June	0.0018
July	0.0005
August	0.0002
September	0.0001
October	0.0003
November	0.0007
December	0.0020

Table D-35 summarizes the monthly average runoff volumes from each subwatershed draining to the El Dorado Park lakes.

Table D-35. Monthly Average Runoff Volumes (ac-ft/month) from the El Dorado Park Lakes Subwatersheds

Subwatershed:	Northern	Southern
Month — Land Use:	**Parkland**	**Parkland**
January	0.2377	0.0435
February	0.4130	0.0755
March	0.5308	0.0971
April	0.3167	0.0579
May	0.1060	0.0194
June	0.0277	0.0051
July	0.0071	0.0013
August	0.0025	0.0005
September	0.0013	0.0002
October	0.0051	0.0009
November	0.0106	0.0019
December	0.0311	0.0057
Annual Volume (ac-ft/yr)	**1.69**	**0.309**

D.8.2 SEDIMENT LOADS

Sediment loads from each subwatershed are based on simulated runoff volumes and suspended sediment event mean concentrations. The assumed suspended sediment event mean concentration for the LSPC model for other urban areas is 56.5 mg/L (Table D-7). Table D-36 summarizes the monthly sediment loads from each subwatershed in El Dorado Park. See example calculations in Section D.3.

Table D-36. Monthly Average Sediment Loads (lbs) from the El Dorado Park Lakes Subwatersheds

Month	Northern Lake System	Southern Lake System
January	36.5	6.68
February	63.5	11.6
March	81.6	14.9
April	48.7	8.90
May	16.3	2.98
June	4.26	0.778
July	1.09	0.199

Month	Northern Lake System	Southern Lake System
August	0.382	0.070
September	0.195	0.036
October	0.787	0.144
November	1.63	0.299
December	4.79	0.875
Annual Load (lb/yr)	**259.6**	**47.5**

D.8.3 NUTRIENT LOADS

Nutrient loads are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles. For "other urban" land uses, the total nitrogen event mean concentration is 4.41 mg-N/L and the total phosphorus event mean concentration is 0.67 mg-P/L (Table D-7). Table D-37 and Table D-38 summarize the total nitrogen and total phosphorus loads, respectively, from each subwatershed draining to the El Dorado Park lakes. See example calculations in Section D.3.

Table D-37. Monthly Average Nitrogen Loads (pounds) from the El Dorado Park Lakes Subwatersheds

Month	Northern Lake System	Southern Lake System
January	2.85	0.521
February	4.95	0.906
March	6.37	1.16
April	3.80	0.695
May	1.27	0.232
June	0.332	0.061
July	0.085	0.015
August	0.030	0.005
September	0.015	0.003
October	0.061	0.011
November	0.127	0.023
December	0.374	0.068
Annual Load (lb/yr)	**20.26**	**3.71**

Table D-38. Monthly Average Phosphorus Loads (lbs) from the El Dorado Park Lakes Subwatersheds

Month	Northern Lake System	Southern Lake System
January	0.433	0.0792
February	0.752	0.138
March	0.967	0.177
April	0.577	0.106
May	0.193	0.0353
June	0.0505	0.0092
July	0.0129	0.0024
August	0.0045	0.0008
September	0.0023	0.0004
October	0.0093	0.0017
November	0.0194	0.0035
December	0.0567	0.0104
Annual Load (lb/yr)	**3.08**	**0.563**

D.8.4 MERCURY LOADS

Mercury loads from each subwatershed are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Mercury loading is associated with both sediment and runoff from upland areas. To determine sediment loading of mercury, the sediment EMCs and runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data were then applied to the sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations to simulated runoff volumes (Section D.3 provides examples of these calculations). However, during both the February and July 2009 sampling events, the only visible inputs to the El Dorado Park lakes were the groundwater input to Coyote Lake and the potable water input to Nature Center North Lake. Loads associated with these inputs are discussed in Appendix F (Dry Weather Loading).

To estimate loading associated with wet weather events, concentrations measured from culverts and tributaries around Puddingstone Reservoir in the southern subwatershed (Section D.10.4) were assumed representative of concentrations for the El Dorado Park lakes. Puddingstone Reservoir is located in Bonelli Regional Park and the land uses surrounding the reservoir are similar to those in El Dorado Park. Table D-39 and Table D-40 present the assumed concentrations and resulting loads for total mercury and methylmercury, respectively. Example calculations are presented in Section D.3.

Table D-39. Total Mercury Loads Estimated for Each Subwatershed in El Dorado Park

Sub-watershed	Juris-diction	Area (ac)	Summer Water Column Hg (ng/L)[1]	Winter Water Column Hg (ng/L)[1]	Summer Sediment Hg (µg/kg)[2]	Winter Sediment Hg (µg/kg)[2]	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Northern Lake System	Long Beach	185	7.55	2.65	50.3	36.4	0.00643	0.00443	0.0109
Southern Lake System	Long Beach	33.8	7.55	2.65	50.3	36.4	0.00118	0.000810	0.00199

[1] Concentrations are based on observations around Puddingstone Reservoir (Table D-50).

[2] Concentrations are based on observations around Puddingstone Reservoir (Table D-51).

Table D-40. Methylmercury Loads Estimated for Each Subwatershed in El Dorado Park

Sub-watershed	Juris-diction	Area (ac)	Summer Water Column MeHg (ng/L)[1]	Winter Water Column MeHg (ng/L)[1]	Summer Sediment MeHg (µg/kg)[2]	Winter Sediment MeHg (µg/kg)[2]	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Northern Lake System	Long Beach	185	0.046	0.010	0.716	0.002	2.75E-05	7.68E-06	3.52E-05
Southern Lake System	Long Beach	33.8	0.046	0.010	0.716	0.002	5.03E-06	1.40E-06	6.43E-06

[1] Concentrations are based on observations around Puddingstone Reservoir (Table D-50).

[2] Concentrations are based on observations around Puddingstone Reservoir (Table D-51).

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D.9 North, Center, and Legg Lakes

North, Center, and Legg lakes are hydraulically connected waterbodies in Whittier Narrows Regional Park located in the Los Angeles River Basin. Legg Lake is listed as impaired by odor, ammonia, pH, copper, and lead (note: trash impairment has been addressed by a previous TMDL). Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate existing loading rates from upland areas of nutrients, which may be contributing to the odor, ammonia, and pH impairments.

Five subwatersheds comprise the drainage area to these lakes. The northwestern and northeastern subwatersheds each drain to a storm drain that enters North Lake on the north side. Three separate drainage areas have been delineated around the lakes to designate respective overland flow.

The northwestern, northeastern, and direct to North Lake subwatersheds flow into North Lake which is basically separate from Center and Legg lakes during dry periods; North Lake discharges to Morris Creek. Legg Lake receives inputs from the direct to Legg Lake subwatershed, from a Superfund site that discharges treated groundwater to the lake, and from pumped groundwater that is split between North and Center lakes to maintain water levels. Legg Lake drains into Center Lake via a connecting channel which then discharges to Morris Creek. There are two culverts connecting Center and North lakes that allow water to flow between them when levels are sufficiently high.

Figure D-34 shows the MS4 stormwater permittees in the North, Center, and Legg lakes watershed. Loads generated from El Monte, South El Monte, the county of Los Angeles, and Caltrans from either the northwestern or northeastern subwatersheds are assigned wasteload allocations in the TMDLs because they drain to the storm drain network. Loads generated by South El Monte or the county of Los Angeles areas in the direct drainage subwatersheds are assigned load allocations; Caltrans areas in these subwatersheds are assigned wasteload allocations.

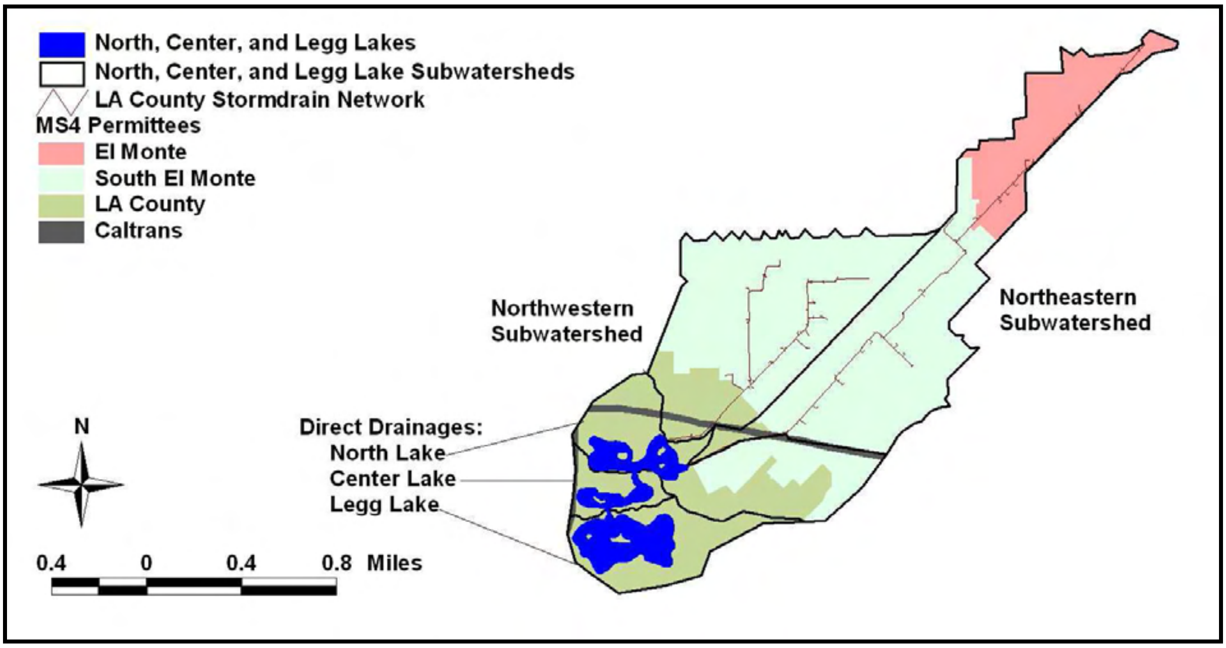

Figure D-30. MS4 Permittees and the County of Los Angeles Storm Drain Network in the North, Center, and Legg Subwatersheds

Land uses identified in the Los Angeles River LSPC model for these subwatersheds are shown in Figure D-35. Tetra Tech reviewed the SCAG 2005 database and current satellite imagery to confirm the acreage of agricultural areas present in the LSPC model. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the following changes were made to maintain consistency with the SCAG 2005 land use database: in the direct drainage subwatershed to Legg Lake, approximately half of the agricultural area was modified as it is actually parkland, and the agricultural areas assigned in the direct to North Lake and north-eastern subwatersheds were changed to vacant land. In addition, the agricultural area present in the northwestern subwatershed is classified by SCAG 2005 as nurseries; however, this area was reclassified to parkland as current satellite imagery shows this area to be Shiveley Park. For the purposes of estimating flows and pollutant loads to this lake system, all agricultural areas were re-assigned as open space, with the exception of 1.04 acres located in the direct to Legg Lake subwatershed. The area classified as “other” is a high school according to SCAG 2005. Table D-41 and Table D-42 summarize the land use types present in the northern two subwatersheds and direct drainage subwatersheds, respectively.

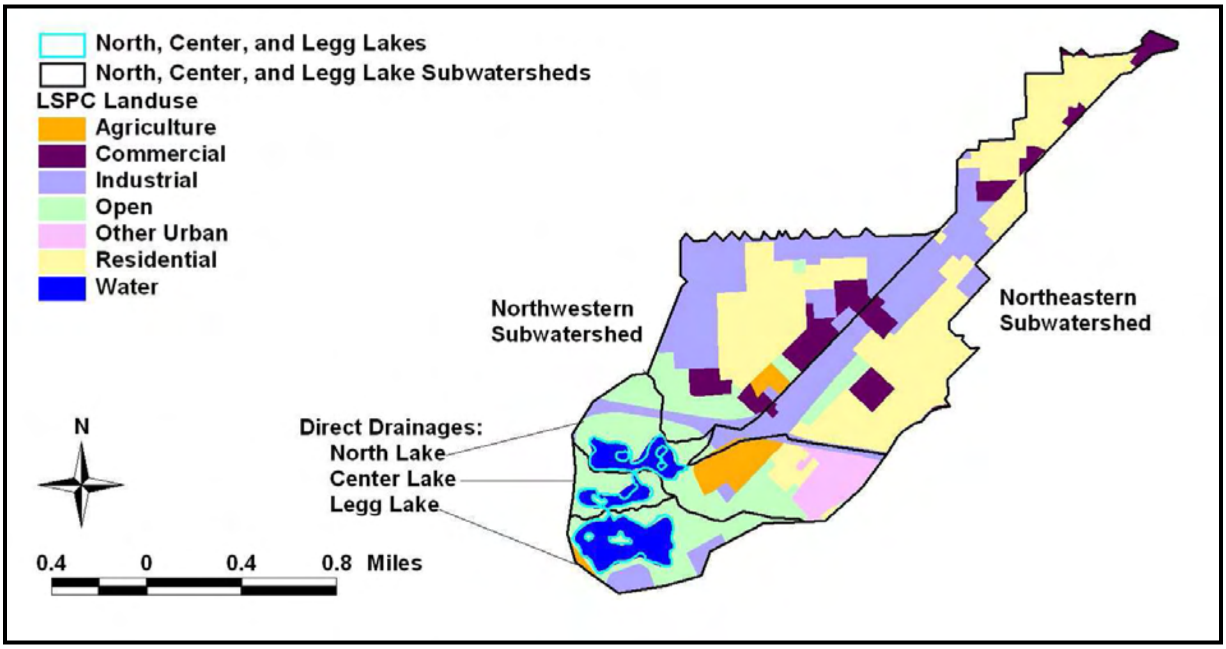

Figure D-31. LSPC Land Use Classes for the North, Center, and Legg Lake Subwatersheds

Table D-41. Land Use Areas (ac) Draining from the Northern Subwatersheds to North, Center, and Legg Lakes

Land Use	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	0	0	0	**0**
Commercial	23.5	58.0	11.9	0	**93.5**
Industrial	6.49	269	13.4	11.5	**300**
Open	0	29.3	44.6	0	**73.9**
Other Urban	0	0.0	0	0	**0**
Residential	104	267	0.271	0	**371**
Total	**134**	**623**	**70.2**	**11.5**	**838**

Table D-42. Land Use Areas (ac) Draining from the Direct Drainage Subwatersheds to North, Center, and Legg Lakes

Land Use	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	1.04	0	**1.04**
Commercial	0	0	0	**0**
Industrial	1.78	24.1	17.6	**43.4**
Open	29.8	202	0	**232**
Other Urban	28.2	12.1	0	**40.3**
Residential	15.8	1.19	0	**17.0**
Total	**75.7**	**240**	**17.6**	**334**

D.9.1 RUNOFF

LSPC-predicted runoff is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-32 shows the simulated annual rainfall for the North, Center, and Legg lakes subwatersheds. The annual average rainfall is 16.5 inches.

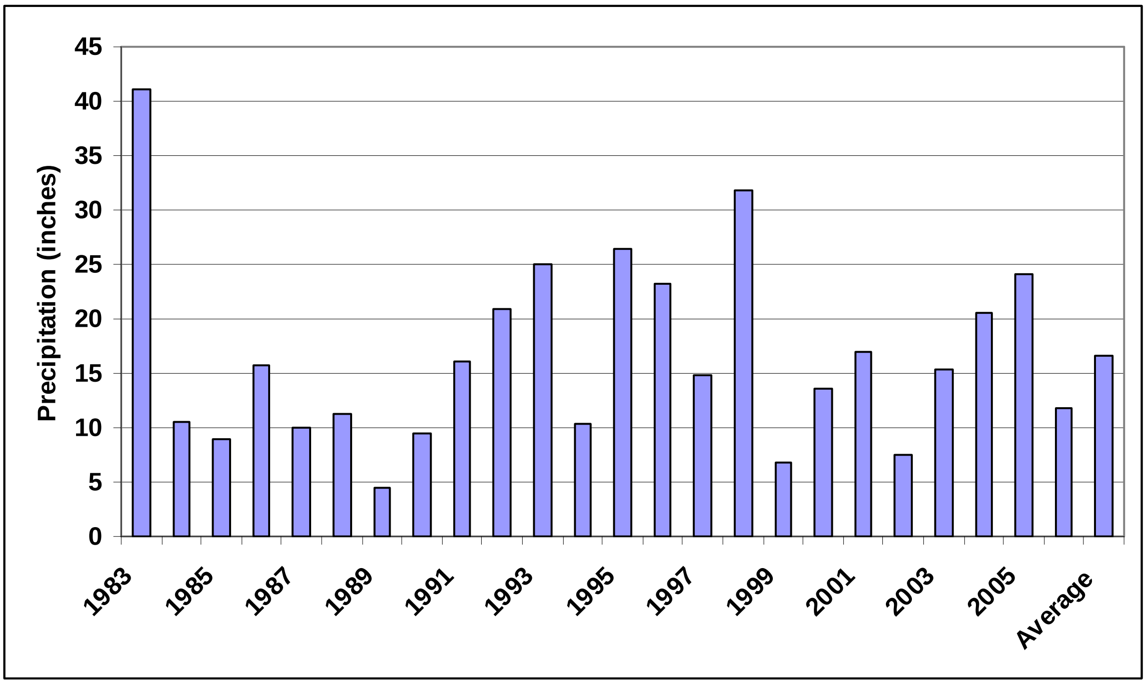

Figure D-32. Annual Rainfall for the North, Center, and Legg Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the North, Center, and Legg lakes subwatersheds are shown in Table D-43.

Table D-43. Monthly Average Runoff Depths (inches/month) for Land Uses in the North, Center, and Legg Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.3332	2.5316	2.2740	0.1144	1.6302	1.5787
February	0.5856	3.2297	2.9171	0.2374	2.1356	2.0731
March	0.3748	2.2020	1.9888	0.1469	1.4557	1.4130
April	0.0825	0.6283	0.5613	0.0372	0.3938	0.3804
May	0.0467	0.2536	0.2289	0.0179	0.1672	0.1622
June	0.0073	0.0825	0.0731	0.0038	0.0494	0.0475
July	0.0002	0.0004	0.0004	0.0001	0.0003	0.0003
August	0.0067	0.0922	0.0814	0.0033	0.0544	0.0522
September	0.0132	0.1843	0.1627	0.0066	0.1086	0.1042
October	0.0378	0.5315	0.4691	0.0188	0.3133	0.3008
November	0.0533	0.7505	0.6623	0.0265	0.4418	0.4242
December	0.1103	1.4977	1.3232	0.0534	0.8870	0.8521

Figure D-33 summarizes the monthly average runoff volumes from each subwatershed from 1983 through 2006. The total annual volume estimated for the lakes is 682 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

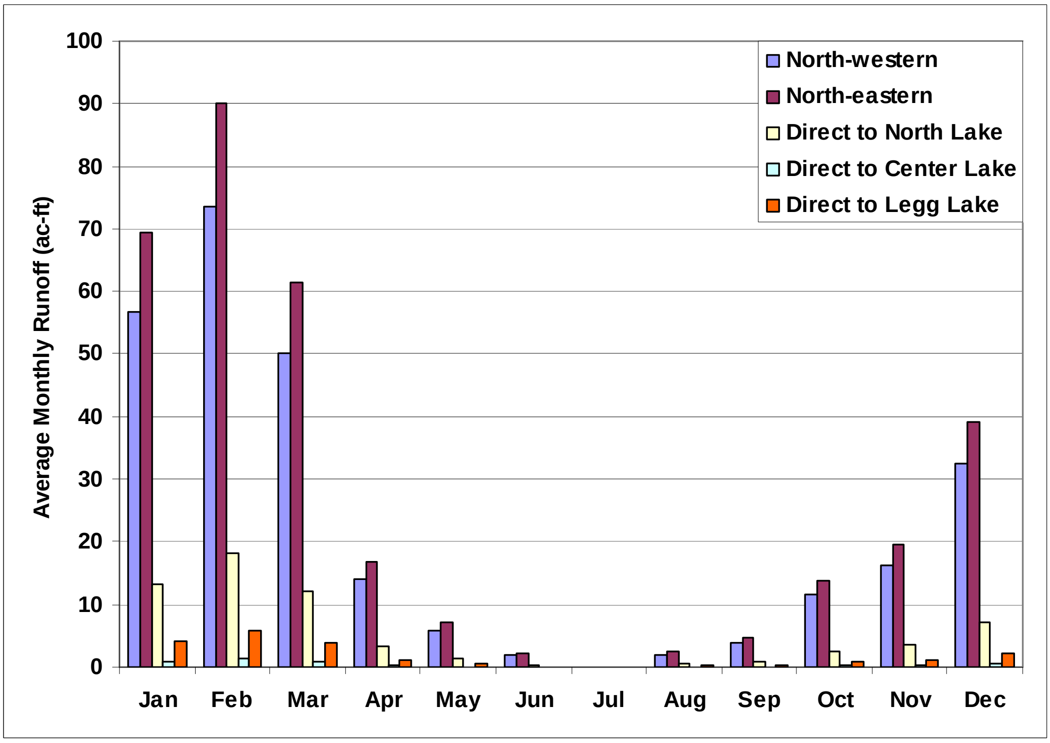

Figure D-33. Monthly Average Runoff Volumes to North, Center, and Legg Lakes (1983-2006)

D.9.2 NUTRIENT LOADS

Nutrient loads are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-44 summarizes the total nitrogen and total phosphorus loads delivered from each subwatershed and jurisdiction contributing to the Legg Lake system. See example calculations in Section D.3.

Table D-44. Average Annual Nutrient Loads to North, Center, or Legg Lakes

Subwatershed	Jurisdiction	Area (ac)	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Direct to Center Lake	Caltrans	3.26	36.1	4.60
Direct to Center Lake	County of Los Angeles	30.4	14.7	0.505
Direct to Legg Lake	Caltrans	0.837	9.28	1.18
Direct to Legg Lake	County of Los Angeles	83.1	228	26.0
Direct to North Lake	Caltrans	13.5	149	19.1
Direct to North Lake	County of Los Angeles	127	226	26.6
Direct to North Lake	South El Monte	75.7	369	55.1
Northwestern	Caltrans	5.32	58.9	7.51
Northwestern	County of Los Angeles	60.1	241	32.4
Northwestern	South El Monte	317	2,982	420
Northeastern	Caltrans	6.18	68.5	8.73
Northeastern	El Monte	134	1,140	179
Northeastern	County of Los Angeles	10.0	73.7	9.24
Northeastern	South El Monte	305	2,716	391
Total		**1,172**	**8,313**	**1,182**

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D.10 Puddingstone Reservoir

Puddingstone Reservoir is located in the San Gabriel River Basin. Impairments include low dissolved oxygen/organic enrichment, mercury, chlordane, DDT, dieldrin, and PCBs. Output from the San Gabriel River LSPC model coupled with regional pollutant event mean concentrations has been used to estimate loads of nutrients from upland areas, which may be contributing to the low dissolved oxygen/organic enrichment impairment. LSPC model output and monitoring data collected in 2009 are used to estimate mercury and OC Pesticides and PCBs loading.

Two subwatersheds comprise the drainage area to Puddingstone Reservoir. The subwatershed draining the northern part of the watershed is 6,959 acres, and the southern subwatershed is 1,169 acres. The subwatershed boundaries were chosen to separate those areas that drain to a storm drain (the northern subwatershed) and those that enter the reservoir via natural tributaries or overland flow (the southern subwatershed).

Figure D-34 shows the MS4 stormwater permittees in the Puddingstone Reservoir watershed. The northern subwatershed is primarily comprised of the county of Los Angeles, Claremont, and La Verne areas with a small amount of San Dimas, Caltrans, and Angeles National Forest areas. Loads generated from these jurisdictions in the northern subwatershed were assigned wasteload allocations because they drain to the Los Angeles County storm drain network. The southern subwatershed is comprised of San Dimas, La Verne, and Pomona areas. Loads from these jurisdictions originating in the southern subwatershed were assigned load allocations. The small amount of Caltrans area in the southern subwatershed were assigned a wasteload allocation.

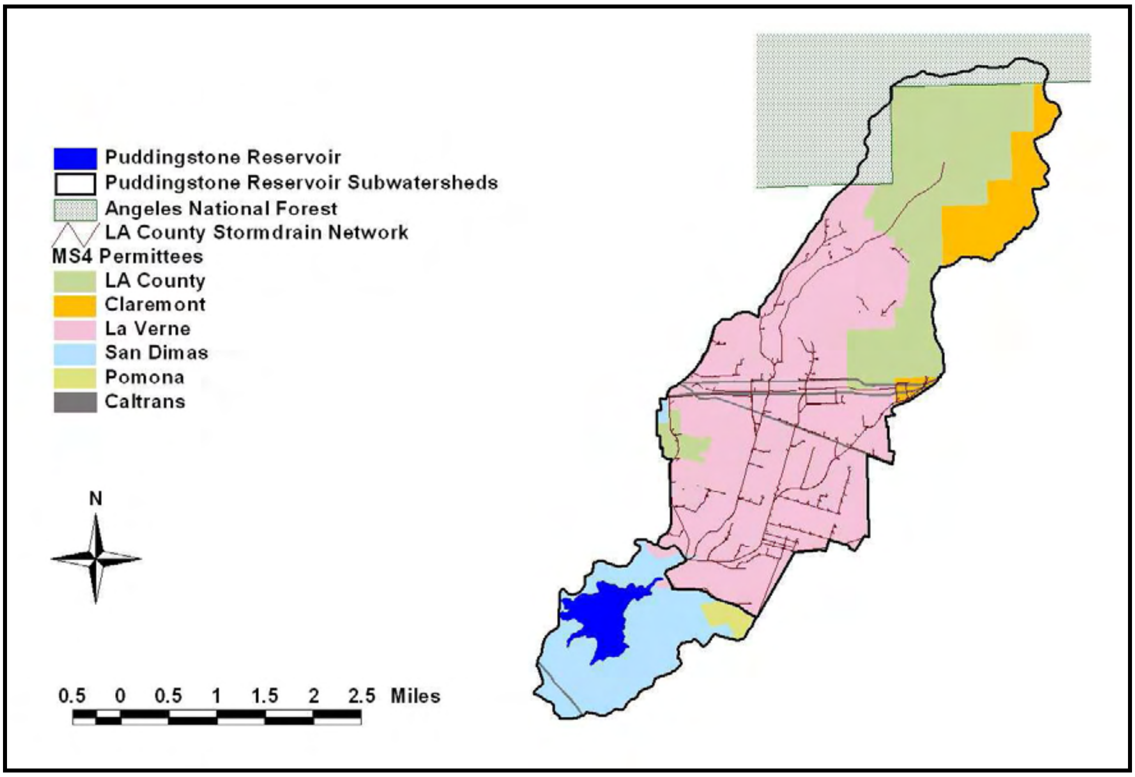

Figure D-34. MS4 Permittees and the Los Angeles County Storm Drain Network in the Puddingstone Reservoir Subwatersheds

Land uses identified in the San Gabriel River LSPC model are shown in Figure D-35. Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that some of the areas classified by the LSPC model as agriculture or strip mines were inaccurate. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the strip mine area in the northern basin (271 ac) was modified as it is currently in residential development; a portion of the agricultural lands in the watershed were changed to either residential or mixed rangeland; and the reservoir identified in the northern basin is a flood control structure that is essentially vacant land based on the aerial, so this area was assigned to mixed rangeland. The “other urban or built-up” areas in the southern subwatershed were reclassified because review of aerial imagery indicates that these areas are currently parkland surrounding the reservoir; therefore, they were simulated as commercial areas with zero percent imperviousness (see discussion in Section D.3). Inaccuracies in land use assignment were corrected for each subwatershed and jurisdiction to reflect the more recent SCAG 2005 dataset and current satellite imagery. All areas within the Caltrans jurisdiction were simulated as transportation. Table D-45 and Table D-46 summarize the land use areas used to estimate pollutant loading from upland areas draining to Puddingstone Reservoir.

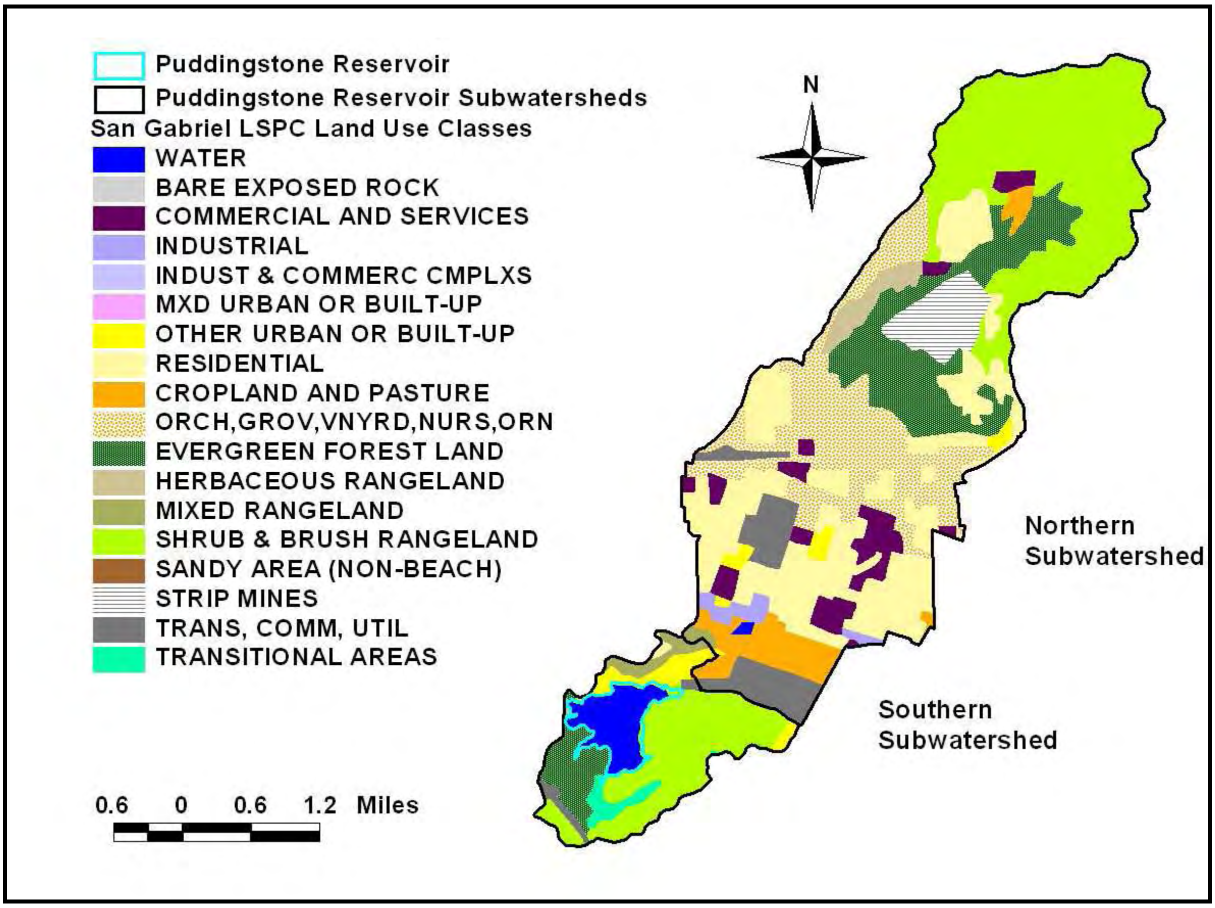

Figure D-35. LSPC Land Use Classes for the Puddingstone Reservoir Subwatersheds

Table D-45. Land Use Areas (ac) Draining from the Northern Subwatershed of Puddingstone Reservoir

Land Use	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Commercial and services	0	38.8	295	0.291	11.0	0	**0**	**345**
Cropland and pasture	2.91	22.5	199	0	0	0	**0**	**225**
Evergreen forest land	42.9	378	376	0	0	0	**0**	**797**
Herbaceous rangeland	0	0	123	0	0	0	**0**	**123**
Industrial	0	0	82.3	0	0	0	**0**	**82.3**
Mixed rangeland	0	21.5	111	1.08	1.95	0	**0**	**135**
Other urban or built-up	8.07	9.24	58.2	0.005	2.90	0	**0**	**78.4**
Residential	28.4	467	2,469	0.260	10.0	0	**0**	**2,975**
Shrub & brush rangeland	496	926	19.7	0.097	0.53	0	**293**	**1,736**
Transportation, communications, utilities	0	0.97	346	3.55	2.12	110	**0**	**463**
Transitional areas	0	0	0	0	0	0	**0**	**0**
Total	**578**	**1,865**	**4,079**	**5.28**	**28.5**	**110**	**293**	**6,959**

Table D-46. Land Use Areas (ac) Draining from the Southern Subwatershed of Puddingstone Reservoir

Land Use	La Verne	Pomona	San Dimas	Caltrans	Total
Commercial and services	0	0	0	0	**0**
Cropland and pasture	0	0	0	0	**0**
Evergreen forest land	0	0	184	0	**184**
Herbaceous rangeland	0	0	4.33	0	**4.33**
Industrial	0	0	0	0	**0**
Mixed rangeland	23.7	0	48.5	0	**72.2**
Other urban or built-up	1.35	19.1	101	0	**122**
Residential	0	0	10.7	0	**10.7**
Shrub & brush rangeland	0.006	62.1	602	0	**664**
Transportation, communications, utilities	8.44	0.616	23.0	11.6	**43.6**
Transitional areas	0	0	68.2	0	**68.2**
Total	**33.5**	**81.8**	**1,042**	**11.6**	**1,169**

D.10.1 RUNOFF

LSPC-predicted runoff from the Puddingstone Reservoir subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-36 shows the simulated annual rainfall for the Puddingstone Reservoir subwatersheds. The annual average rainfall is 17.4 inches.

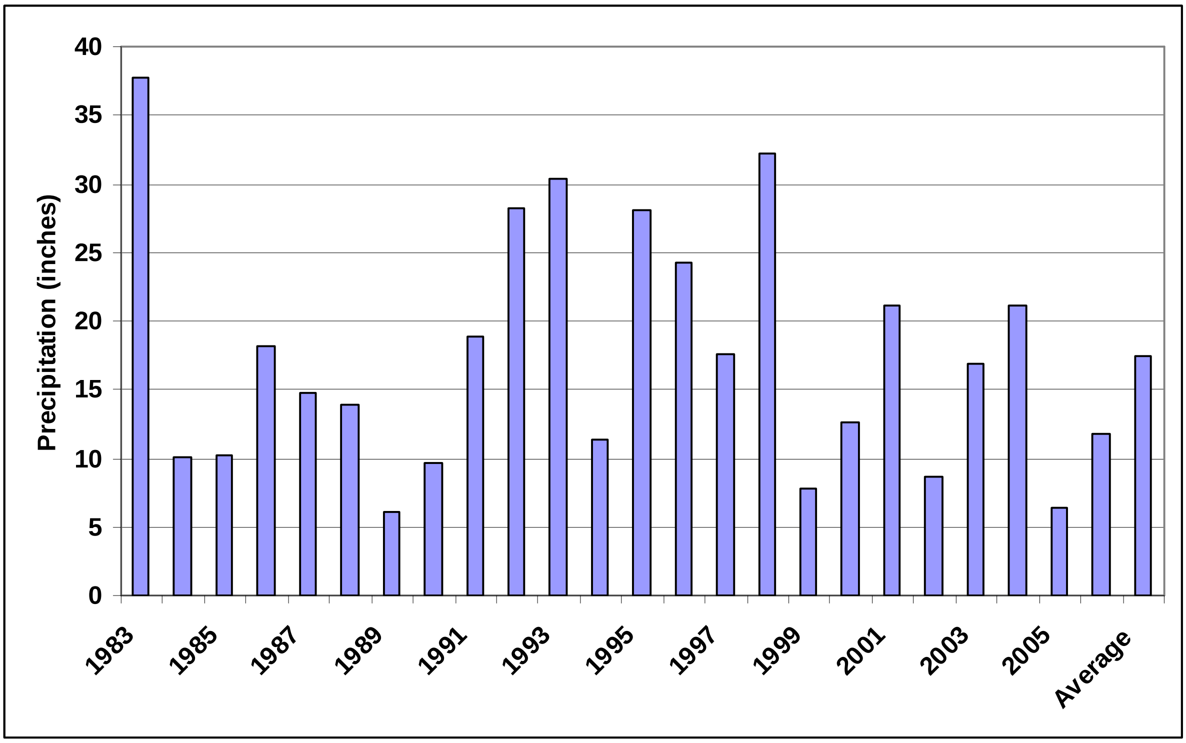

Figure D-36. Annual Rainfall for the Puddingstone Reservoir Subwatersheds

The simulated monthly average runoff depths for land uses in the Puddingstone Reservoir subwatersheds are shown in Table D-47.

Table D-47. Monthly Average Runoff Depths (inches/month) for Land Uses in the Puddingstone Reservoir Subwatersheds, 1983 - 2006

Land Use	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Commercial and services	2.545	3.469	2.162	0.777	0.227	0.133	0.040	0.040	0.124	0.624	1.029	1.636
Cropland and pasture	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Evergreen forest land	0.150	0.204	0.214	0.138	0.104	0.081	0.074	0.069	0.068	0.070	0.074	0.086
Herbaceous rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Industrial	2.576	3.501	2.198	0.807	0.251	0.154	0.059	0.058	0.141	0.643	1.050	1.660

Land Use	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mixed rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Other urban or built-up	2.104	2.866	1.807	0.664	0.208	0.127	0.049	0.048	0.115	0.523	0.853	1.350
Residential	0.737	1.002	0.699	0.305	0.144	0.100	0.072	0.068	0.086	0.208	0.309	0.465
Shrub & brush rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Transportation, communications, utilities	2.545	3.469	2.162	0.777	0.227	0.133	0.040	0.040	0.124	0.624	1.029	1.636
Transitional areas	0.471	0.638	0.484	0.236	0.132	0.096	0.077	0.072	0.081	0.147	0.202	0.293
Parkland*	0.192	0.252	0.267	0.175	0.129	0.097	0.088	0.082	0.080	0.085	0.092	0.112

*Previously “other urban or built-up” areas in the southern subwatershed (see discussion in Section D.3).

Figure D-37 summarizes the monthly average runoff volumes delivered to Puddingstone Reservoir from 1983 through 2006. The total annual runoff to the reservoir is 2,692 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

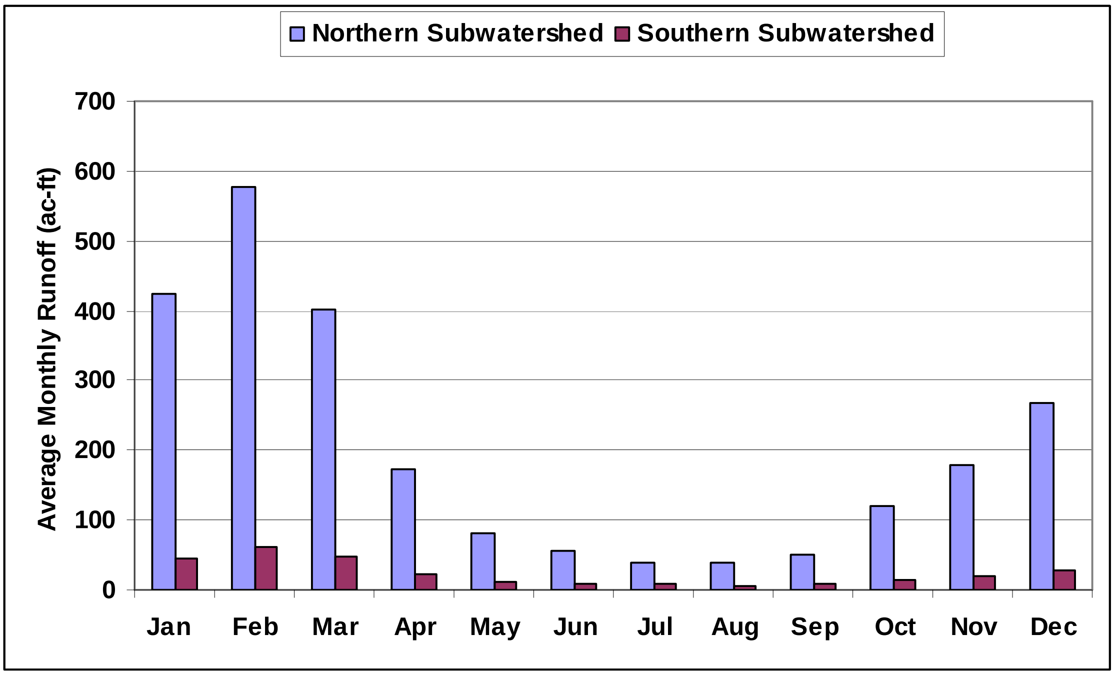

Figure D-37. Monthly Average Runoff Volumes to Puddingstone Reservoir (1983-2006)

Though the Metropolitan Water District can divert water to Puddingstone Reservoir from outside the watershed, this practice is seldom used (personal communication, Adam Walden, Los Angeles County Department of Public Works, 9/16/09) and does not impact the average conditions for this reservoir.

D.10.2 SEDIMENT LOADS

Sediment loads associated with upland areas are calculated from simulated runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-48 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-48. Average Annual Sediment Loads to Puddingstone Reservoir

Subwatershed	Jurisdiction	Sediment (tons/yr)
Northern	Caltrans	13.5
Northern	Claremont	4.49
Northern	County of Los Angeles	27.7
Northern	La Verne	197
Northern	Pomona	0.473
Northern	San Dimas	1.63
Northern	Angeles National Forest	1.36
Southern	Caltrans	1.42
Southern	La Verne	1.24
Southern	Pomona	1.68
Southern	San Dimas	14.8
Total		**266**

Sedimentation data collected by the USACE from 1925 to 1980 indicate that approximately 31 acre-feet per year (approximately 1.5 inches per year) have been delivered to Puddingstone Reservoir in the past (Department of Boating and Waterways and State Coastal Conservancy, 2002). Measurements occurred in 10- to 20-year increments that likely captured anomalous events such as flooding and fires followed by precipitation that typically result in mass wasting of sediment. In addition, rates were measured during periods of rapid development when the use of erosion control practices on construction sites was uncommon. During this development period, natural channels were replaced with hardened structures, decreasing sediment loading associated with channel erosion. Though these sediment loads have impacted Puddingstone Reservoir in the past, they are not considered to represent average current conditions (the average annual sediment load of 266 tons/year is equivalent to 0.00465 inches per year; Table D-48). Also, large pulses of sediment are likely delivered during a few events with much of the associated pollutant loading quickly buried and sequestered and therefore unavailable for release to the water column or entrance to the food chain via benthic organisms. Thus, no additional pollutant loads were assumed for mass wasting events.

D.10.3 NUTRIENT LOADS

Nutrient loads from upland areas are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-49 summarizes the total nitrogen and total phosphorus loads delivered to Puddingstone Reservoir from each jurisdiction and subwatershed. See example calculations in Section D.3.

Table D-49. Average Annual Nutrient Loads to Puddingstone Reservoir

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Northern	Caltrans	1,409	214
Northern	Claremont	766	52.3
Northern	County of Los Angeles	4,011	467
Northern	La Verne	21,698	3,254
Northern	Pomona	51.6	7.71
Northern	San Dimas	244	37.2
Northern	Angeles National Forest	301	10.3
Southern	Caltrans	148	22.5
Southern	La Verne	147	19.4
Southern	Pomona	276	34.5
Southern	San Dimas	2,433	272
Total		**31,484**	**4,391**

D.10.4 MERCURY LOADS

Mercury loads resulting from upland areas are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Water column mercury concentrations measured from major inputs to the lakes are applied to simulated runoff volumes and input sediment mercury concentrations are applied to the calculated sediment loads (Section D.10.2) to estimate water column and sediment associated mercury loads, respectively. Figure D-38 shows the locations of the monitoring stations in the Puddingstone Reservoir watershed. Stations PR19 and PR19SD are in close proximity and display as one yellow square in Figure D-38.

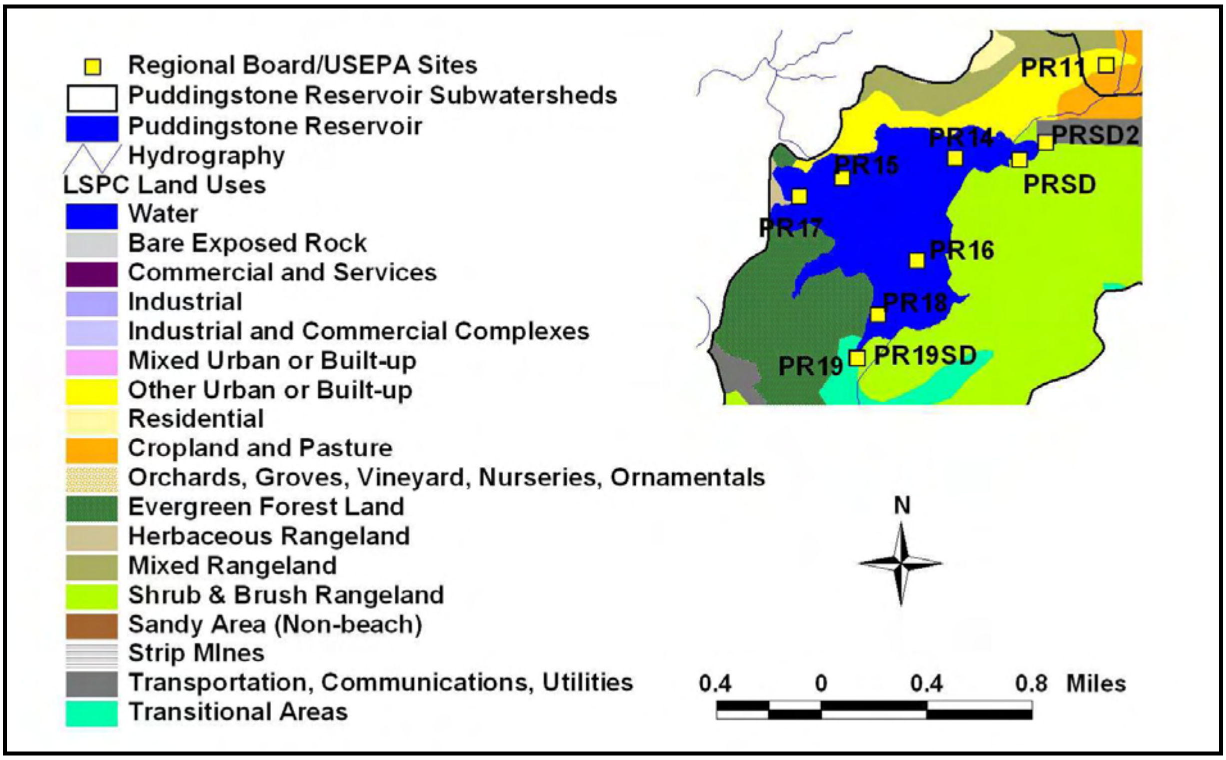

Figure D-38. Monitoring Stations in the Puddingstone Reservoir Watershed

Table D-50 and Table D-51 present the methyl and total mercury concentrations observed at the mouth of each major input in the water column and sediments, respectively. More details regarding these data are presented in Appendix G (Monitoring Data).

Table D-50. Tributary/Inflow Mercury Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)
PR11	2/24/2009	14:30	0.043	3.52
PRSD		13:10	<0.020	2.65
PR11	7/16/2009	11:45	0.553	4.24
PRSD2		13:10	0.046	7.55

Table D-51. Inflow Mercury Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
PR11	2/24/2009	14:30	<0.011	52.9
PR11	7/16/2009	11:45	1.71	73.1
PR19		14:05	0.068	34.3
PR19SD		14:10	0.940	66.2
PRSD2		13:10	1.14	50.4

Concentrations of total and methylmercury vary seasonally at each input. Water column and sediment concentration data are available during both the summer and winter season at station PR11, which represents loading from the northern subwatershed. Mercury loading is associated with both sediment and runoff from upland areas and given the availability of seasonal data, concentrations of total and methylmercury in water (Table D-50) and sediment (Table D-51) varied for the summer and winter. To determine sediment loading of mercury, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data (Table D-51) were then applied to these sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations (Table D-50) to LSPC simulated runoff volumes (Section D.3 provides examples of these calculations). The July 2009 monitoring data were used to estimate loads for the summer season (May through October), and the February 2009 data were used to estimate loads for the winter season (November through April). The sediment methlymercury concentration was below the detection limit for the winter sampling so it was assumed equal to one-half the detection limit, or 0.006 μg/kg (Table D-51).

In the southern subwatershed, similar calculations were performed to estimate mercury loading associated with runoff and sediment; however, additional monitoring stations were available to represent the loading throughout this subwatershed. Specifically, water column concentrations of methyl and total mercury were measured at two storm drain outlets located near the campground in the northeastern section of the watershed. Each drain was measured during only one season. Measurements at PRSD were used to estimate winter concentrations, and measurements at PRSD2 were used to estimate summer concentrations. The winter methylmercury concentration in water was below the detection limit so it was assumed to be equal to one-half the detection limit, or 0.01 ng/L (Table D-50). Mercury sediment concentrations for inlets located in the southern subwatershed were measured only during the summer season. The summer season total mercury sediment concentration was assumed equal to the average of the concentrations measured at PR19, PR19SD, and PRSD2 (50.3 μg/kg; Table D-51). The winter season total mercury sediment concentration (36.4 μg/kg) was assumed equal to the summer concentration divided by the ratio of summer to winter total mercury sediment concentrations observed at PR11 (73.1 μg/kg ÷ 52.9 μg/kg = ratio of 1.38; Table D-51). Similar assumptions were used to estimate the summer and winter methylmercury sediment concentrations applicable to the southern subwatershed.

The assumed concentrations were applied to the runoff and sediment loads estimated from each jurisdiction within the watershed. Assumed total mercury concentrations and resulting loads are summarized in Table D-52. See example calculations in Section D.3. Results for methylmercury are presented in Table D-53. Approximately 92 percent of the wet weather total mercury load and 98 percent of the wet weather methylmercury load originate in the northern subwatershed, which accounts for 86 percent of the watershed area.

Table D-52. Total Mercury Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed

Sub-watershed	Jurisdiction	Area (ac)	Summer Water Column Hg (ng/L)	Winter Water Column Hg (ng/L)	Summer Sediment Hg (µg/kg)	Winter Sediment Hg (µg/kg)	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Northern	Caltrans	110	4.24	3.52	73.1	52.9	0.520	0.672	1.19
Northern	Claremont	578	4.24	3.52	73.1	52.9	0.372	0.239	0.611
Northern	County of Los Angeles	1,865	4.24	3.52	73.1	52.9	1.68	1.45	3.13
Northern	La Verne	4,079	4.24	3.52	73.1	52.9	7.97	10.1	18.1
Northern	Pomona	5.28	4.24	3.52	73.1	52.9	0.019	0.024	0.043
Northern	San Dimas	28.5	4.24	3.52	73.1	52.9	0.090	0.082	0.172
Northern	Angeles National Forest	293	4.24	3.52	73.1	52.9	0.161	0.074	0.234
Southern	Caltrans	11.6	7.55	2.65	50.3	36.4	0.047	0.049	0.096
Southern	La Verne	33.5	7.55	2.65	50.3	36.4	0.054	0.043	0.097
Southern	Pomona	81.8	7.55	2.65	50.3	36.4	0.108	0.058	0.166
Southern	San Dimas	1,043	7.55	2.65	50.3	36.4	1.05	0.522	1.57
Total		**8,128**	**N/A**	**N/A**	**N/A**	**N/A**	**12.1**	**13.3**	**25.4**

N/A = Not applicable

Table D-53. Methylmercury Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed

Sub-watershed	Jurisdiction	Area (ac)	Summer Water Column MeHg (ng/L)	Winter Water Column MeHg (ng/L)	Summer Sediment MeHg (µg/kg)	Winter Sediment MeHg (µg/kg)	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Northern	Caltrans	110	0.553	0.043	1.710	0.006	1.31E-02	2.01E-03	1.51E-02
Northern	Claremont	578	0.553	0.043	1.710	0.006	1.96E-02	2.00E-03	2.16E-02
Northern	County of Los Angeles	1,865	0.553	0.043	1.710	0.006	7.34E-02	9.95E-03	8.34E-02
Northern	La Verne	4,079	0.553	0.043	1.710	0.006	2.52E-01	5.65E-02	3.08E-01
Northern	Pomona	5.28	0.553	0.043	1.710	0.006	5.09E-04	7.31E-05	5.82E-04
Northern	San Dimas	28.5	0.553	0.043	1.710	0.006	2.48E-03	2.77E-04	2.75E-03
Northern	Angeles National Forest	293	0.553	0.043	1.710	0.006	9.38E-03	7.34E-04	1.01E-02
Southern	Caltrans	11.6	0.046	0.010	0.716	0.002	2.03E-04	8.84E-05	2.92E-04
Southern	La Verne	33.5	0.046	0.010	0.716	0.002	2.46E-04	9.55E-05	3.41E-04
Southern	Pomona	81.8	0.046	0.010	0.716	0.002	5.00E-04	1.57E-04	6.58E-04
Southern	San Dimas	1,043	0.046	0.010	0.716	0.002	5.01E-03	1.70E-03	6.70E-03
Total		**8,128**	**N/A**	**N/A**	**N/A**	**N/A**	**0.376**	**0.074**	**0.450**

N/A = Not applicable

D.10.5 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant using monitoring data collected by USEPA, the Regional Board, and UCLA between 2008 and 2009. Only data from sites representing inflows are used; these include locations in an inflow or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Puddingstone Reservoir, this included PR-11, PR-19, PR-19SD, and PR-SD2 (Figure D-39).

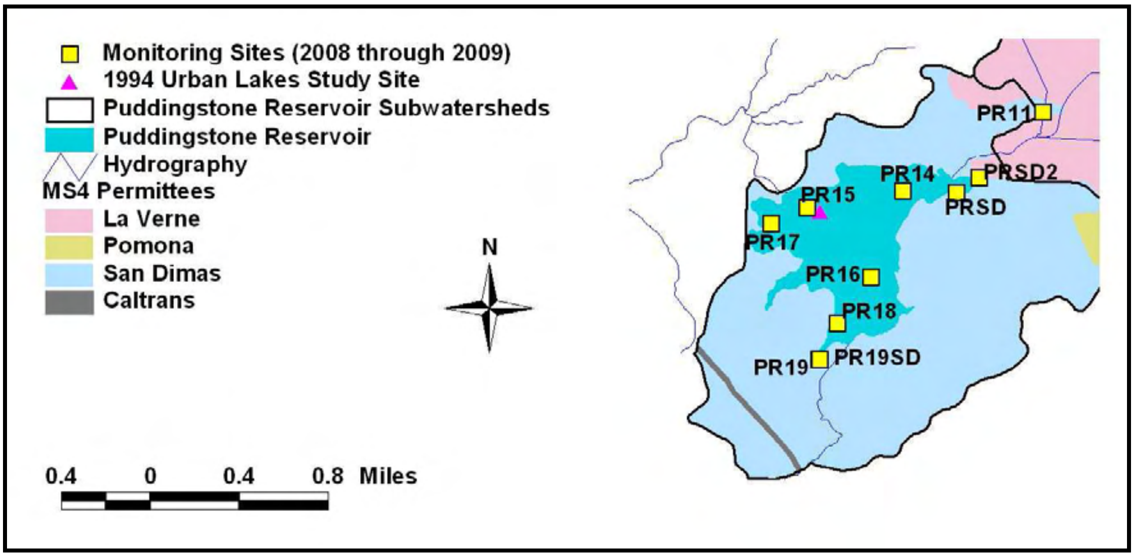

Figure D-39. Puddingstone Reservoir Monitoring Stations

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads were associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For OC Pesticides and PCBs where some of the samples had detectable quantities of a pollutant, the average inflow concentration was calculated assuming samples analyzed below detection limits were equal to one-half the detection limit. All dieldrin samples were below detection limits; for dieldrin the concentration was calculated directly from the detection limits. The inflow sediment data are summarized in Table D-54 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-54. Summary of Sediment Data near Inflow Locations for Puddingstone Reservoir

Parameter	No. of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit (µg/kg dry weight)
Chlordane	3	3	5.11	0.39-1.58
DDT	3	2	5.50	0.77-3.17
Total PCBs	3	3	50.3	0.39-1.58
Dieldrin	3	0	< 1.0	1.0

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

These input sediment concentrations were applied to the calculated sediment loads (Section D.10.2) to estimate the sediment-associated OC Pesticides and PCBs loads entering the lake. Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the two Puddingstone Reservoir subwatersheds in Table D-55 along with the predicted sediment loads for each land use. Dissolved concentrations in inflows are assumed insignificant.

Table D-55. Annual Sediment Load for Puddingstone Reservoir

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Northern	Caltrans	110	13.5	5.10%
Northern	Claremont	578	4.5	1.69%
Northern	County of Los Angeles	2,056	27.7	10.43%
Northern	La Verne	4,181	168	63.23%
Northern	General Industrial Stormwater Permittees* (in the city of La Verne)	1,865	24.8	9.33%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	4,079	4.5	1.70%
Northern	Pomona	5.28	0.5	0.18%
Northern	San Dimas	28.5	1.6	0.62%
Northern	Angeles National Forest	293	1.4	0.51%
Southern	Caltrans	11.6	1.4	0.54%
Southern	La Verne	33.5	1.2	0.47%
Southern	Pomona	81.8	1.7	0.63%
Southern	San Dimas	1,042	14.8	5.59%
Southern	County of Los Angeles (Irrigation)		0.0	0.00%
Total		**8,128**	**265.5**	**100%**

*The disturbed area associated with general construction and general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

The chlordane, PCB, DDT, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-54) to the calculated sediment loads (Table D-55; 265.5 tons per year). See example calculations in Section D.3. The dieldrin calculation is based on the detection limit of 1 μg/kg dry weight. Loads for each jurisdiction are shown by subwatershed in Table D-56.

Table D-56. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Watershed (g/yr)

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual Dieldrin Loads	Annual DDT Load	Percent of Total Load
Northern	Caltrans	0.62	0.063	0.012	0.068	5.10%
Northern	Claremont	0.20	0.021	0.004	0.022	1.69%
Northern	County of Los Angeles	1.30	0.128	0.025	0.138	10.43%
Northern	La Verne	7.68	0.778	0.152	0.838	63.23%
Northern	General Industrial Stormwater Permittees* (in the city of La Verne)	1.13	0.115	0.022	0.124	9.33%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	0.21	0.021	0.004	0.022	1.69%
Northern	Pomona	0.02	0.002	0.000	0.002	0.18%
Northern	San Dimas	0.07	0.008	0.001	0.008	0.62%
Northern	Angeles National Forest	0.06	0.006	0.001	0.007	0.51%
Southern	Caltrans	0.06	0.007	0.001	0.007	0.54%
Southern	La Verne	0.06	0.006	0.001	0.006	0.47%
Southern	Pomona	0.08	0.008	0.001	0.008	0.63%
Southern	San Dimas	0.68	0.069	0.002	0.074	5.59%
Southern	County of Los Angeles (Irrigation)	0.00	0.000	0.013	0.00	0.00%
Total		**12.12**	**1.23**	**0.24**	**1.32**	**100%**

*The disturbed area associated with general construction and general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

D.11 Santa Fe Dam Park Lake

Santa Fe Dam Park Lake is located in the San Gabriel River Basin. Though the park lake is located near the Santa Fe Dam Diversion Channel, it does not receive any diverted flow from the San Gabriel River (personal communication, Arthur Gotingco, Los Angeles County Public Works Department, 7/13/2009).

Impairments of this lake include pH, copper, and lead. Output from the San Gabriel River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads of nutrients from upland areas, which may be contributing to the pH impairment.

One subwatershed comprises the drainage area to Santa Fe Dam Park Lake. This subwatershed is comprised of 362 acres. No storm water sewer system is present in the watershed, so all allocations for the TMDLs were load allocations.

Figure D-40 shows the MS4 stormwater permittees in the Santa Fe Dam Park Lake watershed. Most of the area is located in Irwindale, with a small portion in Azusa.

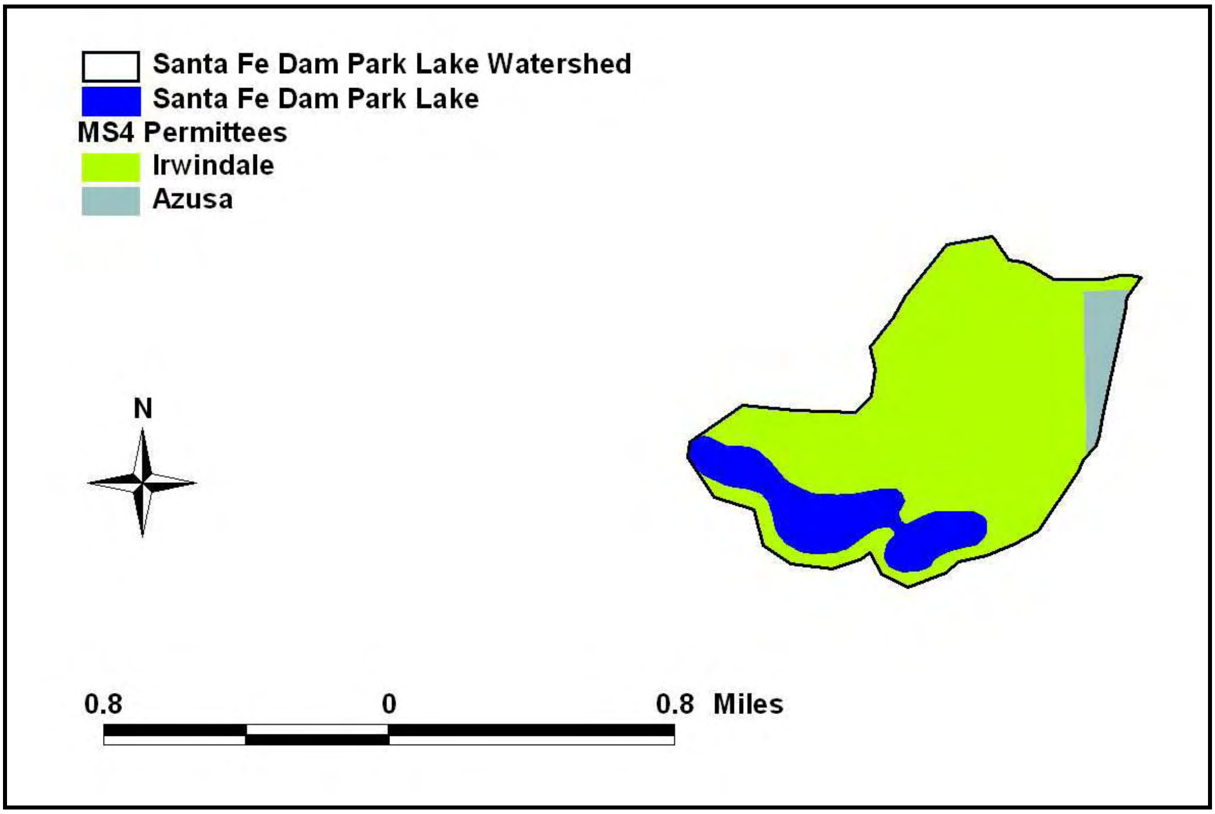

Figure D-40. MS4 Permittees in the Santa Fe Dam Park Lake Subwatershed

Land uses identified in the San Gabriel River LSPC model are shown in Figure D-41. Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that the portion of area classified by the LSPC model as strip mines had not been mined for some time. The SCAG 2005 database classified this area as vacant; the current satellite imagery shows this area to be re-established shrub/brush rangeland. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the 6.25 acres classified by the LSPC model as strip mines were therefore converted to shrub and brush rangeland for this loading analysis. Table D-57 summarizes the post-processed land use areas used to estimate pollutant loading from upland areas draining to Santa Fe Dam Park Lake.

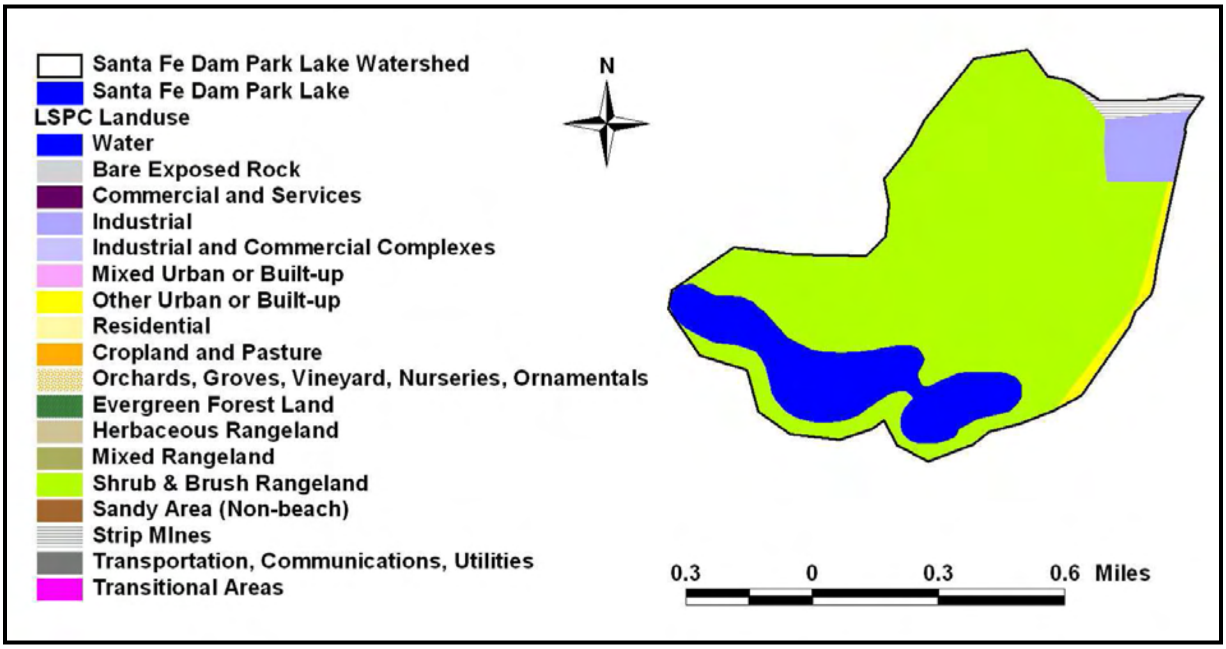

Figure D-41. LSPC Land Use Classes for the Santa Fe Dam Park Lake Subwatershed

Table D-57. Land Use Areas (ac) Draining to Santa Fe Dam Park Lake

Land Use	Azusa	Irwindale	Total
Industrial	11.5	7.16	**18.7**
Other urban or built-up	3.94	4.54	**8.48**
Shrub & brush rangeland	6.94	328	**335**
Total	**22.4**	**340**	**362**

D.11.1 RUNOFF

LSPC-predicted runoff from the Santa Fe Dam Park Lake subwatershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-42 shows the simulated annual rainfall for the Santa Fe Dam Park Lake subwatershed. The annual average rainfall is 18.5 inches.

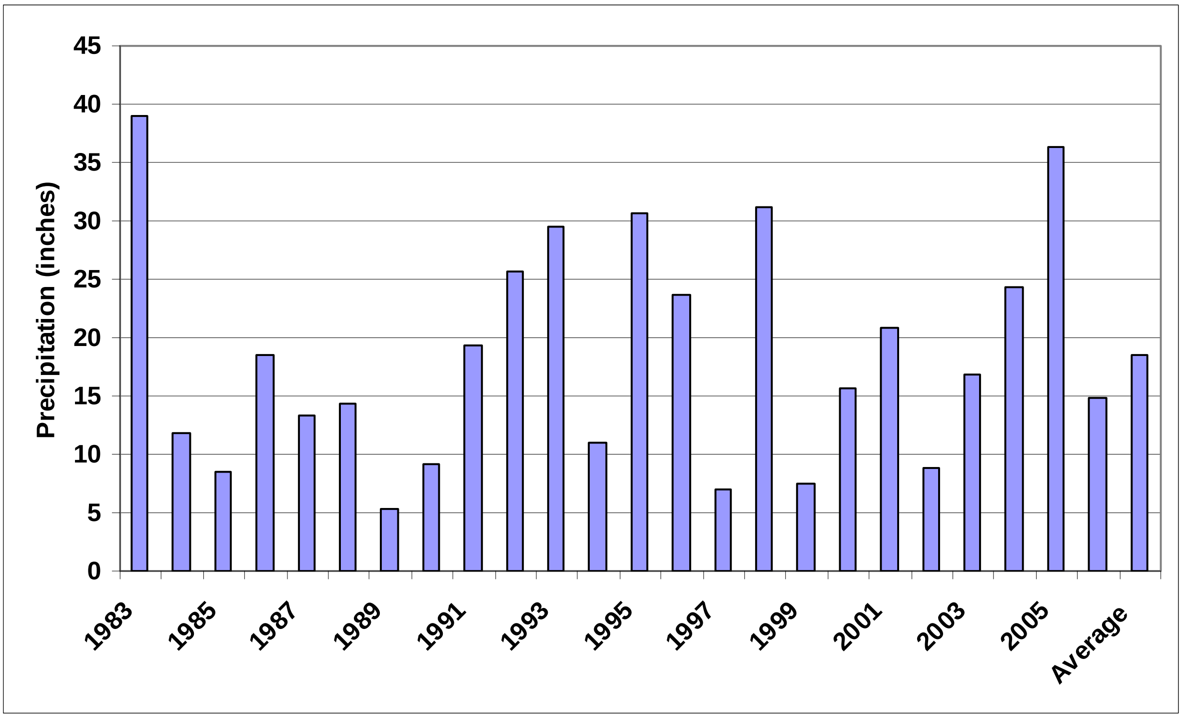

Figure D-42. Annual Rainfall for the Santa Fe Dam Park Lake Subwatershed

The simulated monthly average runoff depths for land uses in the Santa Fe Dam Park Lake subwatershed are shown in Table D-58.

Table D-58. Monthly Average Runoff Depths (inches/month) for Land Uses in the Santa Fe Dam Park Lake Subwatershed, 1983 - 2006

Month	Other Urban or Built Up	Heavy Industrial	Shrub/Brush Rangeland
January	2.2518	2.7522	0.0771
February	3.1443	3.8356	0.1393
March	1.8647	2.2633	0.1246
April	0.7178	0.8713	0.0462
May	0.2278	0.2735	0.0261
June	0.0950	0.1149	0.0076
July	0.0105	0.0121	0.0031
August	0.0504	0.0616	0.0018
September	0.1539	0.1890	0.0016
October	0.4457	0.5482	0.0019
November	0.7481	0.9202	0.0022
December	1.3029	1.6019	0.0061

Figure D-43 summarizes the monthly average runoff volumes delivered to Santa Fe Dam Park Lake. The total annual runoff to the reservoir is 40.9 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

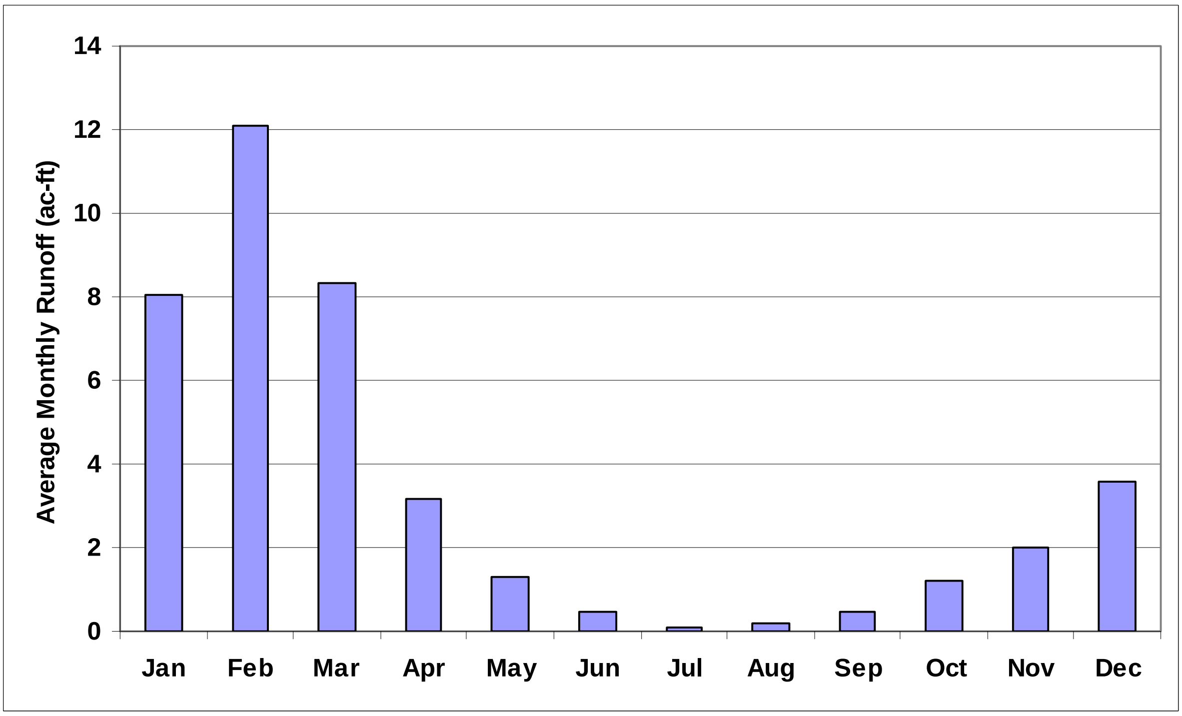

Figure D-43. Monthly Average Runoff Volumes to Santa Fe Dam Park Lake

D.11.2 Nutrient Loads

Nutrient loads are estimated from simulated runoff volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-59 summarizes the total nitrogen and total phosphorus loads delivered to Santa Fe Dam Park Lake from each jurisdiction. See example calculations in Section D.3.

Table D-59. Average Annual Nutrient Loads to Santa Fe Dam Park Lake

Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Azusa	205	27.0
Irwindale	253	23.8
Total	**458**	**50.8**

D.12 Lake Sherwood

Lake Sherwood is located in the Santa Monica Bay Basin and is impaired by mercury (note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL). For consistency with the other two mercury impaired lakes addressed by this TMDL (Puddingstone Reservoir and the El Dorado Park lakes), the upland mercury loads were calculated from tributary monitoring data collected in 2009 and estimates of runoff volumes and sediment loading predicted by an LSPC model. Though an LSPC model has not been developed for the Santa Monica Bay Basin, the land use coverage for the Los Angeles River Basin LSPC model covers the drainage area to Lake Sherwood and was used to predict runoff volumes and sediment loads by land use to Lake Sherwood.

Six subwatersheds comprise the drainage area (10,656 acres) to Lake Sherwood. Figure D-44 shows the MS4 stormwater permittees comprising each subwatershed. Ventura County is the only stormwater permittee in the Western Subwatershed. The Hidden Valley Wash subwatershed is mostly in Ventura County with small portion in Thousand Oaks. The Northern, Near Lake Undeveloped, and Near Lake Developed subwatersheds are comprised of both Ventura County and Thousand Oaks MS4 areas. The Carlisle Canyon subwatershed contains Ventura and Los Angeles County areas as well as Thousand Oaks, California Department of Transportation (Caltrans), and California State Park areas. Neither Ventura or Los Angeles counties (the MS4 stormwater permittees in the watershed) maintain storm drain systems in the Lake Sherwood watershed. However, there are residential developments in the vicinity of the lake which drain to culverts and storm drains. These areas are generally associated with the Sherwood Valley Homeowner’s Association (SVHOA) and Sherwood Development Company. All subwatersheds will receive wasteload allocations except for the Carlisle Canyon and Near Lake Undeveloped subwatersheds. The small Caltrans area in the Carlisle Canyon subwatershed will also receive a wasteload allocation.

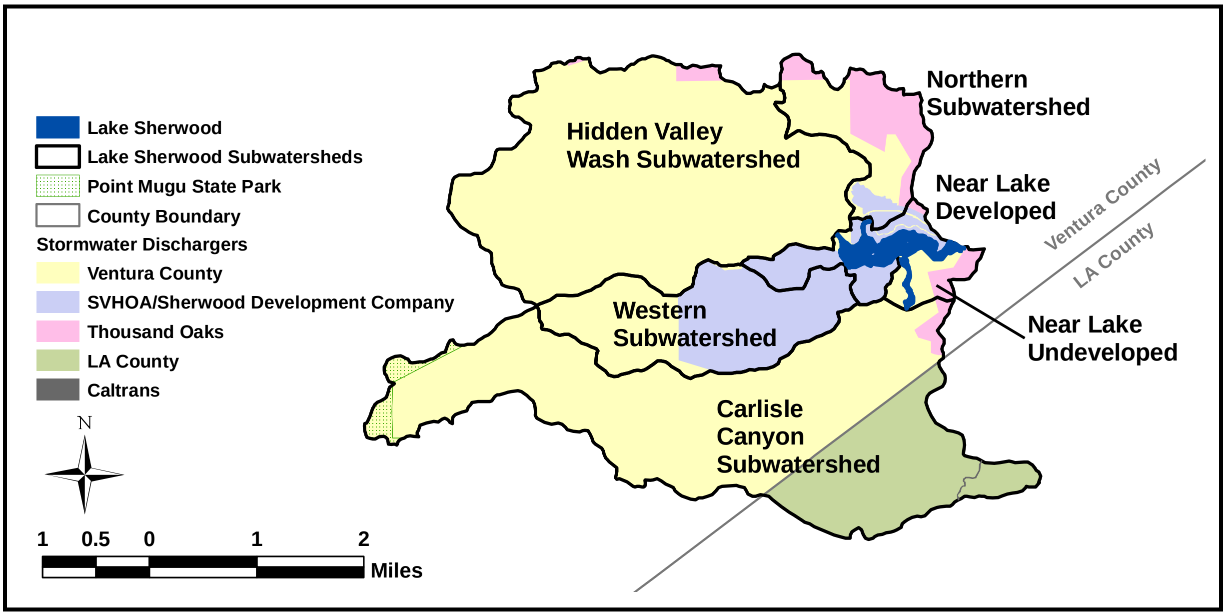

Figure D-44. MS4 Permittees in the Lake Sherwood Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-45. The watershed is comprised mostly of open space, agriculture, residential, and other urban areas. A single parcel of commercial development was identified in the Near Lake Developed Subwatershed. Review of SCAG 2005 land use data confirmed that much of the watershed is currently used for agriculture. The area in the Carlisle Canyon subwatershed under the Caltrans jurisdiction (Figure D-44) was simulated as industrial to estimate sediment loading and runoff volumes from the area associated with this State highway (i.e., changed from open to industrial land use). This was the only modification to the land use classifications for the Lake Sherwood subwatersheds. Table D-60 through Table D-65 summarize the land use areas used to estimate pollutant loading from upland areas draining to Lake Sherwood.

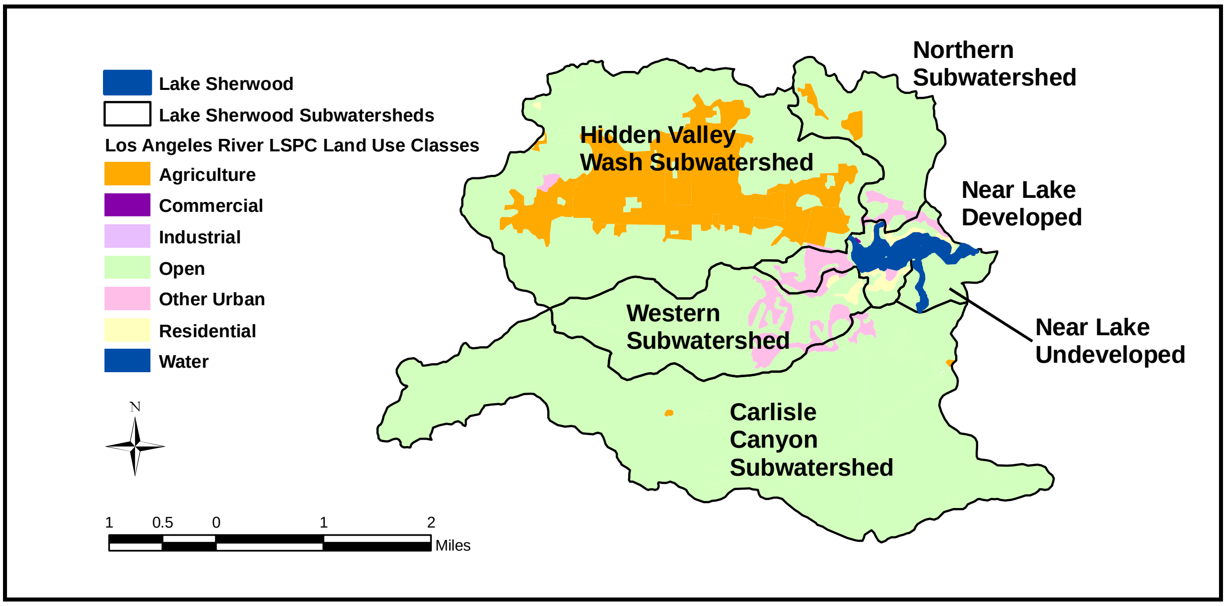

Figure D-45. LSPC Land Use Classes for the Lake Sherwood Subwatersheds

Table D-60. Land Use Areas (ac) Draining from the Northern Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	SVHOA	Total
Agriculture	42	0	0	**42**
Commercial	0	0	0	**0**
Industrial	0	0	0	**0**
Open	301	338	29	**669**
Other Urban	7.2	0	34	**41**
Residential	0.20	0	2	**2**
Total	**351**	**338**	**65**	**754**

Table D-61. Land Use Areas (ac) Draining from the Hidden Valley Wash Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	1,328	0	**1,328**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	2,441	40.4	**2,482**
Other Urban	19.7	0	**20**
Residential	3.97	0	**4**
Total	**3,793**	**40.4**	**3,833**

Table D-62. Land Use Areas (ac) Draining from the Western Subwatershed to Lake Sherwood

Land Use	Ventura County	SVHOA	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	548	587	**1,136**
Other Urban	0	165	**165**
Residential	0	20	**20**
Total	**548**	**772**	**1,321**

Table D-63. Land Use Areas (ac) Draining from the Carlisle Canyon Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	County of Los Angeles	Caltrans	Point Mugu State Park	Total
Agriculture	5.24	0	0.118	0	0	**5.36**
Commercial	0	0	0	0	0	**0**
Industrial	0	0	0	2.75	0	**2.75**
Open	2,866	50.4	1,149	0	101	**4,166**
Other Urban	34.2	0	0.06	0	0	**34**
Residential	0	0	0	0	0	**0**
Total	**2,905**	**50**	**1,149**	**2.75**	**101**	**4,209**

Table D-64. Land Use Areas (ac) Draining from the Near Lake Undeveloped Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	126	70.9	**197**
Other Urban	0	0	**0**
Residential	0.004	0	**0.004**
Total	**126**	**70.9**	**197**

Table D-65. Land Use Areas (ac) Draining from the Near Lake Developed Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	SVHOA	Total
Agriculture	0	0	0	**0.0**
Commercial	1.13	0	0	**1.1**
Industrial	0	0	0	**0**
Open	15	8.8	143	**167**
Other Urban	3.3	0	110	**113**
Residential	4.4	0	57	**61**
Total	**24**	**8.8**	**310**	**343**

D.12.1 RUNOFF

LSPC-based runoff from the Lake Sherwood subwatersheds is primarily driven by the land use and soil characteristics of a nearby drainage area and the nearest meteorological station represented in the model. Figure D-46 shows the simulated annual rainfall for the Lake Sherwood subwatersheds. The annual average rainfall is 17.5 inches.

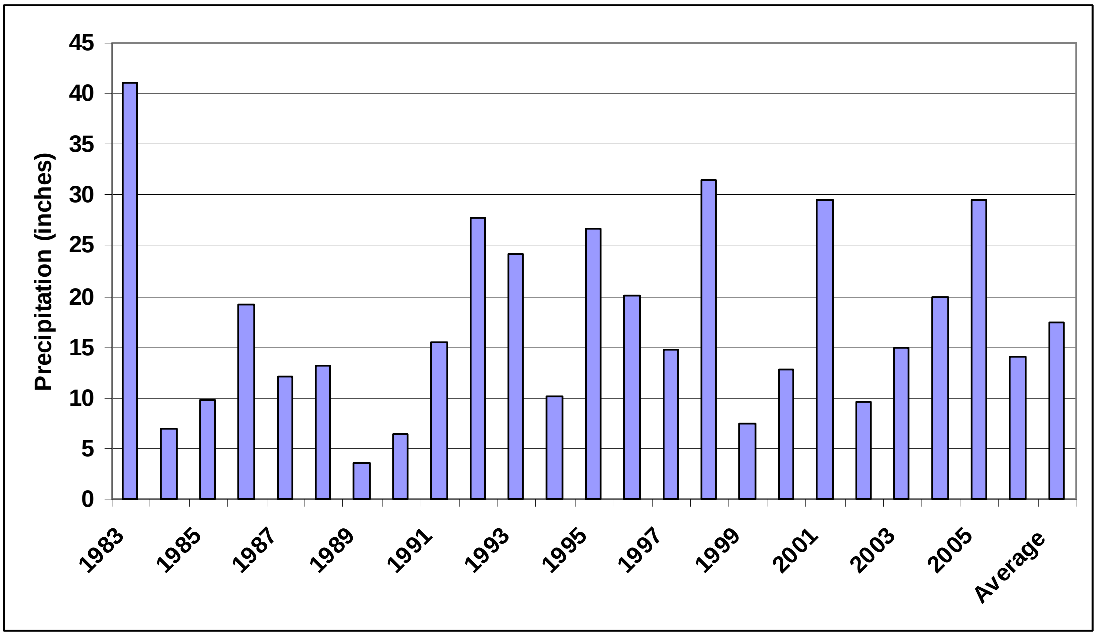

Figure D-46. Annual Rainfall for the Lake Sherwood Subwatersheds

The simulated monthly average runoff depths for land uses in the Lake Sherwood subwatersheds are shown in Table D-66.

Table D-66. Monthly Average Runoff Depths (in/mo) for Land Uses in the Lake Sherwood Subwatersheds, 1983 – 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.3808	2.6527	2.3868	0.1271	1.7220	1.6687
February	0.8634	3.7749	3.4439	0.3202	2.6159	2.5495
March	0.5419	2.2277	2.0322	0.2219	1.5434	1.5042
April	0.1031	0.7350	0.6590	0.0473	0.4689	0.4536
May	0.0452	0.2213	0.2008	0.0174	0.1493	0.1452
June	0.0032	0.0224	0.0199	0.0020	0.0138	0.0134
July	0.0007	0.0037	0.0033	0.0005	0.0024	0.0023
August	0.0016	0.0205	0.0181	0.0009	0.0121	0.0116
September	0.0112	0.1552	0.1370	0.0056	0.0915	0.0878
October	0.0385	0.5420	0.4784	0.0192	0.3193	0.3065
November	0.0689	0.9656	0.8524	0.0342	0.5691	0.5464
December	0.1247	1.6257	1.4379	0.0585	0.9685	0.9309

Figure D-47 summarizes the monthly average runoff volumes delivered to Lake Sherwood. The total annual volume delivered to the lake is 1,205 ac-ft. The months May through October each contribute less than 3 percent of the annual runoff volume.

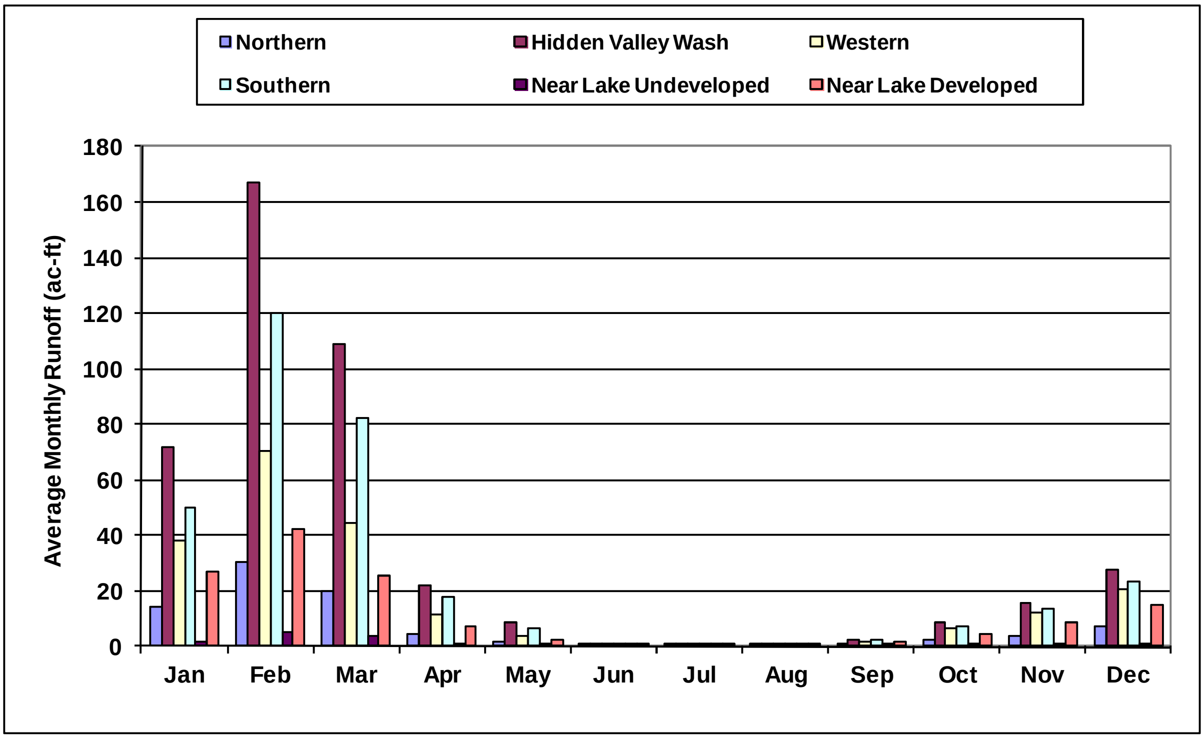

Figure D-47. Monthly Average Runoff Volumes to Lake Sherwood

D.12.2 SEDIMENT LOADS

Sediment loads associated with upland areas are calculated from simulated runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-67 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-67. Average Annual Sediment Loads to Lake Sherwood

Subwatershed	Jurisdiction	Sediment (tons/yr)
Western	Ventura County	1.53
Western	SVHOA	11.60
Hidden Valley Wash	Thousand Oaks	0.11
Hidden Valley Wash	Ventura County	507.42
Near Lake Undeveloped	Thousand Oaks	0.198
Near Lake Undeveloped	Ventura County	0.353
Near Lake Developed	Thousand Oaks	0.02

Subwatershed	Jurisdiction	Sediment (tons/yr)
Near Lake Developed	Ventura County	0.54
Near Lake Developed	SVHOA	9.29
Northern	Thousand Oaks	0.94
Northern	Ventura County	17.02
Northern	SVHOA	2.01
Carlisle Canyon	Caltrans	0.31
Carlisle Canyon	County of Los Angeles	3.26
Carlisle Canyon	Thousand Oaks	0.14
Carlisle Canyon	Ventura County	11.83
Carlisle Canyon	Point Mugu State Park	0.28
Total		**567**

For Lake Sherwood, the reported average annual sedimentation rate measured from 1905 to 1938 ranged from 2.5 to 10 acre-feet per year (0.22 to 0.88 inches per year) (Department of Boating and Waterways and State Coastal Conservancy, 2002). These measurements likely capture anomalous events such as flooding and fires that result in mass wasting of sediment and are not considered average conditions for the lake (the predicted average annual sediment load of 567 tons/yr is equal to 0.018 in/yr; Table D-67). Because large pulses of sediment are likely delivered during a few events, much of the associated pollutant loading is quickly buried and sequestered and therefore unavailable for release to the water column or entrance to the food chain via benthic organisms. Thus no additional pollutant loads were assumed for mass wasting events.

D.12.3 Mercury Loads

Mercury loads from each subwatershed are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Water column mercury concentrations measured from major inputs to the lakes are applied to simulated runoff volumes, and input mercury sediment concentrations are applied to the calculated sediment loads (see Section D.12.2) to estimate water column and sediment associated mercury loads, respectively. Figure D-48 shows the locations of the monitoring stations in the Lake Sherwood Watershed.

Table D-68 and Table D-69 present the methyl and total mercury concentrations observed at the mouth of each major input in the water column and sediments, respectively. More details regarding this data are presented in Appendix G (Monitoring Data).

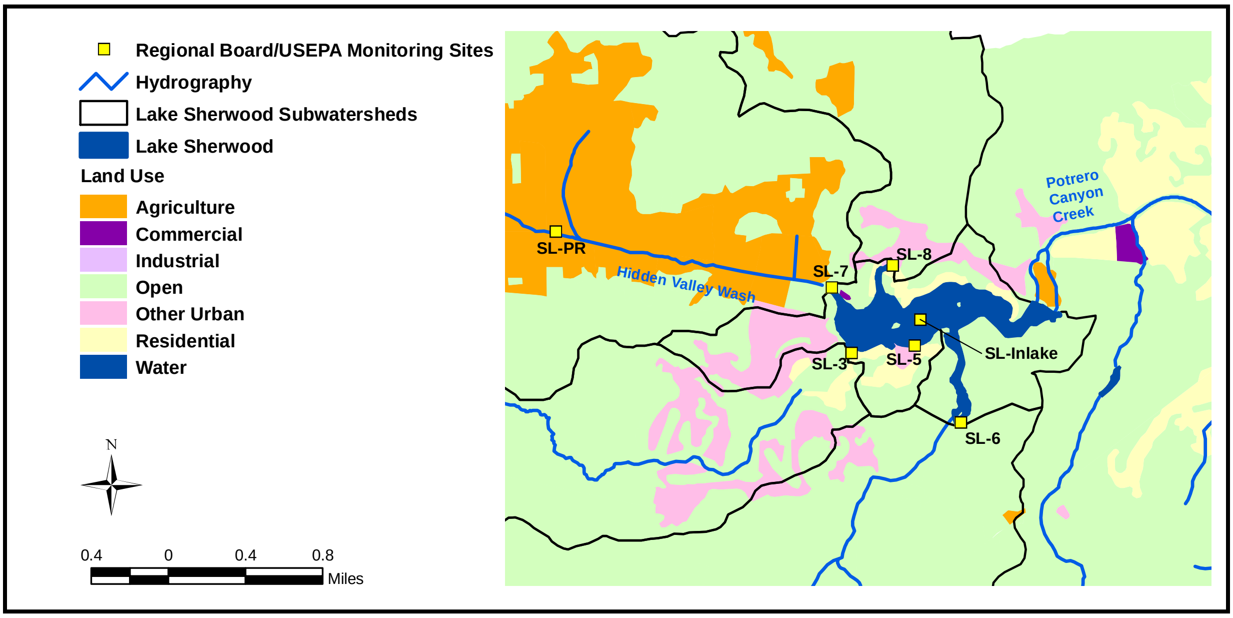

Figure D-48. Monitoring Stations in the Lake Sherwood Watershed

Table D-68. Tributary/Inflow Mercury Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)
SL-3	2/25/2009	13:00	0.157	6.00
SL-6		11:00	0.216	2.96
SL-8		11:45	0.025[1]	23.9
SL-3	7/13/2009	8:55	0.536	4.58
SL-3D		8:55	NA	4.63
SL-7		10:15	3.41	11.3
SL-8		10:00	0.096	54.0

[1] Temperature requirements for methylmercury analysis not met.

Table D-69. Inflow Mercury Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
SL-3	2/25/2009	13:00	0.269	92.7
SL-6		11:00	0.136	129
SL-5		13:15	0.145	51.0
SL-7[1]		08:30	2.53	243
SL-3	7/13/2009	8:55	0.397	392
SL-3D		8:55	NA	265

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
SL-5		9:45	0.657	62.9
SL-7		10:15	0.453	275
SL-PR		10:50	0.009	60.3
SL-8		10:00	0.696	63.3

[1] Sediment concentration data indicated that this site, which is located in a stagnant backwater area at the mouth of Hidden Valley Wash, may be a “hot spot” for methylation and not representative of the sediment methylmercury concentrations delivered from the subwatershed as a whole (sediment methylmercury concentrations were an order of magnitude greater than those observed at other sites in the watershed).

Concentrations of total and methylmercury vary seasonally at each input. Water column and sediment concentration data are available during both the summer and winter season at SL-3, which represents loading from the Western Subwatershed. Mercury loading is associated with both sediment and runoff from upland areas and given the availability of seasonal data, concentrations of total and methylmercury in water (Table D-68) and sediment (Table D-69) varied for the summer and winter. To determine sediment loading of mercury, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads and the sediment mercury concentrations from monitoring data (Table D-69) were then applied to these sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations (Table D-68) to LSPC simulated runoff volumes (Section D.3 provides examples of these calculations). The July 2009 monitoring data were used to estimate loadings for the summer season (May through October) and the February 2009 data were used to estimate loadings for the winter season (November through April). Similar calculations were performed to estimate mercury loading associated with runoff and sediment in the other subwatersheds.

The northern subwatershed is represented by Site SL-8, which has water column total mercury concentration data during both seasons but sediment data only in the summer season. Winter season total mercury concentration in the sediments (32.1 µg/kg) was assumed equal to the summer season total mercury concentration divided by the average observed ratio of summer to winter sediment total mercury concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69). In addition, the water column methylmercury sample collected during the winter event was not maintained within the temperature constraints required for accurate analysis. Therefore, the winter season water column methylmercury concentration for the northern subwatershed (0.028 ng/L) was assumed equal to the summer season concentration divided by the ratio of summer to winter methylmercury observed at site SL-3 (0.536 ng/L ÷ 0.157 ng/L = ratio of 3.41; Table D-68) [water column data were not available for either season at SL-5, and SL-7 is likely not reflective of methylmercury concentrations in the watershed as a whole]. Winter sediment concentrations of methylmercury (0.232 µg/kg) were assumed equal to the summer concentration divided by the average observed ratio at SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69).

Sediment concentration data are available for both seasons at SL-5, which represents loading from the Near Lake Developed Subwatershed. Water was not flowing at SL-5 during either monitoring event so SL-8 data and related assumptions are used to represent water column concentrations for the Near Lake Developed Subwatershed.

Observations at SL-6 are used to estimate loading from the Carlisle Canyon and Near Lake Undeveloped Subwatersheds; the land use in both of these subwatersheds is primarily undeveloped. However, monitoring data are only available at SL-6 during the winter sampling event. To estimate summer season total mercury concentrations (3.4 ng/L in the water column; 254.1 µg/kg in the sediment), the average observed ratio of summer to winter water column concentrations at SL-3 (based on an average of SL-3 and its duplicate, SL-3D) and SL-8 (average of [4.60 ng/L ÷ 6.0 ng/L; 54.0 ng/L ÷ 23.9 ng/L] = ratio of 1.51 ; Table D-68) and sediment concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69) are applied to the winter concentrations. To estimate summer season methylmercury concentrations at SL-6 (0.737 ng/L in the water column; 0.408 µg/kg in the sediment), the average observed ratio of summer to winter water column concentrations at SL-3 (0.536 ng/L ÷ 0.157 ng/L = ratio of 3.41; Table D-68) and sediment concentrations observed at SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69) are applied to the winter concentrations.

Site SL-7 was originally chosen to represent water column and sediment concentrations from the Hidden Valley Wash Subwatershed. Sediment concentration data collected during the winter monitoring event indicated that this site, which is located in a stagnant backwater area at the mouth of Hidden Valley Wash, may be a “hot spot” for methylation and not representative of the subwatershed as a whole (sediment methylmercury concentrations were an order of magnitude greater than those observed at other sites in the watershed). Water column concentrations observed during the summer event confirm this assumption as methylmercury concentrations were again an order of magnitude higher than those observed at other sites in the watershed. For the summer sampling event, Site SL-7 was re-sampled and site SL-PR was added as an upstream site on Hidden Valley Wash. Both water and sediment were sampled during this event at SL-7, but water was not flowing at SL-PR, so only sediment was sampled.

Summer sediment concentrations of total and methylmercury for the Hidden Valley Wash subwatershed were assumed equal to those observed during the summer at SL-PR. Winter sediment concentrations (30.6 µg/kg total mercury; 0.003 µg/kg methylmercury) were scaled down based on the average ratio of summer to winter sediment concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69) for total mercury and at sites SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69) for methylmercury. Water column summer total mercury concentrations for Hidden Valley Wash were assumed equal to those observed during the summer event at SL-7 because 1) no data were available at SL-PR, and 2) total mercury water column concentrations were within the range of those observed at other sites in the watershed during the summer event. To estimate winter water column concentrations of total mercury at this site (7.48 ng/L), the average observed ratio of summer to winter water column concentrations at SL-3 (based on an average of SL-3 and its duplicate, SL-3D) and SL-8 (average of [4.61 ng/L ÷ 6.0 ng/L; 54.0 ng/L ÷ 23.9 ng/L] = ratio of 1.51; Table D-68) was applied. Summer (0.667 ng/L) and winter (0.374 ng/L) methylmercury water column concentrations were estimated from the total mercury concentrations assumed for each season multiplied by the fraction of mercury observed in the methyl form at other sites. For the summer methyl fraction, the average ratio observed at SL-3 and SL-8 was used (average of [0.536 ng/L ÷ 4.58 ng/L; 0.096 ng/L ÷ 54.0 ng/L] = ratio of 0.059; Table D-68); for the winter methyl fraction, the average ratio observed at SL-3 and SL-6 was used (average of [0.157 ng/L ÷ 6.00 ng/L; 0.216 ng/L ÷ 2.96 ng/L] = ratio of 0.050; Table D-68).

The assumed concentrations were applied to the runoff and sediment loads estimated from each jurisdiction within the watershed. Assumed total mercury concentrations and resulting loads are summarized in Table D-70. See example calculations in Section D.3. Results for methylmercury are presented in Table D-71. The Hidden Valley Wash subwatershed generates approximately 60 percent of the total and methylmercury loads to Lake Sherwood due to its acreage and predominance of agricultural land use relative to the other subwatersheds. Based on monitoring data collected in 2009, these loads are discharged to a stagnant, backwater area that exhibits high rates of methylation. These loads are thus greater and more bioavailable relative to other sources in the watershed.

Table D-70. Total Mercury Loads Estimated for Each Jurisdiction and Subwatershed in the Lake Sherwood Watershed

Subwatershed	Jurisdiction	Area (ac)	Summer Water Column Hg (ng/L)	Winter Water Column Hg (ng/L)	Summer Sediment Hg (µg/kg)	Winter Sediment Hg (µg/kg)	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Western	Ventura County	548	4.6	6.0	328.5	92.7	0.286	0.146	0.432
Western	SVHOA	772	4.6	6.0	328.5	92.7	1.253	1.142	2.395
Hidden Valley Wash	Thousand Oaks	40	11.3	7.5	60.3	30.6	0.027	0.003	0.031
Hidden Valley Wash	Ventura County	3,793	11.3	7.5	60.3	30.6	4.083	14.725	18.808
Near Lake Undeveloped	Thousand Oaks	70.9	4.48	2.96	254.1	129	0.019	0.024	0.043
Near Lake Undeveloped	Ventura County	126	4.48	2.96	254.1	129	0.034	0.043	0.077
Near Lake Developed	Thousand Oaks	9	54	23.9	62.9	51.0	0.020	0.001	0.021
Near Lake Developed	Ventura County	24	54	23.9	62.9	51.0	0.243	0.025	0.268
Near Lake Developed	SVHOA	310	54	23.9	62.9	51.0	4.060	0.437	4.497
Northern	Thousand Oaks	338	54	23.9	63.3	32.1	0.757	0.029	0.786
Northern	Ventura County	351	54	23.9	63.3	32.1	1.080	0.519	1.599
Northern	SVHOA	65	54	23.9	63.3	32.1	0.871	0.062	0.934
Carlisle Canyon	Caltrans	2.75	4.48	2.96	254.1	129	0.010	0.039	0.049
Carlisle Canyon	County of Los Angeles	1,149	4.48	2.96	254.1	129	0.307	0.401	0.708
Carlisle Canyon	Thousand Oaks	50.4	4.48	2.96	254.1	129	0.013	0.017	0.031
Carlisle Canyon	Ventura County	2,905	4.48	2.96	254.1	129	0.861	1.457	2.318
Carlisle Canyon	Point Mugu State Park	101	4.48	2.96	254.1	129	0.027	0.035	0.062
Total		**10,655**	**N/A**	**N/A**	**N/A**	**N/A**	**13.95**	**19.11**	**33.06**

N/A = Not applicable

Table D-71. Methylmercury Loads Estimated for Each Jurisdiction and Subwatershed in the Lake Sherwood Watershed

Subwatershed	Jurisdiction	Area (ac)	Summer Water Column MeHg (ng/L)	Winter Water Column MeHg (ng/L)	Summer Sediment MeHg (µg/kg)	Winter Sediment MeHg (µg/kg)	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Western	Ventura County	548	0.536	0.157	0.397	0.269	8.54E-03	3.83E-04	8.92E-03
Western	SVHOA	772	0.536	0.157	0.397	0.269	3.85E-02	2.92E-03	4.15E-02
Hidden Valley Wash	Thousand Oaks	40	0.672	0.370	0.009	0.003	1.37E-03	3.40E-07	1.37E-03
Hidden Valley Wash	Ventura County	3,793	0.672	0.370	0.009	0.003	2.05E-01	1.51E-03	2.07E-01
Near Lake Undeveloped	Thousand Oaks	70.9	0.737	0.216	0.408	0.136	1.52E-03	2.70E-05	1.55E-03
Near Lake Undeveloped	Ventura County	126	0.737	0.216	0.408	0.136	2.71E-03	4.82E-05	2.76E-03
Near Lake Developed	Thousand Oaks	9	0.096	0.028	0.657	0.145	2.47E-05	3.86E-06	2.85E-05
Near Lake Developed	Ventura County	24	0.096	0.028	0.657	0.145	3.06E-04	8.79E-05	3.94E-04
Near Lake Developed	SVHOA	310	0.096	0.028	0.657	0.145	5.12E-03	1.52E-03	6.64E-03
Northern	Thousand Oaks	338	0.096	0.028	0.696	0.232	9.42E-04	2.19E-04	1.16E-03
Northern	Ventura County	351	0.096	0.028	0.696	0.232	1.35E-03	3.91E-03	5.26E-03
Northern	SVHOA	65.08	0.096	0.028	0.696	0.232	1.10E-03	4.81E-04	1.58E-03
Carlisle Canyon	Caltrans	2.75	0.737	0.216	0.408	0.136	8.40E-04	4.36E-05	8.84E-04
Carlisle Canyon	County of Los Angeles	1,149	0.737	0.216	0.408	0.136	2.46E-02	4.45E-04	2.51E-02
Carlisle Canyon	Thousand Oaks	50.4	0.737	0.216	0.408	0.136	1.08E-03	1.92E-05	1.10E-03
Carlisle Canyon	Ventura County	2,905	0.737	0.216	0.408	0.136	6.92E-02	1.62E-03	7.08E-02
Carlisle Canyon	Point Mugu State Park	101	0.737	0.216	0.408	0.136	2.16E-03	3.84E-05	2.20E-03
Total		**10,655**	**N/A**	**N/A**	**N/A**	**N/A**	**0.36**	**0.01**	**0.38**

N/A = Not applicable

D.13 References

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Appendix E. Pollutant Loading from Atmospheric Deposition

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E.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure E-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board) . Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, dieldrin, DDT, PCBs, and trash. These impairments are typically associated with pollutant loading from various sources, one of which may be atmospheric deposition.

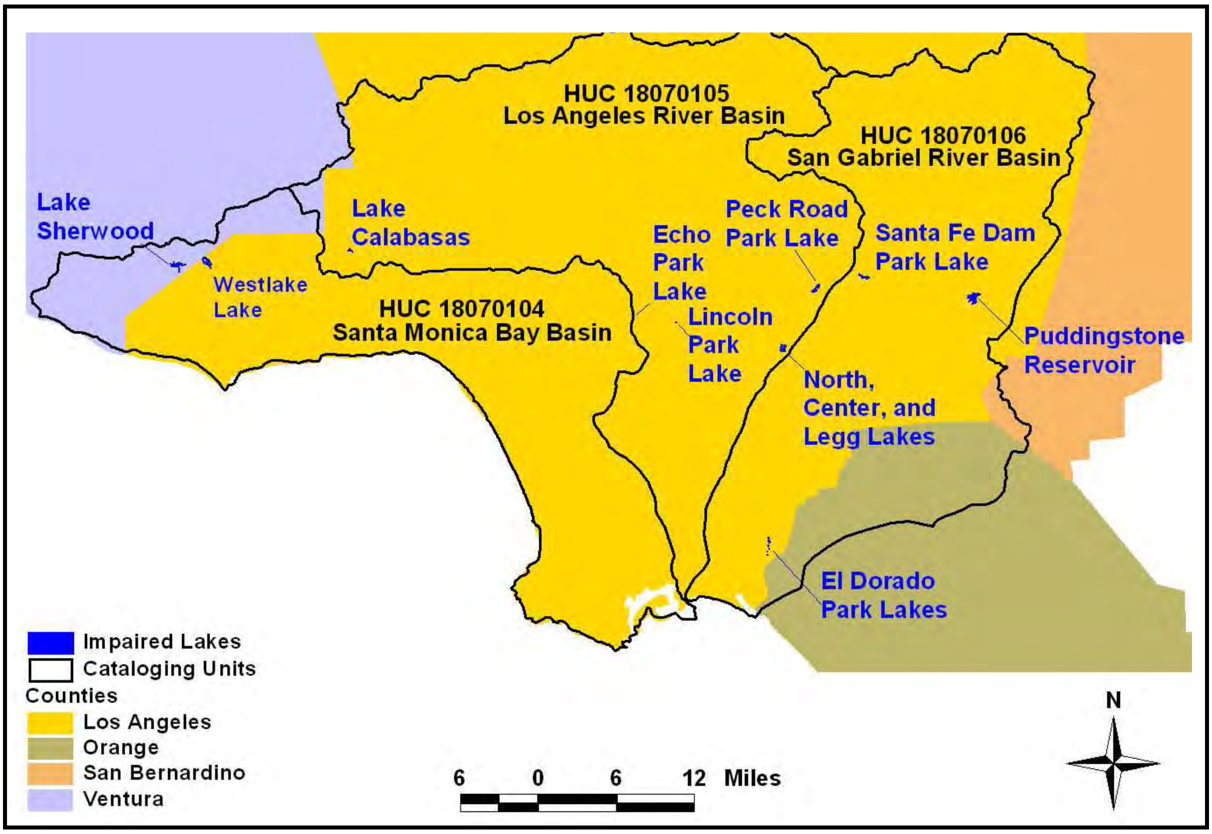

Figure E-1. Location of Impaired Lakes

Atmospheric deposition of pollutants may occur as either wet deposition (associated with precipitation) or dry deposition (associated with particulates). Wet deposition of nitrate, sulfate, and mercury are monitored nationally by the National Atmospheric Deposition Program (NADP) and the Mercury Deposition Network (MDN). Dry deposition of these parameters is less frequently monitored. Pollutants such as Organochlorine (OC) Pesticides and PCBs s have been studied regionally.

This Appendix summarizes the monitoring data, modeling efforts, and regional studies available to estimate pollutant loading from atmospheric deposition to the water surfaces of the lakes addressed by this TMDL.

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E.2 Phosphorus Deposition

Eight lakes shown in Figure E-1 (all except Lake Sherwood and Westlake Lake) have impairments addressed by this TMDL report that may be due to excessive nutrient loading. A potential source of phosphorus loading to a lake surface is atmospheric deposition. However, phosphorus does not have a significant gaseous phase, and atmospheric deposition is primarily due to fugitive dust. Phosphorus deposition rates are typically much lower than other pollutant deposition rates and are not included in the NADP monitoring program.

Currently, direct measurements of phosphorus deposition rates in Southern California are not available. Given the likelihood that direct deposition of phosphorus to a waterbody is insignificant relative to other sources of loading, the nutrient TMDLs for these eight lakes will assume zero phosphorus loading from atmospheric deposition. The Southern California Coastal Water Research Project (SCCWRP) has recently begun a deposition monitoring study that will measure phosphorus, but the results are not expected to be published until 2011. If this study indicates that atmospheric deposition of phosphorus is a significant source of phosphorus to waterbodies in the region, the nutrient TMDLs may be amended to reflect these data.

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E.3 Nitrate Deposition

The National Atmospheric Deposition Program (NADP) monitors wet nitrate and sulfate deposition at two active and two inactive stations in southern California (Figure E-2). [Though site CA94 is also a NADP site, the period of record is not sufficient to assess nitrate trends with time.] Originally, data from these stations were to be combined to develop a regression equation that could be used to predict annual precipitation-weighted nitrate concentrations and sulfate at each impaired lake. Table E-1 lists the NADP monitoring stations, elevations, and periods of record used for the analysis.

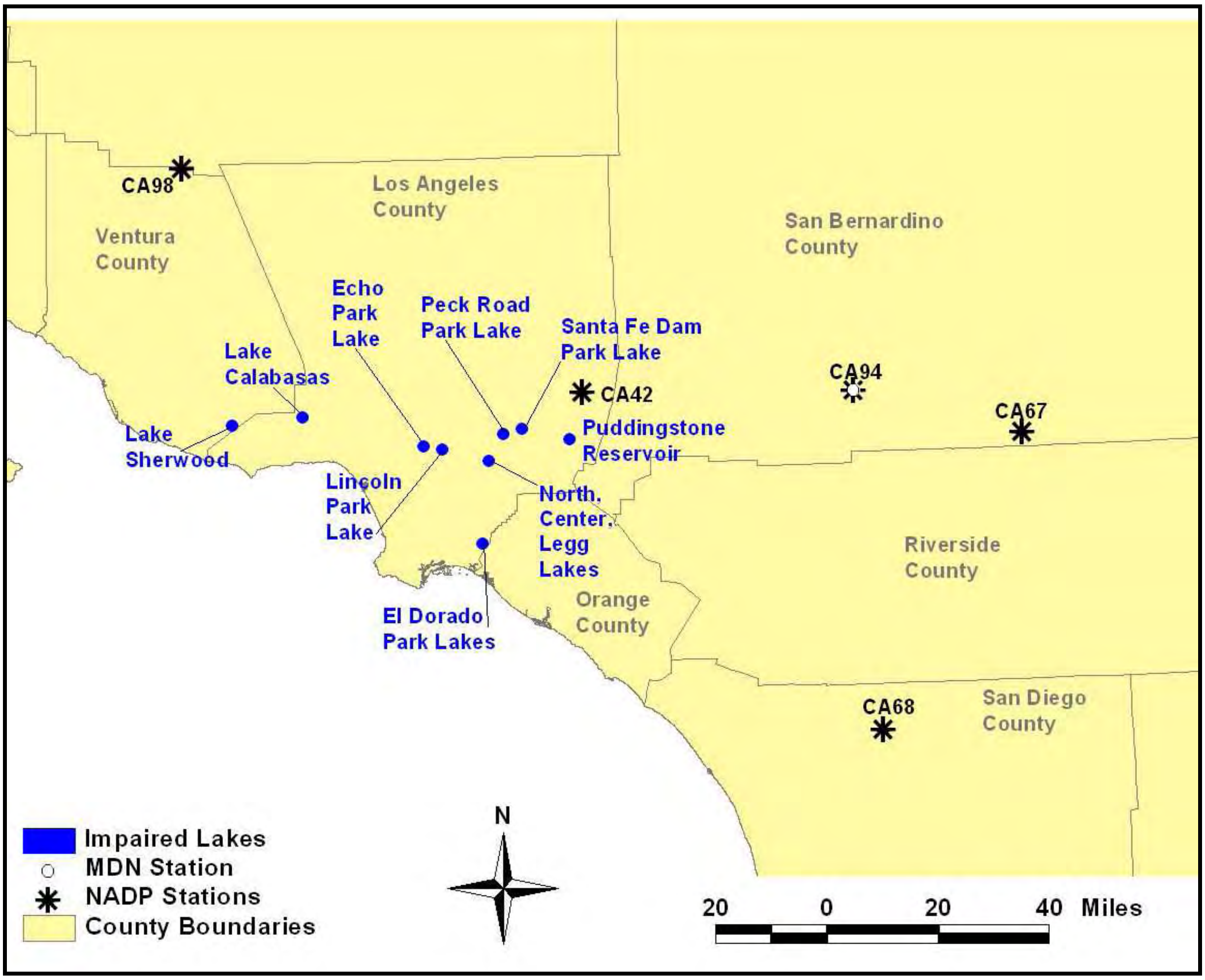

Figure E-2. Location of NADP Monitoring Stations

Table E-1. NADP Stations Used to Develop Nitrate Regression Based on Elevation and Year

ID	Name	Period of Record	Elevation (m)
CA42	Tanbark Flat	January 1982 to February 2008	853
CA67	Joshua Tree	September 2000 to February 2008	1,239
CA68	Palomar Mountain	March 1983 to January 1988	1,695
CA98	Chuchupate Ranger Station	March 1983 to January 1996	1,614

Figure E-3 shows the annual precipitation-weighted nitrate concentrations at the four sites used to develop the regression analysis. At each of the four stations, concentrations of nitrate show a decreasing trend with time.

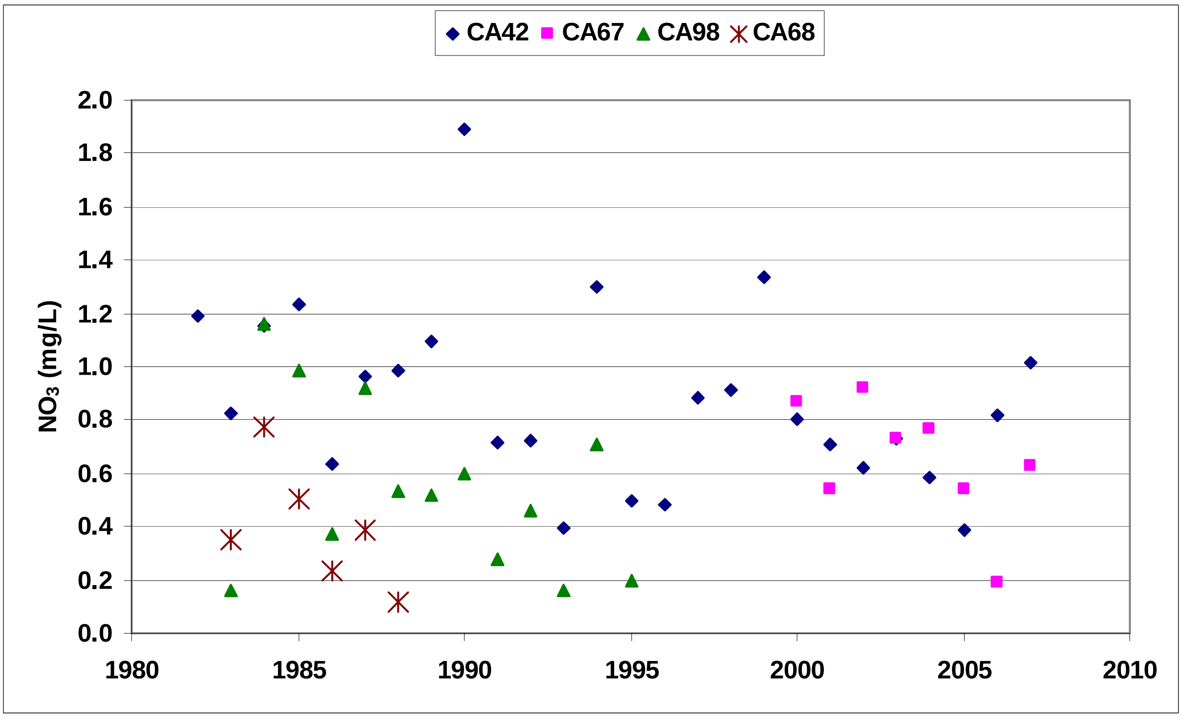

Figure E-3. Annual Precipitation-Weighted Nitrate Concentrations at Four Locations in Southern California

The regression analysis combining the elevation of each station along with year resulted in the following equation for predicting annual precipitation-weighted nitrate concentrations:

$$LOG_{10}\ (NO_3,\ mg/L) = 88.56 - 25.82\ LOG_{10}\ (Year) - 1.167\ LOG_{10}\ (Elevation,\ m),\ R^2 = 31.2\%$$

In the past, Tetra Tech has used this regression approach to estimate nitrate concentrations at varying elevations for TMDLs developed in Colorado, Arizona, and elsewhere in California. Unfortunately, the elevations of the impaired lakes addressed by this TMDL, ranging from 7 meters to 293 meters, are significantly less than the elevations of the four NADP stations available for developing the regression (853 meters to 1,695 meters). The predicted nitrate concentrations over the range of elevations of the impaired lakes are therefore significantly overestimated. Figure E-4 shows the predicted nitrate concentrations, respectively, for an example year (2000) over a range of elevations.

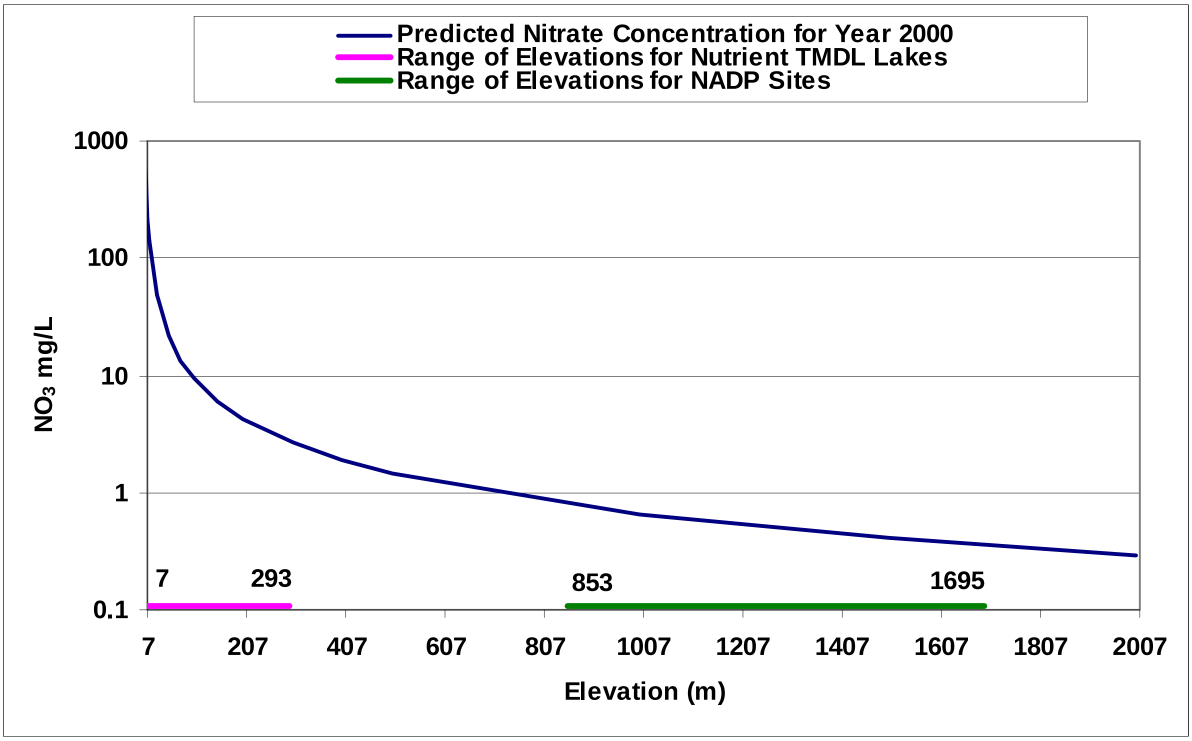

Figure E-4. Predicted Precipitation-Weighted Nitrate Concentrations for the Year 2000

As an alternative approach for predicting annual precipitation-weighted nitrate concentrations for the impaired lakes, Tetra Tech downloaded geospatial annual isopleth maps published by NADP and extracted the nitrate concentrations for grid cells overlaying each lake. NADP has produced the isopleth maps for 1994 to 2006. Tetra Tech extended the time series to previous years to correspond with available LSPC model output (Appendix D, Wet Weather Loading) for other source load estimates by developing a regression equation for each location based on year and cumulative precipitation. Table E-2 presents the annual precipitation-weighted concentrations measured (1994 to 2006) or estimated (1983 to 1993) at each lake. Although this TMDL does not address nutrient impairments at Lake Sherwood, the nitrate analysis is relevant for the mercury wet deposition estimates (Section E.4).

Table E-2. Annual Precipitation-Weighted Nitrate Concentrations (mg-NO_3/L)

Year	Peck Road Park Lake	Lincoln Park Lake	Echo Park Lake	Lake Calabasas	El Dorado Park Lakes	Puddingstone Reservoir	Legg, Center, and North Lakes	Santa Fe Dam Park Lake	Lake Sherwood
1983	0.73	0.74	0.73	0.64	1.07	1.10	0.79	0.74	0.58
1984	1.29	1.30	1.29	1.29	1.41	1.54	1.44	1.03	1.24
1985	1.33	1.25	1.25	1.21	1.38	1.49	1.44	1.06	1.17
1986	1.11	1.04	1.03	1.01	1.23	1.32	1.26	0.94	0.97
1987	1.19	1.22	1.22	1.13	1.30	1.34	1.35	0.99	1.09
1988	1.14	1.16	1.16	1.09	1.27	1.31	1.29	0.97	1.05
1989	1.30	1.28	1.28	1.26	1.34	1.40	1.41	1.06	1.22
1990	1.20	1.20	1.19	1.18	1.25	1.30	1.27	1.01	1.15
1991	0.98	0.98	0.97	0.99	1.08	1.10	1.09	0.89	0.96
1992	0.84	0.79	0.79	0.73	0.95	0.90	0.95	0.81	0.70
1993	0.75	0.75	0.75	0.78	0.87	0.83	0.83	0.76	0.75
1994	1.30	1.29	1.29	1.27	1.29	1.30	1.30	1.30	1.26
1995	0.49	0.48	0.47	0.35	0.48	0.49	0.49	0.49	0.29
1996	0.48	0.48	0.48	0.47	0.48	0.48	0.48	0.48	0.47
1997	0.89	0.88	0.88	0.87	0.88	0.89	0.89	0.89	0.86
1998	0.92	0.92	0.92	0.91	0.92	0.92	0.95	0.92	0.91
1999	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.32
2000	0.80	0.80	0.80	0.79	0.80	0.80	0.80	0.80	0.78
2001	0.71	0.71	0.71	0.72	0.71	0.71	0.71	0.71	0.73
2002	0.62	0.62	0.62	0.63	0.63	0.62	0.62	0.63	0.63
2003	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73
2004	0.58	0.59	0.59	0.59	0.59	0.58	0.58	0.58	0.58
2005	0.39	0.39	0.39	0.39	0.39	0.39	0.39	0.39	0.39
2006	0.81	0.81	0.81	0.81	0.81	0.81	0.81	0.81	0.80

Table E-3 lists the surface areas and annual precipitation assumed for each impaired lake. A discussion of the weather stations used to estimate annual precipitation at each lake is discussed in Appendix D (Wet Weather Loading).

The annual direct deposition load to a water surface depends on the amount of precipitation, the lake surface area, and the precipitation-weighted nitrate concentration measured or estimated for that year. For example, the nitrogen load deposited to the surface of Peck Road Park Lake in 1983 may be estimated as follows:

1) Convert the units of the precipitation-weighted nitrate concentration for 1983 from NO_3 to N.

$$\frac{0.73mg-NO_3}{L}\cdot\frac{1mmolNO_3}{62mgNO_3}\cdot\frac{1mmolN}{1mmolNO_3}\cdot\frac{14mgN}{mmolN}=\frac{0.165mgN}{L}$$

2) Estimate the volume of precipitation to the lake surface in 1983.

$$41.2in\cdot 87.4ac\cdot\frac{1ft}{12in}=300.07ac-ft$$

3) Multiply concentration by volume to calculate load.

$$\frac{0.165mgN}{L}\cdot 300.7ac-ft\cdot\frac{43{,}560ft^2}{1ac}\cdot\frac{28.32L}{1ft^3}\cdot\frac{g}{1000mg}\cdot\frac{1lb}{453.6g}=134.5lbN$$

Table E-4 presents the average nitrogen load to each lake due to atmospheric deposition.

Table E-3. Annual Precipitation (inches) and Surface Area of Impaired Lakes

Year	Peck Road Park Lake (87.4 ac)	Lincoln Park Lake (4.9 ac)	Echo Park Lake (14.1 ac)	Lake Calabasas (17.8 ac)	El Dorado Park Lakes (35.3 ac)	Puddingstone Reservoir (252.4 ac)	Legg, Center, and North Lakes (76.6 ac)	Santa Fe Dam Park Lake (70.6 ac)	Lake Sherwood (136.8 ac)
1983	41.2	34.0	34.0	41.0	26.7	37.7	41.1	39.1	41.0
1984	11.9	8.9	8.9	6.9	8.5	10.1	10.6	11.9	6.9
1985	8.4	9.7	9.7	9.8	8.2	10.2	8.9	8.4	9.8
1986	18.6	18.1	18.1	19.1	13.5	18.1	15.7	18.6	19.1
1987	13.4	9.1	9.1	12.0	8.2	14.7	10.0	13.3	12.0
1988	14.7	10.7	10.7	13.1	8.1	13.9	11.2	14.4	13.1
1989	5.7	4.5	4.5	3.6	3.0	6.1	4.4	5.4	3.6
1990	9.3	7.2	7.2	6.4	5.7	9.6	9.5	9.2	6.4
1991	19.7	15.7	15.7	15.6	12.2	18.9	16.1	19.4	15.6
1992	25.7	22.8	22.8	27.7	16.9	28.2	20.8	25.6	27.7
1993	28.9	23.5	23.5	24.2	19.2	30.3	25.0	29.6	24.2
1994	11.0	8.7	8.7	10.1	8.4	11.3	10.4	11.0	10.1
1995	30.6	23.7	23.7	26.6	21.5	28.2	26.5	30.6	26.6
1996	25.0	17.4	17.4	20.1	14.9	24.3	23.3	23.8	20.1
1997	12.8	10.2	10.2	14.8	12.4	17.5	14.9	6.9	14.8
1998	31.2	27.3	27.3	31.5	24.1	32.2	31.8	31.2	31.5
1999	7.8	8.0	8.0	7.5	6.8	7.8	6.8	7.5	7.5
2000	16.1	12.0	12.0	12.8	8.5	12.6	13.6	15.7	12.8
2001	25.6	17.0	17.0	29.5	4.7	21.1	17.0	20.8	29.5
2002	8.9	7.3	7.3	9.5	2.7	8.7	7.6	8.8	9.5
2003	16.8	13.4	13.4	14.9	9.1	16.8	15.3	16.8	14.9
2004	24.4	19.8	19.8	19.9	14.9	21.1	20.5	24.4	19.9
2005	36.3	25.6	25.6	29.6	14.2	6.4	24.0	36.3	29.6
2006	14.9	11.6	11.6	14.0	7.8	11.8	11.9	14.9	14.0
Average	**19.1**	**15.3**	**15.3**	**17.5**	**11.7**	**17.4**	**16.5**	**18.5**	**17.5**

Table E-4. Annual Nitrogen Load (lb) from Atmospheric Deposition to Impaired Lakes

Year	Peck Road Park Lake	Lincoln Park Lake	Echo Park Lake	Lake Calabasas	El Dorado Park Lakes	Puddingstone Reservoir	Legg, Center, and North Lakes	Santa Fe Dam Park Lake	Lake Sherwood
1983	134.5	6.3	17.9	23.9	51.6	535.6	127.3	104.5	166.5
1984	68.7	2.9	8.3	8.1	21.6	200.9	59.8	44.3	59.9
1985	50.0	3.0	8.7	10.8	20.4	196.3	50.2	32.2	80.3
1986	92.3	4.7	13.5	17.6	30.0	308.6	77.5	63.2	129.7
1987	71.3	2.8	8.0	12.4	19.3	254.4	52.9	47.6	91.6
1988	74.9	3.1	9.0	13.0	18.6	235.2	56.6	50.5	96.3
1989	33.1	1.4	4.2	4.1	7.3	110.3	24.3	20.7	30.7
1990	49.9	2.2	6.2	6.9	12.9	161.2	47.3	33.6	51.5
1991	86.3	3.9	11.0	14.1	23.8	268.5	68.8	62.4	104.8
1992	96.5	4.5	13.0	18.4	29.0	327.8	77.5	74.9	135.7
1993	96.9	4.4	12.7	17.2	30.2	324.8	81.3	81.3	127.1
1994	64.0	2.8	8.1	11.7	19.6	189.7	53.0	51.7	89.1
1995	67.1	2.9	8.0	8.5	18.6	178.5	50.9	54.2	54.0
1996	53.7	2.1	6.0	8.6	12.9	150.6	43.8	41.3	66.1
1997	50.9	2.3	6.5	11.7	19.7	201.2	52.0	22.2	89.1
1998	128.4	6.3	18.1	26.1	40.0	382.6	118.4	103.7	200.7
1999	46.4	2.7	7.7	9.1	16.3	134.0	35.4	36.0	69.3
2000	57.6	2.4	6.9	9.2	12.3	130.2	42.6	45.4	69.9
2001	81.3	3.0	8.7	19.3	6.0	193.5	47.3	53.4	150.7
2002	24.7	1.1	3.3	5.5	3.1	69.7	18.5	20.0	41.9
2003	54.8	2.5	7.1	9.9	12.0	158.4	43.8	44.3	76.1
2004	63.3	2.9	8.4	10.7	15.9	158.1	46.6	51.1	80.8
2005	63.3	2.5	7.2	10.5	10.0	32.2	36.7	51.1	80.8
2006	54.0	2.4	6.8	10.3	11.4	123.4	37.8	43.6	78.4
Average	**69.3**	**3.1**	**9.0**	**12.4**	**19.3**	**209.4**	**56.3**	**51.4**	**92.5**

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E.4 Mercury Deposition

Mercury deposition from the atmosphere to the earth's surface may occur in several forms: gaseous elemental mercury (Hg(0)), divalent ionic mercury (Hg(II)), reactive gaseous mercury (RGM), and aerosol particulate mercury (Hg-P). Atmospheric deposition can be divided into short-range or near-field deposition, which includes deposition from sources located near the watershed, and long-range or far-field deposition, which includes mercury deposition from regional and global sources. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport.

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). There is one MDN site in Southern California, but it has only been active since May of 2006. The rates of wet mercury deposition to each lake water surface were estimated with a regression approach that utilizes nitrate and sulfate wet deposition data collected by the National Atmospheric Deposition Program (NADP) along with mercury wet deposition data collected by the MDN.

Dry deposition is more difficult to monitor and less localized data are available to estimate this component. To estimate loading from dry deposition, grid-cell output from regional deposition models developed by USEPA was obtained for each lake impaired by mercury.

To evaluate potential near-field sources at each impaired lake, the USEPA Toxics Release Inventory (TRI) was used to determine the proximity of point sources that may contribute to airborne mercury loads including coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories.

Precipitation events following recent forest fires also result in increased loads of total and methylmercury from the watershed and release of elemental mercury to the atmosphere which is then available for deposition.

E.4.1. Near Field Sources of Atmospheric Mercury

Major atmospheric point sources of mercury can cause locally elevated areas of near-field atmospheric deposition downwind. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport. Reactive gaseous mercury and particulate mercury are also associated with manmade sources and typically deposit within approximately 100 miles of the source.

Significant potential near-field emission sources of airborne mercury include coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories. Emissions from such sources are summarized in USEPA's Toxic Release Inventory (TRI). Facilities that reported emissions of mercury in southern California in 2007 to the USEPA (2009) within 100 miles of the El Dorado Park lakes, Puddingstone Reservoir, or Lake Sherwood are shown in Figure E-5, Figure E-6, and Figure E-7, respectively. Emissions data for 2008 have not yet been released.

Table E-5 summarizes the loads reported from each facility in the 2007 TRI within 100 miles of either of these three waterbodies. Thirty-five out of 64 facilities listed in the database reported zero pounds of mercury released in 2006 (these are not included in the table); 19 reported emissions less than 10 pounds per year. Four of the top five sources of mercury emissions were due to cement manufacturing facilities; one of the top five is an oil refinery. Total reported mercury air emissions in 2007 within 100 miles of these three mercury impaired lakes were 1,043 pounds.

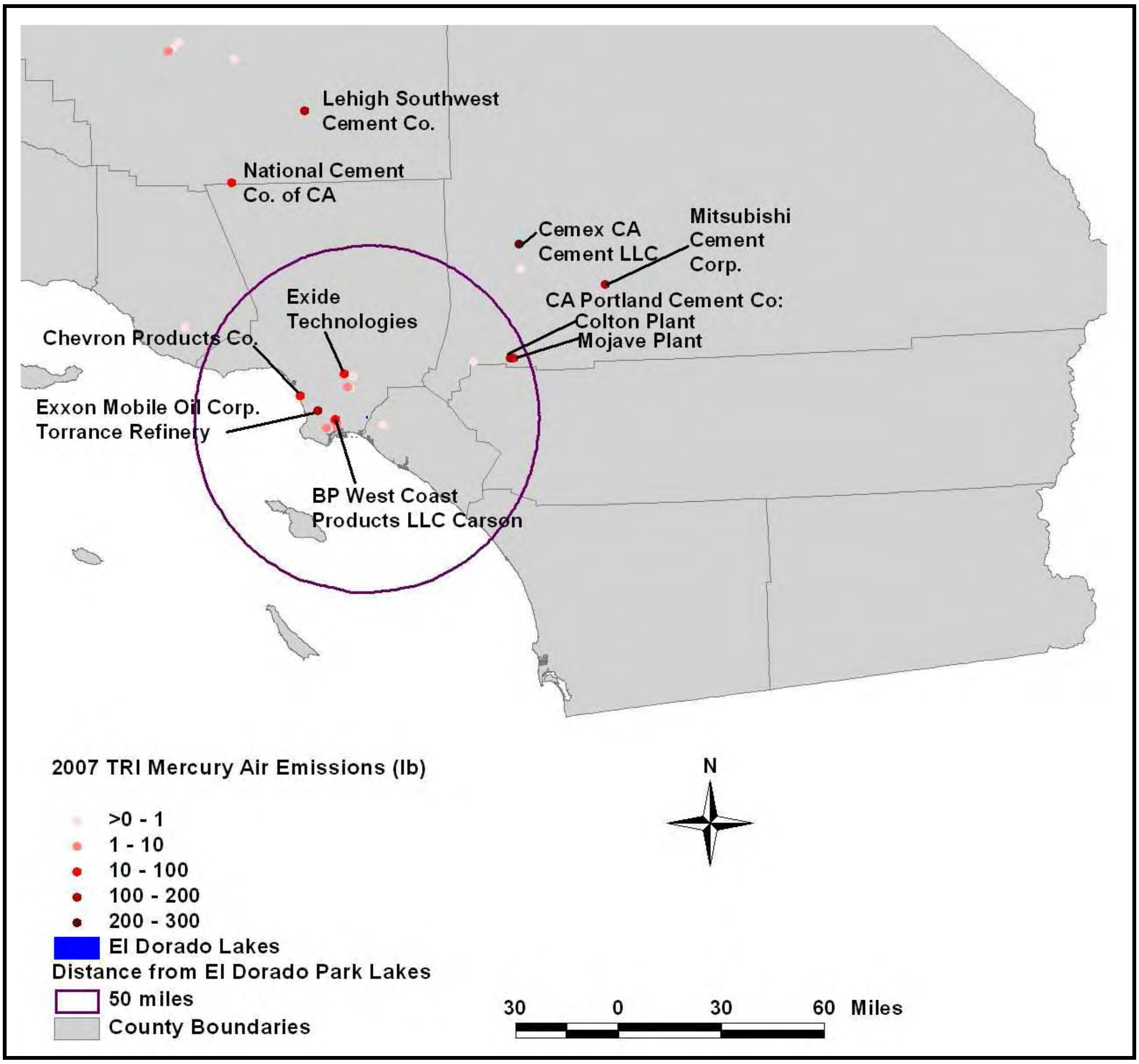

Figure E-5. Location of Facilities Reporting Mercury Emissions within 50 miles of the El Dorado Park Lakes

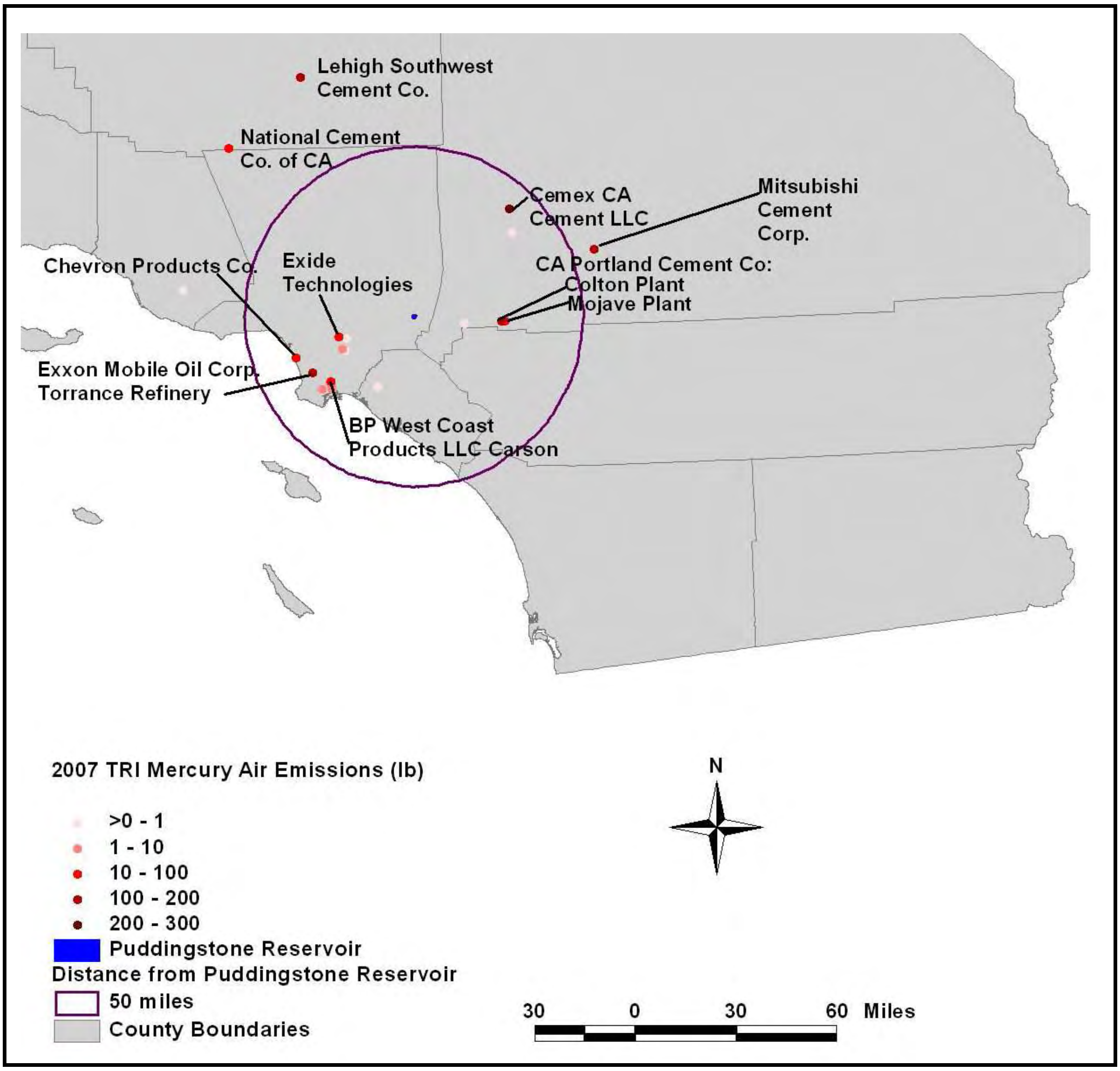

Figure E-6. Location of Facilities Reporting Mercury Emissions within 50 miles of Puddingstone Reservoir

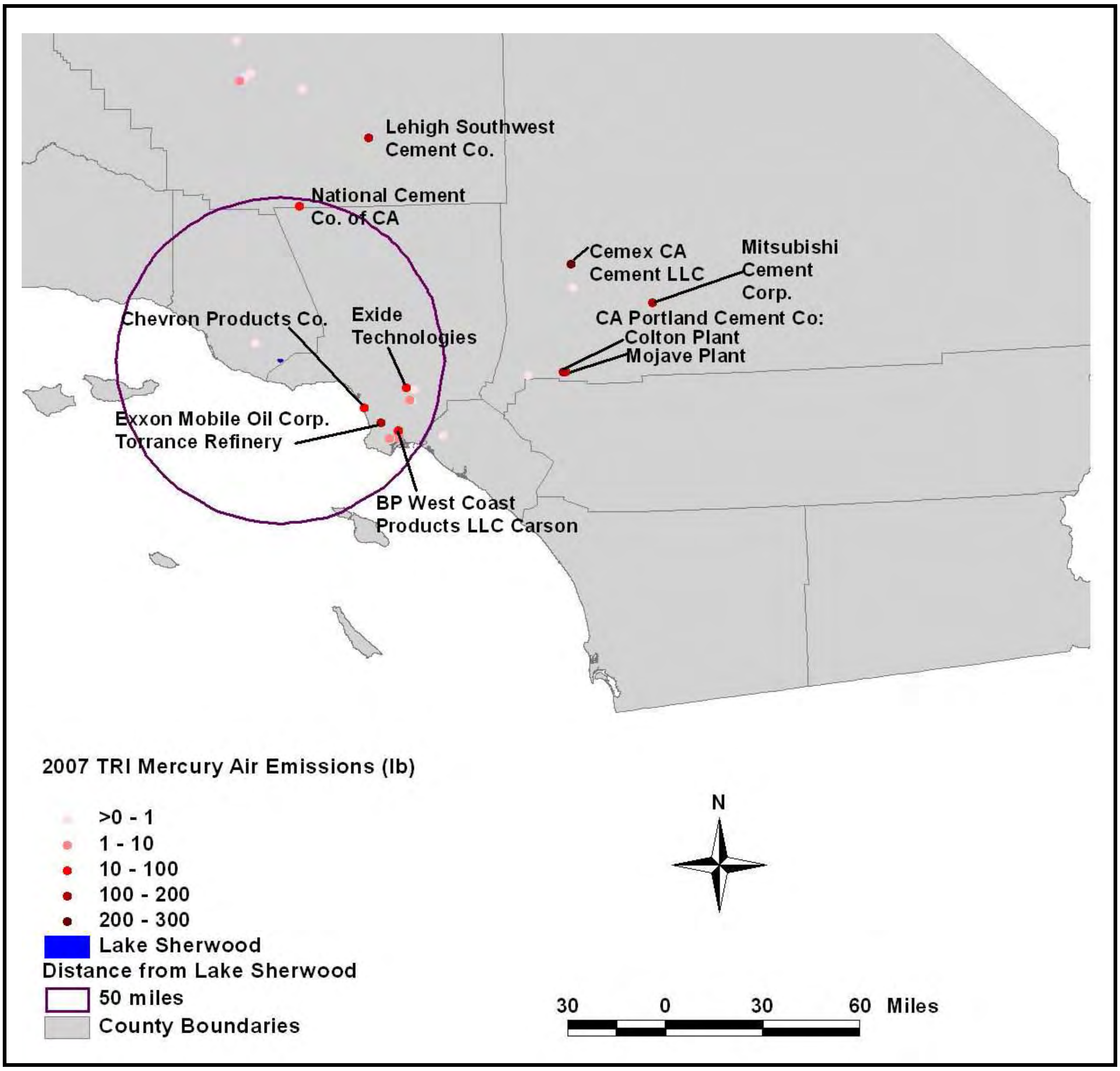

Figure E-7. Location of Facilities Reporting Mercury Emissions within 50 miles of Lake Sherwood

Table E-5. Mercury Emissions Reported in the 2007 USEPA Toxic Release Inventory

Facility Name	Total Air Emissions (lbs)
Cemex California Cement LLC	273.30
Exxon Mobil Oil Corp - Torrance Refinery	162.70
Mitsubishi Cement Corp.	160.00
Lehigh Southwest Cement Co.	144.12
California Portland Cement Co. Colton Plant	124.61
National Cement Co Of California Inc	55.00
Exide Technologies	51.74
Bp West Coast Products LLC Carson	17.10
Chevron Products Co. Div Of Chevron USA Inc.	14.90
California Portland Cement Co. Mojave Plant	13.40
Ultramar Inc. Wilmington Refinery	4.51
Conoco Phillips Co La Refinery Wilmington Plant	3.80
Tesoro Los Angeles Refinery	3.40
Arnco	3.00
Conoco Phillips Co Los Angeles Refinery Carson Plant	2.30
Big West Of California Refinery	1.90
Mt. Poso Cogeneration	1.41
Commerce Refuse-To-Energy Facility	1.29
Rio Bravo Poso	0.96
Rio Bravo Jasmin	0.86
GHN Neon Inc	0.74
Tin, Inc DBA Temple Inland	0.60
Lunday-Thagard Co.	0.53
Alltech Associates Inc.	0.50
San Joaquin Refining Co Inc.	0.33
Big West Of California Refinery	0.30
Teledyne Imaging Sensors	0.20
Tricor Refining LLC	0.01
GS Roofing Products Co Inc. (DBA Certainteed)	0.01
Total	**1,043.52**

E.4.2. SIMULATED MERCURY DEPOSITION RATES

USEPA has undertaken several national-scale modeling efforts to characterize mercury deposition. For the 1997 Report to Congress, USEPA developed the Regional Lagrangian Model of Air Pollution (RELMAP) modeling (USEPA, 1997, Section 5.1.3) to produce gridded estimates of deposition rates. The report included comparisons between wet deposition of mercury from local anthropogenic sources and a global-scale background concentration. While the RELMAP modeling is now believed to be outdated and does not fully reflect the current state of understanding of atmospheric chemistry leading to deposition of mercury (personal communication, O. Russell Bullock, USEPA, to J. B. Butcher, Tetra Tech, July 25, 2001), these results suggested that the deposition of mercury in the southwest has a strong global or long-range component.

The RELMAP modeling had considerable uncertainty, particularly for the Southwest, where monitoring data were scarce and dry deposition of mercury may play a larger role. The broad-scale RELMAP modeling also could not take into account the effects of local topography on deposition, nor did it account for the interaction of chloride ions in power plant emissions with elemental mercury to form species such as mercuric chloride that are subject to more rapid deposition. USEPA subsequently developed a more sophisticated regional mercury transport model (Community Multiscale Air Quality (CMAQ-Hg)) based on the Models-3/CMAQ system (Byun and Ching, 1999), which incorporated a more sophisticated representation of mercury chemistry. In support of the Clean Air Mercury Rule, the CMAQ-Hg model was used to predict mercury deposition for the 2001 base case on a 36x36 km model grid (USEPA, 2005). The baseline scenario was used to estimate wet and dry mercury deposition rates.

The CMAQ 2001 analysis was also conducted with US power plant emissions set to zero. Wet and dry rates of deposition were not distinguished in the output supplied to Tetra Tech. In most of the southwest region of the US, turning off US power plants in the model did not significantly impact the rate of total mercury deposition (see the bottom row of Table E-6). Simulated mercury deposition rates for the CMAQ grid cells that contain each impaired lake are summarized in Table E-6.

Table E-6. CMAQ 2001 Output for Grid Cells Underlying the Watersheds of the Mercury Impaired Lakes

	Mercury Deposition Rate g/km^2/yr		
Component	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Wet – Baseline	9.6988	4.1082	2.9007
Dry – Baseline	77.5962	29.8365	12.1748
Total – Baseline	87.2950	33.9447	15.0755
Total – Zero Power Plant Emissions	87.2822	33.9293	15.0682

An additional run of the CMAQ model was undertaken for 2002 meteorological conditions, with alterations to the functional description of processes leading to the dry deposition of mercury. The 2002 CMAQ results are summarized in Table E-7. At the El Dorado Park lakes, the CMAQ 2001 simulation predicts higher rates of both wet and dry deposition, and the total deposition rate is approximately 44 percent higher than the 2002 simulation results. For Puddingstone Reservoir, the 2001 simulation predicts a higher wet deposition rate, but the 2002 simulation predicts a higher dry deposition rate. The total deposition rate predicted by the 2002 simulation is approximately 11 percent higher than the 2001 simulation. At Lake Sherwood, the 2001 simulation predicts higher rates of both wet and dry deposition, and the total deposition rate is approximately 240 percent higher than the 2002 prediction. Both the 2001 and 2002 CMAQ simulations estimate that the rate of dry mercury deposition is higher than the rate of wet mercury deposition. The CMAQ 2002 results are assumed to represent a more accurate estimate of dry deposition because this model included alterations to the functional deposition of processes associated with the dry deposition of mercury. Therefore, the dry deposition is primarily based on the CMAQ 2002 results, with the exception of Lake Sherwood. For Lake Sherwood, the CMAQ 2001 dry deposition rates will be used as described in Section E.4.5).

Table E-7. CMAQ 2002 Output for Grid Cell Underlying the Watersheds of the Mercury Impaired Lakes

Component	Mercury Deposition Rate $g/km^2/yr$		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Wet	3.5642	2.5400	0.5863
Dry	57.0656	35.2323	5.6784
Total	60.6298	37.7723	6.2647

E.4.3. WET DEPOSITION MONITORING

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate or gaseous settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). In May 2006, a MDN station was installed at Converse Flats, California in San Bernardino County. Quality-assured data are available from the MDN website through December 2007; provisional data were provided to Tetra Tech through December 2008.

Figure E-8 through Figure E-10 show the measurements of precipitation, mercury concentration, and mercury deposition at Converse Flats. Points connected by lines indicate successive weeks with measured precipitation and mercury wet deposition measurements. Single points indicate that no precipitation fell the week prior or the week after. Weekly precipitation measurements range from 0 to 130 mm (0 to 5.1 inches). The average observed mercury concentration during precipitation events is 18.5 ng/L, and the volume-weighted average concentration is 11.2 ng/L. Weekly deposition rates measured at Converse Flats range from 0 to 1,442 ng/m^2, and the average annual deposition rate, including weeks with zero precipitation, is 3.48 $g/km^2/yr$.

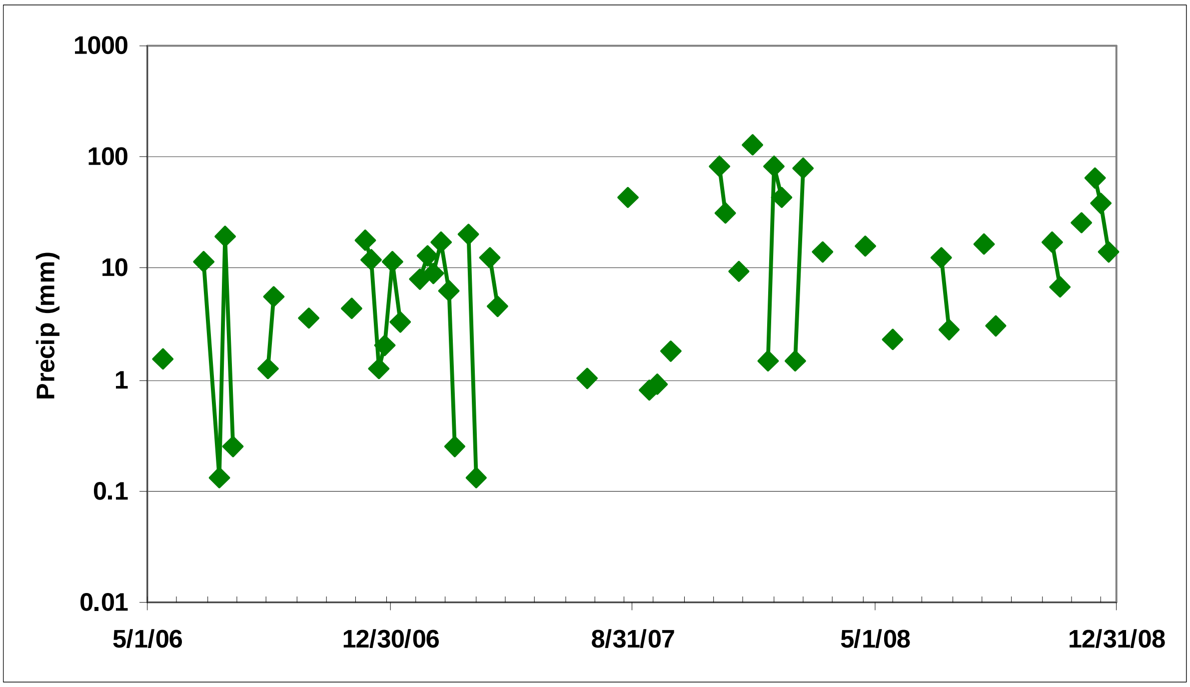

Figure E-8. Weekly Precipitation Measurements at CA94

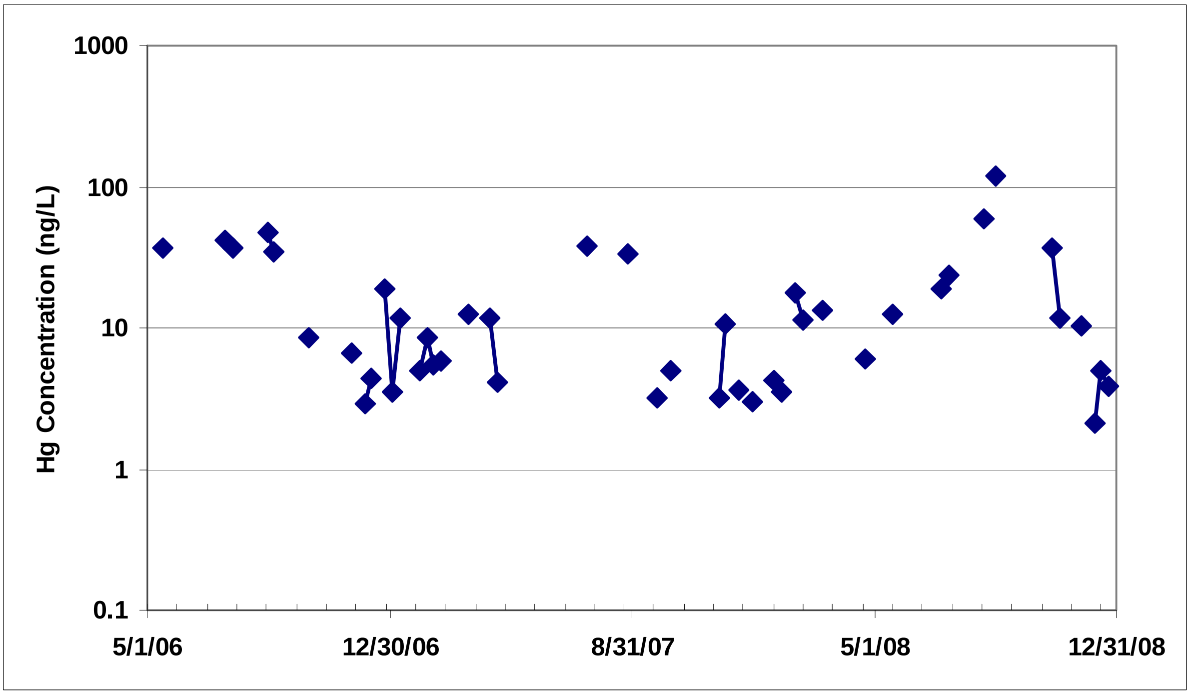

Figure E-9. Weekly Mercury Concentrations at CA94

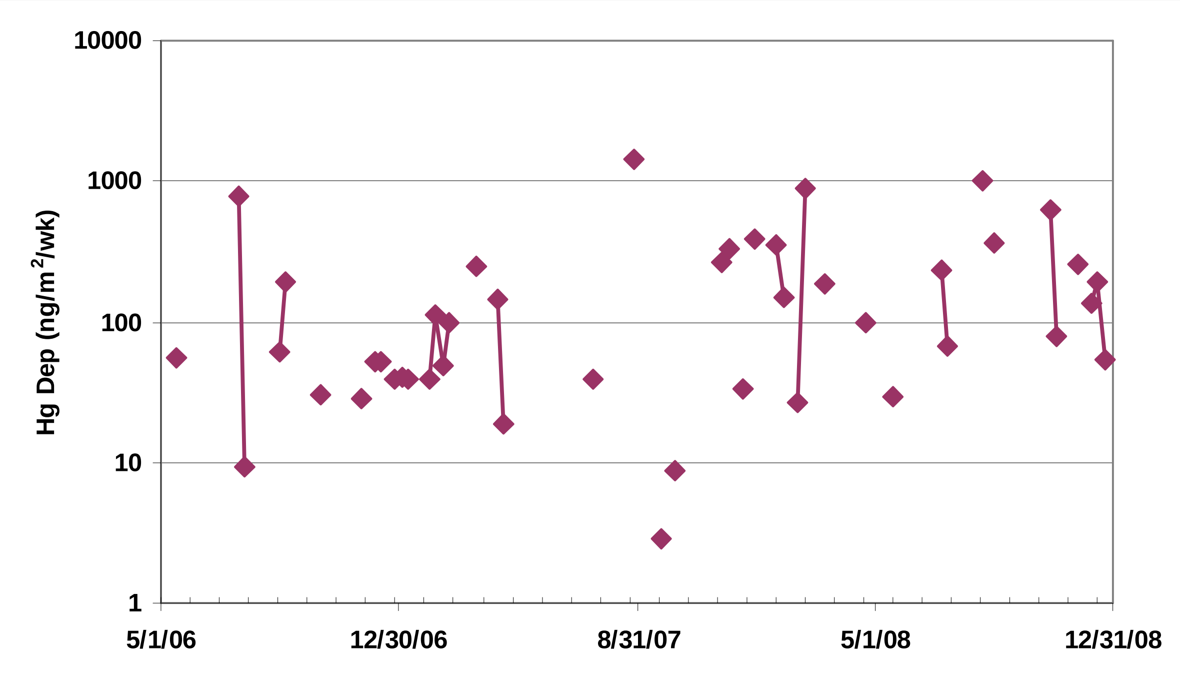

Figure E-10. Weekly Mercury Wet Deposition Rates at CA94

E.4.4. Wet Deposition Estimation

MDN station CA94 (Converse Flats) was installed in May 2006 to support development of mercury TMDLs in Southern California. During the period of record, the average annual wet deposition rate is 3.48 g/km^2/yr. In addition to mercury concentrations, this site also monitored nitrate and sulfate wet deposition concentrations through the National Atmospheric Deposition Program (NADP). Deposition of particulate and reactive gaseous mercury derived from combustion sources is often correlated with nitrate and sulfate deposition. A multiple regression on nitrate and sulfate deposition concentrations measured at CA94 yields an estimate of mercury concentration with an R^2 of 0.54. Figure E-11 shows a comparison of the measured and estimated mercury concentrations resulting from the following equation:

$$LOG_{10}\ (Hg,\ ng/L) = 1.2102 + 0.1285\ LOG_{10}\ (NO_3,\ mg/L) + 0.4579\ LOG_{10}\ (SO_4,\ mg/L),\ R^2 = 53.6\%$$

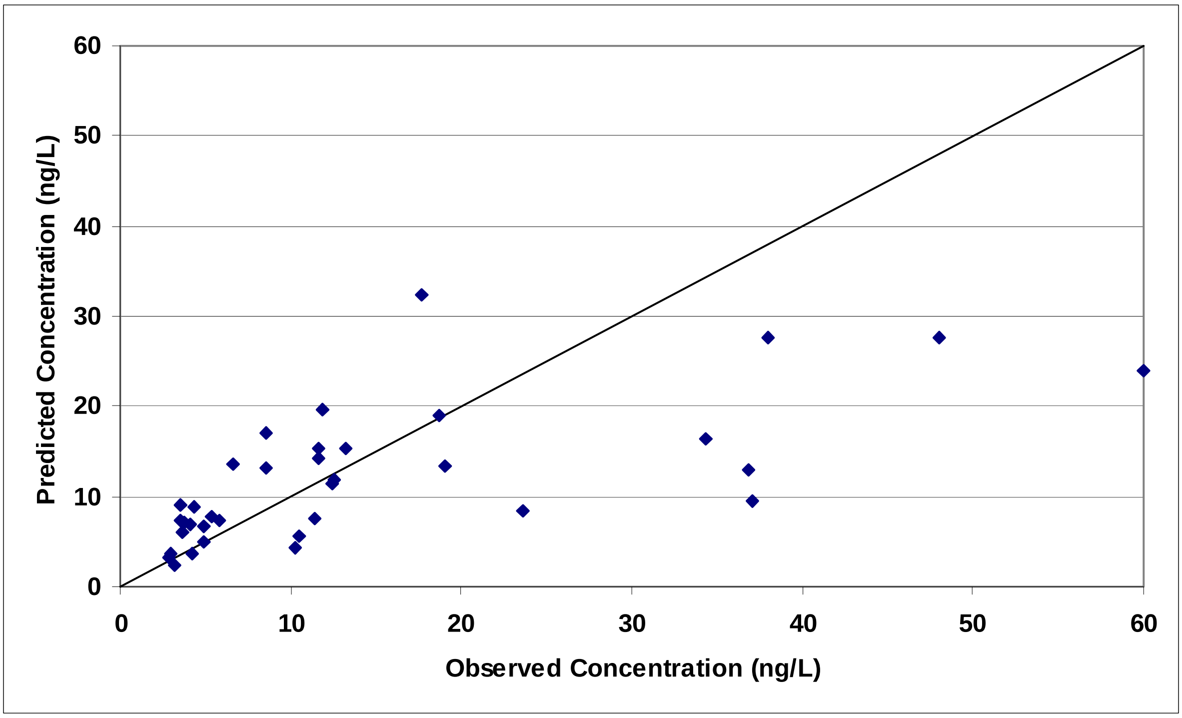

Figure E-11. Comparison of Measured and Predicted Mercury Wet Deposition Concentrations at Converse Flats

In order to use the mercury regression equation to estimate concentrations of mercury in precipitation at other locations, estimates of nitrate and sulfate concentrations are needed at each mercury impaired lake. Section E.3 explained how annual precipitation-weighted nitrate concentrations were obtained for the impaired lakes addressed by this TMDL. A similar method was used to obtain annual precipitation-weighted sulfate concentrations for the three mercury impaired lakes. Table E-8, Table E-9, and Table E-10 list the annual precipitation-weighted nitrate, sulfate, and predicted mercury concentrations for each lake and year.

Table E-8. Annual Precipitation-Weighted Concentrations at El Dorado Park Lakes

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	1.07	0.45	11.37
1984	1.41	0.56	13.03
1985	1.38	0.56	12.91
1986	1.23	0.51	12.24
1987	1.30	0.54	12.61
1988	1.27	0.53	12.47
1989	1.34	0.55	12.82
1990	1.25	0.52	12.40
1991	1.08	0.47	11.59

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1992	0.95	0.43	10.92
1993	0.87	0.40	10.49
1994	1.29	0.60	13.25
1995	0.48	0.26	7.93
1996	0.48	0.27	8.11
1997	0.88	0.42	10.72
1998	0.92	0.43	10.93
1999	1.33	0.57	12.99
2000	0.80	0.36	9.90
2001	0.71	0.39	10.08
2002	0.63	0.30	8.76
2003	0.73	0.30	9.02
2004	0.59	0.33	9.11
2005	0.39	0.30	8.26
2006	0.81	0.46	11.09

Table E-9. Annual Precipitation-Weighted Concentrations at Puddingstone Reservoir

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	1.10	0.46	11.52
1984	1.54	0.62	13.82
1985	1.49	0.61	13.64
1986	1.32	0.55	12.77
1987	1.34	0.56	12.89
1988	1.31	0.55	12.78
1989	1.40	0.59	13.28
1990	1.30	0.55	12.80
1991	1.10	0.48	11.75
1992	0.90	0.41	10.65
1993	0.83	0.38	10.21
1994	1.30	0.60	13.28
1995	0.49	0.26	8.02
1996	0.48	0.27	8.13

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1997	0.89	0.42	10.73
1998	0.92	0.43	10.93
1999	1.33	0.57	13.00
2000	0.80	0.36	9.90
2001	0.71	0.39	10.07
2002	0.62	0.29	8.63
2003	0.73	0.30	8.95
2004	0.58	0.33	9.10
2005	0.39	0.29	8.19
2006	0.81	0.46	11.10

Table E-10. Annual Precipitation-Weighted Concentrations at Lake Sherwood

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	0.58	0.29	8.57
1984	1.24	0.49	12.03
1985	1.17	0.47	11.71
1986	0.97	0.41	10.75
1987	1.09	0.45	11.38
1988	1.05	0.44	11.23
1989	1.22	0.49	12.06
1990	1.15	0.47	11.75
1991	0.96	0.42	10.81
1992	0.70	0.34	9.49
1993	0.75	0.36	9.80
1994	1.26	0.58	13.00
1995	0.29	0.19	6.42
1996	0.47	0.26	7.95
1997	0.86	0.41	10.55
1998	0.91	0.43	10.84
1999	1.32	0.56	12.93
2000	0.78	0.36	9.79
2001	0.73	0.39	10.13

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
2002	0.63	0.30	8.83
2003	0.73	0.31	9.10
2004	0.58	0.33	9.05
2005	0.39	0.30	8.32
2006	0.80	0.46	11.00

Lake surface area and annual precipitation (see Table E-3) combined with precipitation-weighted mercury concentrations provide an estimate of annual wet deposition of mercury to a lake surface. Table E-11 presents the mercury load from wet deposition calculated for each lake.

Table E-11. Mercury Load from Wet Deposition to Mercury Impaired Lakes

Year	Mercury Load from Wet Deposition (g/yr)		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
1980	1.10	11.26	4.94
1981	0.40	3.61	1.17
1982	0.39	3.61	1.61
1983	0.60	5.99	2.89
1984	0.38	4.91	1.93
1985	0.37	4.61	2.07
1986	0.14	2.11	0.61
1987	0.26	3.19	1.07
1988	0.51	5.76	2.36
1989	0.67	7.77	3.69
1990	0.73	8.02	3.33
1991	0.40	3.89	1.84
1992	0.62	5.86	2.40
1993	0.44	5.12	2.25
1994	0.48	4.88	2.20
1995	0.96	9.12	4.79
1996	0.32	2.62	1.36
1997	0.31	3.23	1.75
1998	0.17	5.51	4.20
1999	0.08	1.94	1.18

Year	Mercury Load from Wet Deposition (g/yr)		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
2000	0.30	3.91	1.91
2001	0.49	4.99	2.53
2002	0.42	1.37	3.46
2003	0.32	3.38	2.16
2004	0.453	11.26	4.94
2005	1.10	3.61	1.17
2006	0.40	3.61	1.61
Average	**0.39**	**4.86**	**2.40**

Table E-12 compares the average wet deposition rate based on monitoring data and regression analyses to the CMAQ 2001 and 2002 runs. The calculated rates are generally in agreement with the CMAQ runs with the exception of Lake Sherwood where calculated rates are 50 percent higher than the greater of the two CMAQ estimates. As discussed in Section E.4.6, the calculated wet deposition rates will be used for TMDL development; the CMAQ model runs are only presented for comparison (Note: There are only two published CMAQ model runs for consideration in the analyses and only grid-scale model output was available; therefore, additional model runs could not be performed for TMDL development).

Table E-12. Summary of Wet Deposition Estimates to Each Impaired Lake

Deposition Load	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Lake Surface Area (km^2)	0.143	1.021	0.554
Calculated Wet (g/yr)	0.453	4.86	2.40
CMAQ 2001 Wet (g/yr)	1.39	4.19	1.61
CMAQ 2002 Wet (g/yr)	0.510	2.59	0.325

Note: Shaded cells represent the selected wet deposition loads for each waterbody.

E.4.5. Dry Deposition

Although there are few direct measurements to support well-characterized estimates, dry deposition of mercury often is assumed to be approximately equal to wet deposition (e.g., Lindberg et al., 1991; Lindqvist et al., 1991). This assumption is not always valid in the southwest. Dry and wet deposition were measured in the Pecos River basin of eastern New Mexico in 1993–1994 (Popp et al., 1996). Average weekly deposition rates were calculated to be 140 ng/m^2-wk of mercury from dry deposition and 160 ng/m^2-wk of mercury from wet deposition. These data demonstrate the importance of both dry and wet deposition as sources of mercury. Early throughfall studies in a coniferous forest indicate that dry deposition beneath a forest canopy could be on the order of 50 percent of the wet deposition signal (Lindqvist et al., 1991). However, the local university cooperator at the Caballo, New Mexico MDN station (NM10) estimated dry deposition as up to six times wet deposition at this arid site (Caldwell et al., 2003). A recent study sponsored by the Arizona Department of Environmental Quality indicates that dry deposition may be two to nine times higher than wet deposition (Tetra Tech, 2008).

Atmospheric dry deposition involves three groups of mercury species: reactive gaseous mercury (RGM), aerosol particulate mercury (Hg-P), and gaseous elemental mercury (Hg(0)). All three forms may deposit to land and water surfaces, but there are significant differences in chemistry and rates. Hg(0) is the dominant species in terms of ambient concentration; however, net deposition rates are much higher for the other forms (Lindberg et al., 1992).

Dry mercury deposition to water surfaces is typically comprised of the reactive gaseous and particulate forms of mercury only. Elemental mercury contributes to the loading to land surfaces as it is accumulated in vegetation through stomatal vapor uptake (Eriksen et al., 2003). Contributions to soil systems occur as vegetative material falls and decays on the soil surface. No direct measurements of dry deposition are available for this region. As a conservative estimate, the greater of the two CMAQ simulation results (Section E.4.2) may be used to estimate the rate of dry mercury deposition to the land (direct deposition plus foliar accumulation).

The TMDL process for mercury loading generally divides loading into two components: watershed loading and direct atmospheric deposition to the water surface. Though the watershed load typically originates from atmospheric sources, whether historic, recent, near, or distant, delivery to the waterbody depends on runoff, erosion, and sedimentation processes that occur on the land surface and in the tributary network. In some cases, direct sources of mercury loading may be present in a watershed, such as mine tailings or geological formations with naturally high mercury concentrations. Watershed loading models that predict runoff and sediment delivery to a receiving waterbody are typically coupled with direct measurements of mercury concentrations in the sediments and water column of major tributaries to estimate mercury loading from the watershed.

The direct loading from the atmosphere to water surfaces may be estimated as wet deposition plus total dry deposition minus the foliar accumulation component. Because the CMAQ model runs estimate dry deposition to the land surface, the output includes the amount of mercury that has accumulated in leafy material (via stomatal uptake) and is eventually deposited to the land surface following leaf fall and decomposition. Direct dry deposition to a waterbody should not include this component. Foliar accumulation typically accounts for approximately 7 g/km^2/yr in the southwest region (Tetra Tech, 2008).

The CMAQ 2002 results are assumed to represent a more accurate estimate of dry deposition because this model included alterations to the functional deposition of processes associated with the dry deposition of mercury. Therefore, the dry deposition is primarily based on the CMAQ 2002 results, with the exception of Lake Sherwood. For Lake Sherwood, the 2001 results for dry deposition are assumed because 1) subtracting the foliar accumulation rate from the 2002 results would yield a negative deposition rate and 2) the net 2001 dry deposition rate (minus foliar accumulation) is similar to the 2002 gross dry deposition rate (see Table E-13 for a comparison of the 2001 and 2002 CMAQ results). It is important to note that there are only two published CMAQ model runs and only grid-scale model output was available; therefore, additional model runs could not be performed for TMDL development. The total dry deposition rates to each lake surface are summarized in Table E-14.

Table E-13. CMAQ Output for Grid Cells Underlying the Watersheds of the Mercury Impaired Lakes

Component	Mercury Deposition Rate (g/km^2/yr)		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
CMAQ 2001 Dry Deposition	77.5962	29.8365	12.1748
CMAQ 2002 Dry Deposition	57.0656	35.2323	5.6784

Note: Shaded cells represent the selected dry deposition rates for each waterbody.

Table E-14. Summary of Dry Deposition Estimates to Each Impaired Lake

Calculation Term	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Dry Deposition Rate ($g/km^2/yr$)*	57.0656	35.2323	12.1748
Dry Deposition Rate Minus Foliar Accumulation Rate ($g/km^2/yr$)	50.0656	28.2323	5.1748
Lake Surface Area (km^2)	0.143	1.021	0.554
Direct Dry Deposition Load (g/yr)	7.16	28.82	2.87

*Values are from shaded cells in Table E-13.

E.4.6. Total Mercury Deposition to Lake Surfaces

As discussed previously, mercury deposition to a lake surface may occur in either wet or dry form. Table E-15 summarizes the average wet, dry, and total deposition estimates for each mercury impaired lake.

Table E-15. Summary of Direct Mercury Deposition to Impaired Lakes

Deposition Load	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Lake Surface Area (km^2)	0.143	1.021	0.554
Calculated Wet (g/yr)[1]	0.45	4.86	2.40
Direct Dry Deposition Load (g/yr)[2]	7.16	28.82	2.87
Total (g/yr)	7.61	33.68	5.27

[1] See Table E-12.

[2] See Table E-14.

E.5 Organochlorine Pesticides and PCBs Deposition

An additional source of Organochlorine (OC) Pesticides and PCBs is atmospheric deposition, which occurs as a result of both local and global atmospheric transport. Unfortunately, atmospheric deposition is difficult to measure, and detailed information on atmospheric deposition rates of most OC Pesticides and PCBs is not available for southern California. (SCCWRP recently undertook a study of OC Pesticides and PCBs deposition, but has withdrawn the results based on methodological concerns.) It is well established, however, that atmospheric deposition of OC Pesticides and PCBs plays a significant role in contamination of lakes, even in remote areas, including national parks in the western US (Landers et al., 2010; Hageman et al., 2006).

The current atmospheric flux of OC Pesticides and PCBs to the lakes is thus unknown. Two factors help simplify the TMDL analysis. First, OC Pesticides and PCBs derived from atmospheric deposition on the watershed are implicitly included in estimates of watershed loading. Second, hydrophopic OC Pesticides and PCBs both deposit to and degas from waterbodies, and it is the net balance of these processes that is of most concern for the TMDL. The OC Pesticides and PCBs of concern are no longer in use, with atmospheric deposition rates declining, and elevated fish tissue concentrations appear to be largely due to legacy sediment contamination. In such situations, the net flux is typically outward from contaminated waterbodies to the atmosphere, thus rendering the net atmospheric flux to the lake less than or near zero. In the early 1990s, PCBs and dieldrin in the Great Lakes showed a net loss to the atmosphere, although DDT was still accruing (Hoff et al., 1996). In 1998-99, Park et al. (2002) reported that Corpus Christi Bay in Texas was a net source of PCBs to the atmosphere, and that the annual water-surface-exchange fluxes of most pesticides appeared to be close to a net of zero.

Given these considerations, direct *net* loading to the lake surface is assumed to be near zero. The associated load allocation for atmospheric deposition is also set to zero.

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E.6 References

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Appendix F. Estimation of Loading During Dry Weather

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F.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure F-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board) (Figure F-1). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

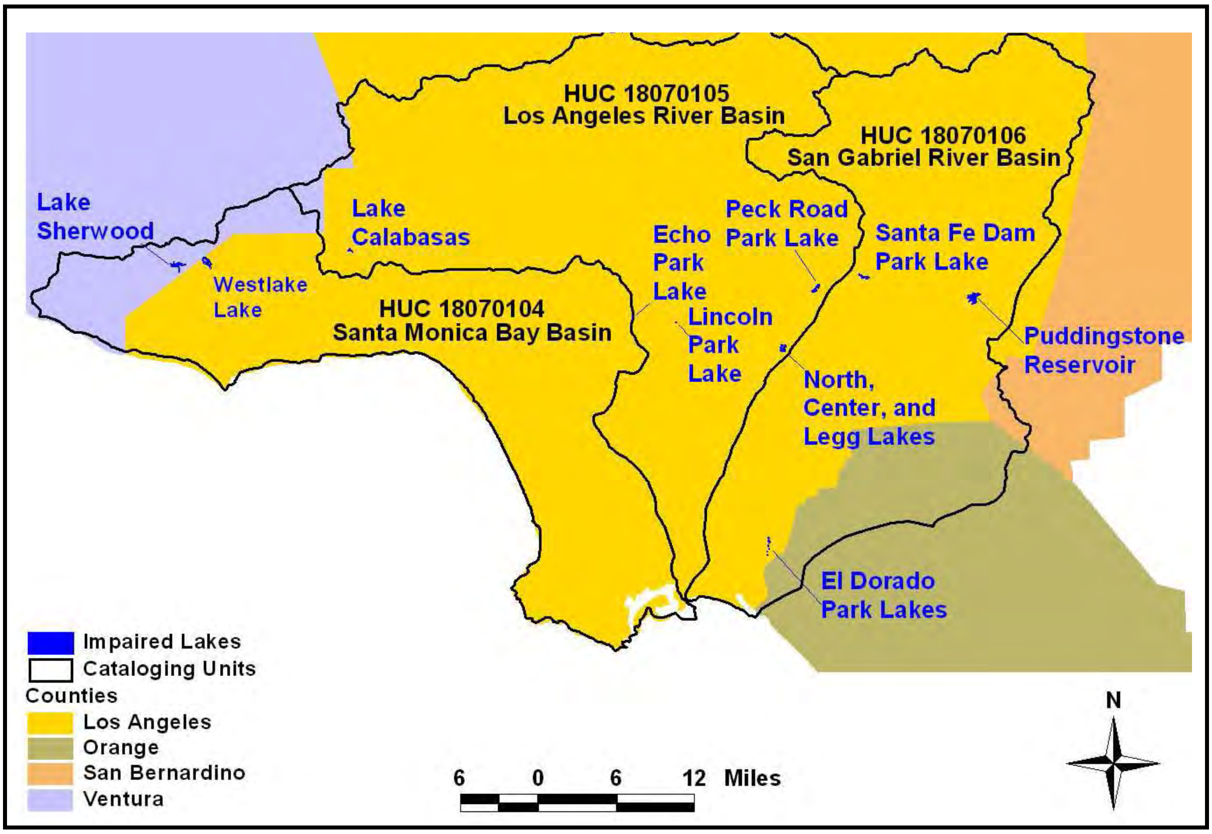

Figure F-1. Location of Impaired Lakes

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. These may include point source discharges, imported water, direct groundwater or potable water inputs, and flows resulting from irrigation. This appendix discusses these sources of pollutant loading and the methods used to estimate average annual dry weather loading to each impaired lake.

Dry weather loading was estimated for constituents with significant dry weather loads. Since organochlorine pesticides (chlordane, DDT, dieldrin) and PCBs are strongly sorbed to sediment, loading and transport during dry weather flow is assumed to be insignificant for these constituents and no separate load calculation is performed for dry weather flows.

The calculated dry weather loads represent a portion of the existing pollutant load to each impaired waterbody. Estimates of loading from other sources are described in other sections or appendices of the TMDL report. The summation of loads from all sources will then be used to estimate existing loading to each lake.

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F.2 Dry Weather Loads from Storm Drains

Two Loading Simulation Program in C++ (LSPC) watershed models were previously developed by Tetra Tech to estimate wet weather loading of metals to the Los Angeles and San Gabriel rivers (Tetra Tech, 2004; Tetra Tech, 2005). The models are large- scale models that estimate loading from water reclamation facilities and major storm drains discharging to either one of the main river bodies or to a major tributary. These models were developed to address wet weather metals impairments and were not used to estimate nutrient loading during dry weather.

The lakes addressed by this TMDL are each in small drainages relative to the dry weather models discussed above. In addition, all of the impaired lakes except Lake Sherwood include nutrient TMDLs. For these reasons, Tetra Tech estimated dry-weather loading from upland areas delivered via storm drains based on dry weather monitoring studies conducted by the Southern California Coastal Water Research Project (SCCWRP).

In 2002 and 2003, SCCWRP measured dry weather flows and concentrations of nutrients, metals, and bacteria in six watersheds in the Los Angeles River Basin (Stein and Ackerman, 2007). Concentration data collected during this study are applicable to a majority of impairments addressed by this TMDL. Two of the watersheds (Walnut Creek and Ballona Creek) do not receive inputs from wastewater treatment plant (WWTP) dischargers, and so the flows and loads measured reflect inputs from storm drains and their upland catchments only. Because the watersheds for these TMDLs do not contain WWTPs, data for the Walnut Creek and Ballona Creek watersheds can be used to represent storm drain inputs to the TMDL watersheds. The SCCWRP study only monitored those parameters for which the corresponding waterbody was listed as impaired. Thus, nutrient monitoring data and corresponding dry weather loading estimates are only available for the Walnut Creek watershed (Ballona Creek is not impaired for nutrients); therefore, only the Walnut Creek data are applicable to calculate nutrient loads to the TMDL watersheds. Table F-1 summarizes the mean concentrations of nutrients measured in the Walnut Creek watershed, which were used to estimate storm drain nutrient loads.

Table F-1. Mean Pollutant Concentrations Measured During Dry Weather Periods

Parameter	Walnut Creek Watershed
Total ammonia (mg-N/L)	0.1
Nitrate plus nitrite (mg-N/L)	1.0
TKN (mg-N/L)	2.0
Total phosphate (mg-P/L)	0.3

Total nitrogen concentration in dry weather runoff may be estimated from the species monitored and is approximately 3 mg-N/L. Total phosphorus concentration was estimated based on a total phosphate concentration of 0.3 mg-P/L and organic fractions observed under median flow conditions on the San Gabriel River (Tetra Tech, 2007). Assuming the median of observed flows is representative of dry weather conditions, the organic fraction observed (50 percent) is a reasonable approximation. Thus, total phosphorus in dry weather flows is approximately 0.6 mg-P/L.

Dry weather flows in urban areas tend to exhibit diurnal variability due to the nature of the primary sources of flow (irrigation, car washing, etc.). In 2005, SCCWRP presented results of a more intensive flow monitoring study where data were collected at five minute increments over a three month period. During periods identified as dry weather, the areal flow rate (flow rate divided by contributing area) was approximately 180 $m^3/d/km^2$, or 2.6 in/yr, in three watersheds (Ackerman and Stein, 2005).

The TMDLs are allocated based on subwatershed and jurisdiction. A GIS environment was used to overlay the subwatersheds, jurisdictions, and storm drain coverage to estimate the upland area that may contribute dry weather loading via storm drains. These areas were then multiplied by the annual average dry weather flow rates (2.6 inches/yr) and loading rates for total nitrogen and phosphorus (1.77 lb-N/ac/yr) and total phosphorus (0.354 lb-P/ac/yr):

$$\frac{3.0mg-N}{L}\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=1.77lb-N/ac/yr$$

$$\frac{0.6mg-P}{L}\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=0.354lb-P/ac/yr$$

For example, 100 acres of area draining to a storm drain network would contribute the following flows and nutrient loads:

$$100ac\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}=21ac-ft/yr$$

$$100ac\cdot 1.77lb-N/ac/yr=177lb-N/yr$$

$$100ac\cdot 0.354lb-P/ac/yr=35.4lb-P/yr$$

F.3 Contributions from Other Dry Weather Inputs

The lakes addressed in this TMDL report may receive inputs from several sources during dry periods. The majority of the impaired lakes receive potable water or groundwater as a supplemental source to offset evaporation and keep lake levels within a normal range.

Water used for irrigation around each lake also has the potential to deliver pollutants via runoff into the lake. Unit areas of urban land were set up for the LSPC modeling subbasins surrounding each impaired lake to estimate the percentage of irrigation water applied that would enter the lake via runoff or interflow.

During the 2009 and 2010 water quality monitoring events, known and accessible dry weather inputs were sampled for mercury, nutrients, and Organochlorine (OC) Pesticides and PCBs. (The groundwater, potable water, and reclaimed water sources at El Dorado Park lakes were also sampled for total mercury and methylmercury.) Measured concentrations were applied to known or estimated volumes to calculate loading to each lake. If water quality or flow estimates were not available for a potential source, assumptions were made to estimate loading.

The following sample calculation estimates average annual nitrogen load given a flowrate of 250 ac-ft/yr and a total nitrogen concentration of 1.2 mg/L:

$$\frac{250ac-ft}{yr}\cdot\frac{1.2mg}{L}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{28.32L}{ft^3}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=816lb-N/yr$$

Mercury loading is calculated in a similar manner, although the units on the concentration and load are different (the gram is used to summarize mercury loads because the pound is too large for the quantities delivered to the impaired waterbodies). To estimate mercury loading from an input that has an average flowrate of 250 ac-ft/yr with an average total mercury concentration of 10 ng/L, the following equation would be used,

$$\frac{250ac-ft}{yr}\cdot\frac{10ng-Hg}{L}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{28.32L}{ft^3}\cdot\frac{1g}{1{,}000{,}000{,}000ng}=3.08g-Hg/yr$$

Table F-2 summaries the dry weather sources that may contribute pollutant loading to each impaired lake. The sections that follow describe the sources and loading estimates specifically for each waterbody.

Table F-2. Dry Weather Loading Sources to the Impaired Lakes

Lake/ Reservoir	Storm Drains	Potable Water	Groundwater	Irrigation	NPDES
Peck Road Park Lake	Yes	No	No	Yes	No
Lincoln Park Lake	No[1]	Yes	No	Yes	No
Echo Park Lake	No[2]	Yes	No	Yes	No
Lake Calabasas	Yes	Yes	No	Yes	No
El Dorado Park Lakes	No	Yes	Yes	Yes	No

Lake/ Reservoir	Storm Drains	Potable Water	Groundwater	Irrigation	NPDES
North, Center, Legg Lakes	Yes	No	Yes	Yes	No
Puddingstone Reservoir	Yes	No	No	Yes	No
Santa Fe Dam Park Lake	No	Yes	Yes	Yes	No
Lake Sherwood	Yes	No	No	NA[3]	No

[1] The storm drain network passes under Lincoln Park Lake with no outfalls to the lake.

[2] Dry weather flows from the storm drain network are diverted downstream of Echo Park Lake.

[3] Information regarding irrigation was not collected because this TMDL does not address nutrient impairments for Lake Sherwood.

F.4 Peck Road Park Lake

Peck Road Park Lake is located in the Los Angeles River Basin. However, the Los Angeles County Department of Public Works (LACDPW) diverts flows from the San Gabriel River to Peck Road Park Lake via the Santa Fe Diversion Channel.

Impairments of this lake include low dissolved oxygen/organic enrichment, eutrophication (originally on the consent decree, but currently delisted), odor, lead, chlordane, dieldrin, DDT, PCBs, and trash. Dry weather contributions include storm drain inputs delivering dry weather flows from upland areas.

F.4.1 POLLUTANT LOADS FROM STORM DRAINS

Three subwatersheds comprise the drainage area to Peck Road Park Lake. The subwatershed draining the western part of the watershed via Santa Anita Wash is 12,686 acres, and the eastern subwatershed draining to Saw Pit Wash is 10,557 acres. There is an inwardly draining mining operation in the southern part of the eastern watershed that has been removed from the loading analysis. The subwatershed surrounding the lake is 321 acres. Each subwatershed drains to a storm sewer system so all allocations for the TMDLs are wasteload allocations, except for the trash TMDL which also has a load allocation.

Figure F-2 shows the MS4 stormwater permittees in the Peck Road Park Lake watershed. The western subwatershed is comprised of the county of Los Angeles, Sierra Madre, Arcadia, Monrovia, Angeles National Forest, and Caltrans areas. The eastern subwatershed is comprised of the county of Los Angeles, Monrovia, Duarte, Bradbury, Arcadia, Irwindale, Angeles National Forest, and Caltrans areas. The county of Los Angeles, Monrovia, Irwindale, Arcadia, and El Monte comprise the drainage around the lake. The park area is comprised of 152 acres adjacent to the lake (see the Peck Road Park Lake chapter for a more detailed map of the park area).

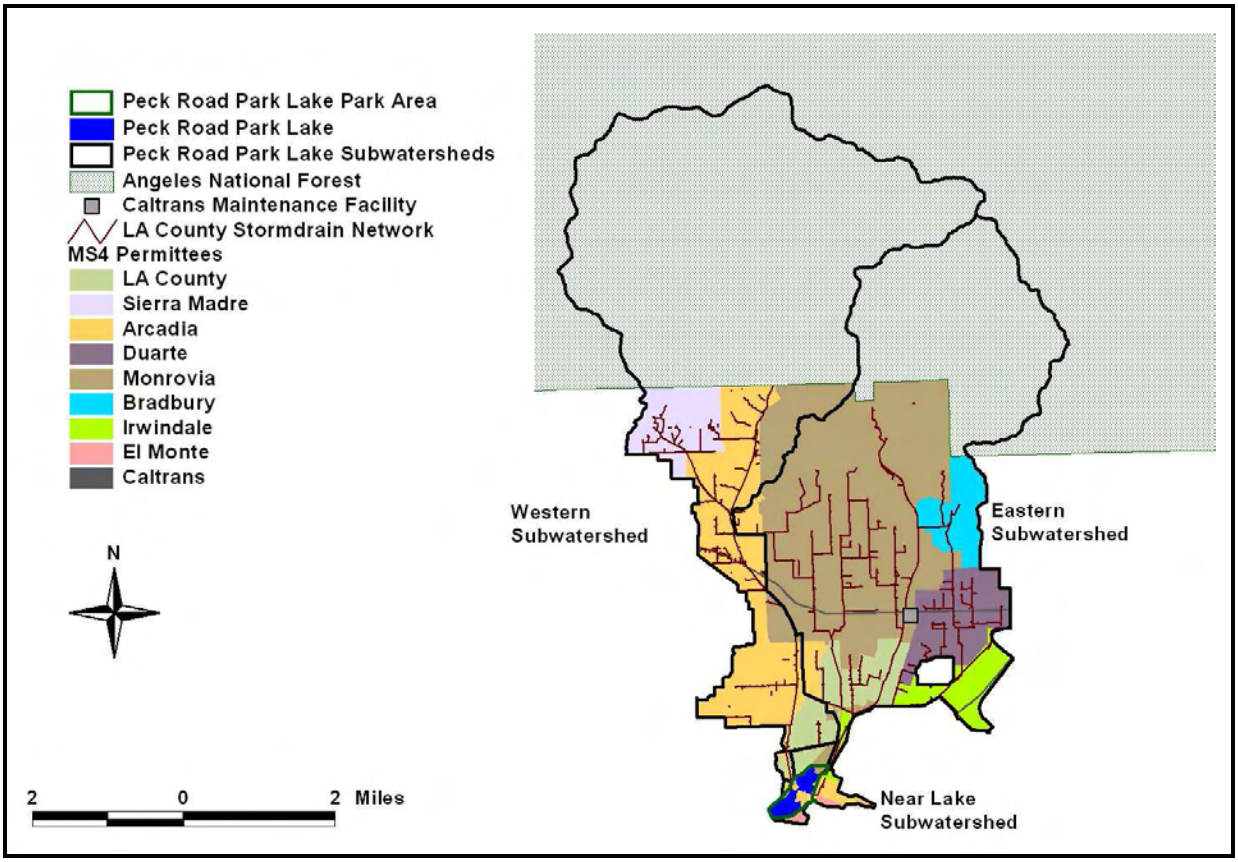

Figure F-2. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Peck Road Park Lake Subwatersheds

Table F-3 summarizes the upland areas draining to Peck Road Park Lake by subwatershed and jurisdiction. Dry weather loading from the Angeles National Forest is assumed to be zero (wet weather loading is described in Appendix D, Wet Weather Loading). Table F-4 through Table F-6 list the estimated dry-weather flows and nutrient loads corresponding to these areas. Sample calculations are provided in Section F.2.

Table F-3. Land Use Areas (ac) Draining to Peck Road Park Lake

Subwater-shed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Angeles National Forest	Total
Western	245	611	0	0	2,030	0	679	0	16.9	**9,104**	**12,686**
Eastern	499	4,456	818	503	209	483	0	0	78.4	**3,511**	**10,557**
Near Lake	67.7	48.1	0	0	139	14.1	0	52.1	0	**0**	**321**
Total	**812**	**5,115**	**818**	**503**	**2,378**	**497**	**679**	**52.1**	**95.3**	**12,615**	**23,564**

Table F-4. Estimated Dry Weather Flows to Peck Road Park Lake (ac-ft/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	52.9	132	0	0	439	0	147	0	3.65	**774**
Eastern	108	963	177	109	45.2	104	0	0	16.9	**1523**
Near Lake	14.6	10.4	0	0	30.0	3.05	0	11.3	0	**69.4**
Total	**175**	**1,105**	**177**	**109**	**515**	**107**	**147**	**11.3**	**20.6**	**2,366**

Table F-5. Estimated Dry Weather Nitrogen Loads to Peck Road Park Lake (lb/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	432	1,077	0	0	3,579	0	1,197	0	29.8	**6,316**
Eastern	880	7,856	1,442	887	369	852	0	0	138	**12,424**
Near Lake	119	84.8	0	0	245	24.9	0	91.9	0	**566**
Total	**1,431**	**9,019**	**1,442**	**887**	**4,193**	**876**	**1,197**	**91.9**	**168**	**19,305**

Table F-6. Estimated Dry Weather Phosphorus Loads to Peck Road Park Lake (lb/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	86.4	215	0	0	717	0	240	0	5.96	**1,263**
Eastern	176	1,571	288	177	73.7	170	0	0	27.6	**2,485**
Near Lake	23.9	17.0	0	0	49.0	4.97	0	18.4	0	**113**
Total	**286**	**1,803**	**288**	**177**	**840**	**175**	**240**	**18.4**	**33.6**	**3,861**

F.4.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

Water levels at Peck Road Park Lake are supplemented with flows from the San Gabriel River through a diversion channel. Estimates of flows and loads from this source are discussed in Appendix D (Wet Weather Loading) because this diversion is only used during wet weather.

A potable water source at Peck Road Park Lake is used to irrigate approximately 2 acres in a picnic area that is approximately 200 yards away from the lake. This area is fertilized when funding permits. Given the distance of this area from the lake, it is unlikely that irrigation or fertilization contributes significant nutrient loads to Peck Road Park Lake.

Other sources of nutrient loading may exist at Peck Road Park Lake such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, birds do feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems) were not accounted for in the Peck Road Park Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

F.5 Lincoln Park Lake

Lincoln Park Lake is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, lead, and trash. Dry weather contributions to this lake include lake filling and irrigation/fertilization of adjacent parkland.

Figure F-3 shows the MS4 stormwater permittee comprising the Lincoln Park Lake watershed (the city of Los Angeles). Though the lake appears to be connected to the county of Los Angeles storm drain network, this system actually passes under Lincoln Park Lake and does not discharge stormwater or dry weather flows to the lake.

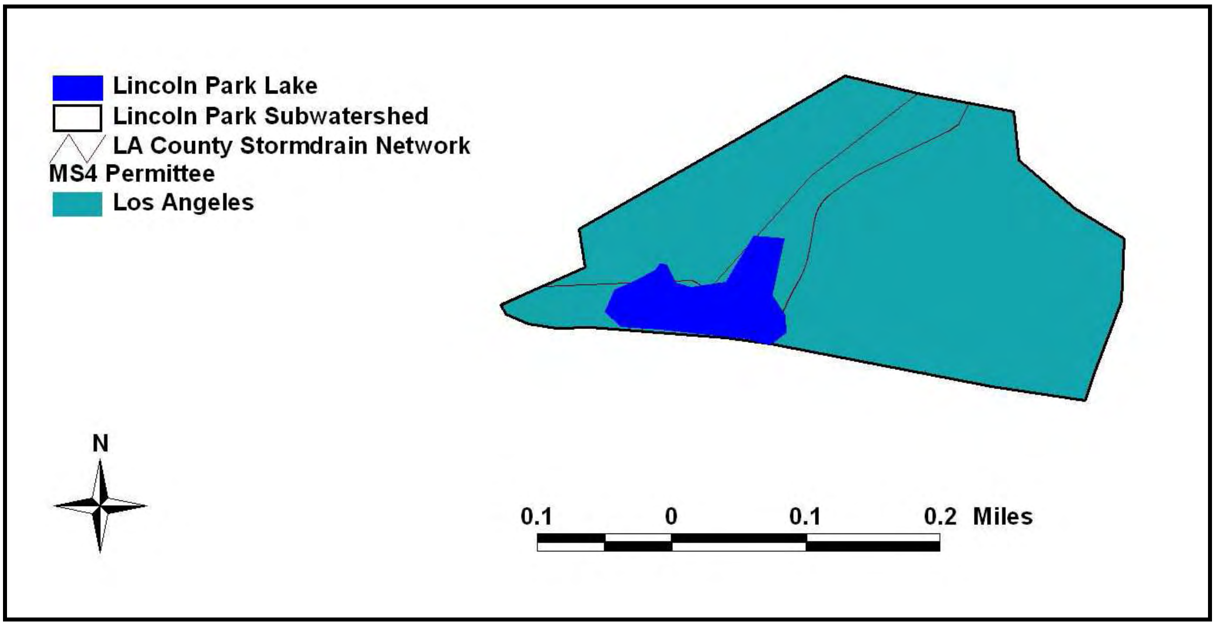

Figure F-3. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lincoln Park Lake Subwatersheds

F.5.1 Pollutant Loads from Storm Drains

Lincoln Park Lake is not hydraulically connected to the county of Los Angeles storm drain system although part of the system passes under the lake. Thus, dry weather loads to this lake delivered from storm drains are zero (wet weather loads from the watershed are discussed in Appendix D).

F.5.2 Pollutant Loads from Other Dry Weather Inputs

A potable water source at Lincoln Park Lake is used for lake filling as well as irrigation of parkland. Based on monthly usage summaries for May 2007 through April 2009, the average annual usage is 30.8 ac-ft/yr. All usage reported is applied directly to the lake to supplement lake levels. Park staff indicate that 32 acres surrounding the lake are irrigated with an additional 1 foot of potable water annually. Water is observed to percolate into the ground. The annual net evapotranspiration minus precipitation depth is 34.5 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model setup in LSPC for this subbasin indicates that approximately 5.6 percent of applied irrigation water reaches Lincoln Park Lake.

The potable water input at Lincoln Park Lake was sampled for water quality by USEPA and the Regional Board in August 2009. Table F-7 summarizes the observed water quality and estimated loads from lake filling and irrigation with the potable water source. See Section F.3 for sample calculations.

Table F-7. Summary of Potable Water Quality and Resulting Direct Loads to Lincoln Park Lake

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Addition (lb-N/yr or lb-P/yr)
Ammonia-N	0.335	1.64	28.1
Nitrate- N	0.33	1.61	27.6
Nitrite-N	0.03	0.147	2.51
TKN (mg-N/L)	0.531	2.60	44.5
Orthophosphate (mg-P/L)	0.017	0.083	1.42
Total Phosphorus (mg-P/L)	0.118	0.58	9.88
Total Nitrogen (calculated) (mg-N/L)	0.891	4.36	74.6

Note: Potable water concentrations are from data collected at Lincoln Park Lake.

The area surrounding Lincoln Park Lake (32 acres) is fertilized twice per year with 16-6-8 fertilizer at a rate of 7.5 lb/1,000 ft^2. The technical sheet for the product recommends applying this fertilizer at a rate of 6.25 lb/1,000 ft^2. It is difficult to estimate nutrient loading from fertilization as application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at Lincoln Park Lake, people were observed feeding the birds and a local person(s) was/were leaving piles of food along the shoreline of the lake. In addition, birds may feed from trash cans and food litter at the park. These practices increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in and around the lake. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, fertilization, etc.) were accounted for in a category called “Additional Parkland Loading.” During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations match those observed.

Precise bird counts for Lincoln Park Lake are not available; however, field notes indicate excess bird populations which are likely a significant portion of the nutrient loading associated with additional parkland areas. At Echo Park Lake, total phosphorus and total nitrogen loads of 78 lb-P/yr and 780 lb-N/yr were estimated for the approximately 1,000 birds observed to reside at that lake (Black and Veatch, 2010). The bird population at Lincoln Park like is likely one-half to one-quarter of that. Thus total phosphorus loads due to the bird population at Lincoln Park Lake likely range from 19.5 lb-P/yr to 39 lb-P/yr; total nitrogen loads range from 195 lb-N/yr to 390 lb-N/yr. The estimated loading from the resident bird population at Lincoln Park Lake is greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the bird population at Lincoln Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading.

F.6 Echo Park Lake

Echo Park Lake is located in the Los Angeles River Basin. Impairments of this lake include odor, ammonia, eutrophication, algae, pH, copper, lead, chlordane, dieldrin, PCBs, and trash. Dry weather contributions to this lake include lake filling and irrigation/fertilization of adjacent parkland.

Two subwatersheds comprise the drainage area to Echo Park Lake. The subwatershed draining the northern part of the watershed is 614 acres, and the southern subwatershed drains 170 acres. Both subwatersheds drain to a storm drain system, so all allocations for the TMDLs are wasteload allocations, except the trash TMDL which also has a load allocation. Dry weather flows from the storm drain system are diverted downstream of Echo Park Lake (Black and Veatch, 2008). Figure F-4 shows the MS4 stormwater permittee in the Echo Park Lake watershed. Both subwatersheds are located entirely within the city of Los Angeles with a small portion of Caltrans area. The park is comprised of 15.5 acres of land adjacent to the lake.

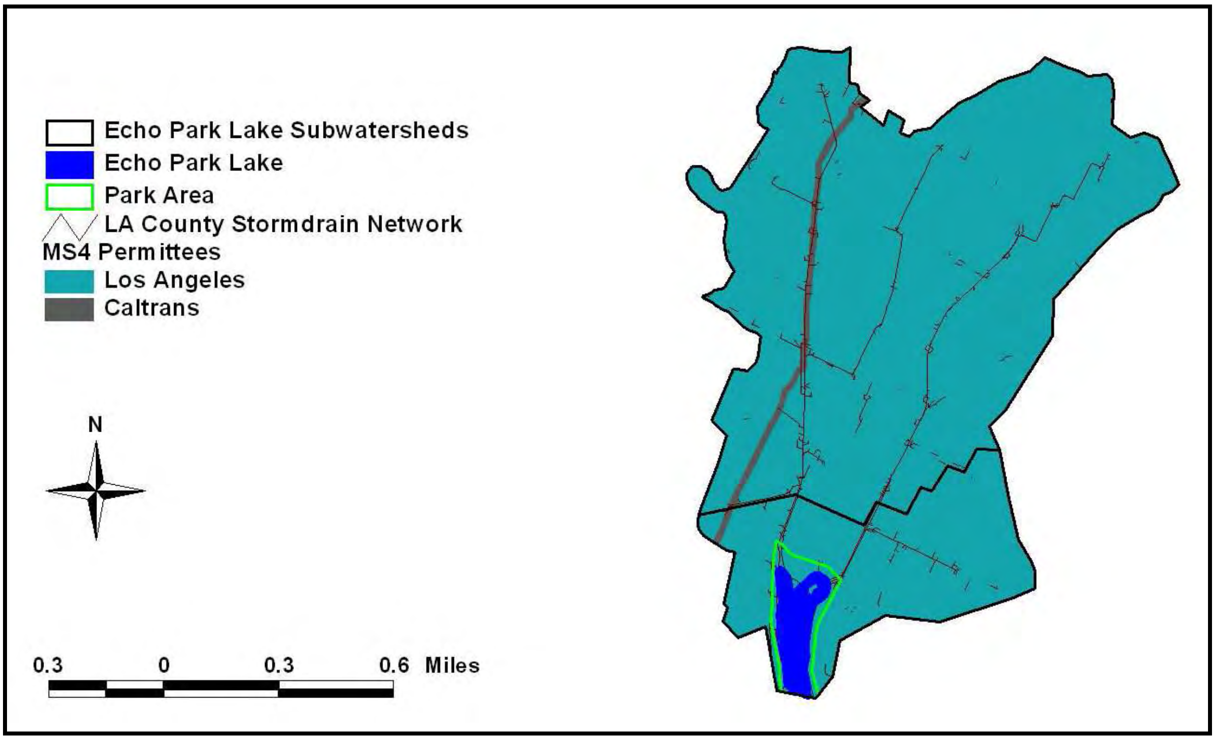

Figure F-4. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Echo Park Lake Subwatersheds

F.6.1 POLLUTANT LOADS FROM STORM DRAINS

A recent study performed for the city of Los Angeles Bureau of Sanitation found that dry season flows through storm drains generally bypassed Echo Park Lake (Black and Veatch, 2008). Thus, dry weather loads to this lake delivered from storm drains are zero (wet weather loads from the MS4 stormwater system are discussed in Appendix D).

F.6.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

A potable water source at Echo Park Lake is used for both lake filling and irrigation of surrounding parklands. According to a hydrologic study of the park lake conducted by Black & Veatch (2008), 162 ac-ft/yr of potable water are pumped annually. Staff at Echo Park indicate that approximately 9 acres in the vicinity of the lake are irrigated at a rate of approximately 1 foot per year and that the water mainly percolates into the ground with occasional runoff into the lake. The annual net evapotranspiration minus precipitation depth is 34.5 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model set up in LSPC for this subbasin indicates that approximately 4.6 percent of applied irrigation water reaches Echo Park Lake. The remainder of the pumped water (162 ac-ft minus 9 ac-ft) is assumed applied directly to the lake to maintain water levels.

The potable water source was sampled and analyzed for nutrients and metals on August 4, 2009. Table F-8 summarizes the nutrient water quality data as well as the resulting loads from irrigation and lake filling. Calculated nitrogen loads assume that parameters analyzed at less than detection have concentrations equivalent to ½ the detection limit. See sample calculations in Section F.3.

Table F-8. Summary of Potable Water Quality and Resulting Direct Loads to Echo Park Lake

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Addition (lb-N/yr or lb-P/yr)
Ammonia-N	<0.03	0.017	6.24
Nitrate- N	0.9	1.024	374.45
Nitrite-N	<0.01	0.006	2.08
TKN	<0.456	0.259	94.86
Orthophosphate	0.020	0.023	8.32
Total Phosphorus	0.122	0.139	50.76
Total Nitrogen (calculated)	1.133	1.289	471

Note: Potable water concentrations are from data collected at Echo Park Lake.

Nine acres surrounding Echo Park Lake are fertilized twice per year with 16-6-8 fertilizer at a rate of 7.5 lb/1,000 ft^2. The technical sheet for the product recommends applying this fertilizer at a rate of 6.25 lb/1,000 ft^2. It is difficult to estimate nutrient loading from fertilization as application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at Echo Park Lake, people were observed feeding the birds and a local person(s) was/were leaving piles of food along the shoreline of the lake. This practice increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in and around the lake. In addition, birds may feed from trash cans and food litter at the park. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, fertilization, etc.) were accounted for in a category called "Additional Parkland Loading." During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations matched those observed.

A significant portion of loading from the additional local sources is likely due to excessive bird populations. According to a recent water quality modeling study conducted by Black and Veatch (2010), there is a year-round, resident bird population of approximately 1,000 Rock Doves and American Coots. Estimates of nutrient loading from these birds were based on literature values and an assumption that all waste generated by the birds would reach the lake (i.e., no uptake or trapping in adjacent areas). The estimated total phosphorus loading from these birds is 78 lb-P/yr, and the estimated total nitrogen loading is 780 lb-N/yr. Both loading estimates are greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the year-round bird population at Echo Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading.

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F.7 Lake Calabasas

Lake Calabasas is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH. A DDT impairment was previously reported for this lake but was delisted by the Regional Board in 2009. Dry weather contributions to this lake include lake filling with a potable water source, storm drain inputs delivering dry weather flows from surrounding development, and irrigation and fertilization of areas around the lake.

F.7.1 Pollutant Loads from Storm Drains

One subwatershed draining 86.5 acres comprises the drainage area to Lake Calabasas. Figure F-5 shows the MS4 stormwater permittee in the Lake Calabasas watershed. The entire subwatershed is located in the city of Calabasas. This subwatershed drains to a storm drain system, so all allocations for the TMDLs are wasteload allocations.

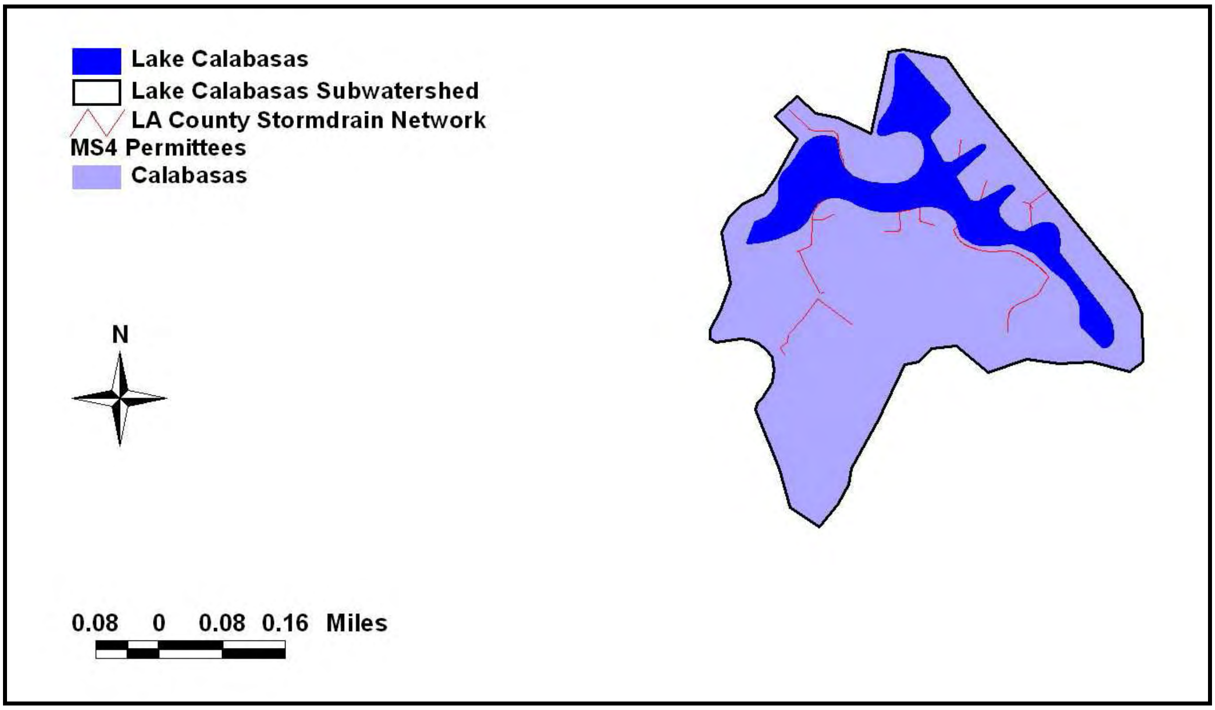

Figure F-5. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lake Calabasas Subwatersheds

Table F-9 summarizes the upland areas draining to Lake Calabasas as well as the associated dry weather flows and nutrient loads. Sample calculations are provided in Section F.2.

Table F-9. Land Use Areas and Associated Dry Weather Inputs to Lake Calabasas

Area (ac)	Flow (ac-ft/yr)	Total Nitrogen (lb/yr)	Total Phosphorus (lb/yr)
86.5	18.7	152	30.5

F.7.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

A potable water source at Lake Calabasas is used for both lake filling and irrigation of approximately 2 acres around the lake. Based on monthly data provided for 1995 to 2009, the average annual water usage is 60.8 ac-ft.

Only one day of irrigation usage has been monitored. On October 6, 2009, 0.011 ac-ft of potable water was applied over a 16-hour period. Assuming irrigation occurs five days a week throughout the year, the total applied irrigation volume is 2.86 ac-ft. The runoff depth based on this assumption is 17 inches. The annual net evapotranspiration minus precipitation depth is 37.6 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. Staff at Lake Calabasas indicate that some of the applied water runs off into the lake. A unit area model setup in LSPC for this subbasin indicates that approximately 5.3 percent of applied irrigation water reaches Lake Calabasas.

The potable water source at Lake Calabasas was sampled for water quality on August 6, 2009. Table F-10 summarizes the nutrient parameters sampled. The total phosphorus concentration was analyzed as less than the detection limit of 0.016 mg-P/L. Phosphate measured greater than the detection limit and is used to estimate total phosphorus loading from this source. The total nitrogen concentration is calculated assuming the nitrite concentration is equal to half the detection limit. Nutrient loading associated with irrigation and lake filling are also presented. Estimated total nitrogen and total phosphorus loads from irrigation are 0.655 lb-N/yr and 0.00852 lb-P/yr, respectively, assuming irrigation occurs five days a week throughout the year. Assuming the remainder of the usage is discharged directly to the lake, the additional nutrient loading to Lake Calabasas is 252 lb-N/yr and 3.28 lb-P/yr. See Section F.3 for sample calculations.

Table F-10. Water Quality Data for the Potable Water Source at Lake Calabasas

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Additions (lb-N/yr or lb-P/yr)
Ammonia-N	0.35	0.143	55.1
Nitrate- N	1.13	0.463	178
Nitrite-N	<0.01	0.0020	0.788
TKN	0.464	0.190	73.1
Orthophosphate	0.0208	0.00852	3.28
Total Phosphorus	<0.016	0.00852	3.28
Total Nitrogen (calculated)	1.60	0.655	252

Note: Potable water concentrations are from data collected at Lake Calabasas.

A portion of the common area surrounding Lake Calabasas is fertilized three times per year. The type of fertilizer applied varies depending on turf requirements. The average rate applied is approximately 1 lb per 250-275 sq ft of turf grass. The shrub and ground cover fertilizer is applied at approximately 5 lbs per 1,000 square feet. Staff at Lake Calabasas indicate that recommended rates are applied.

Residential properties surround Lake Calabasas, and maintenance staff indicate that some homeowners irrigate and fertilize their lawns. It is difficult to estimate nutrient loading from fertilization, from either the residential or common areas, because application rates and methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff. Loads from fertilization, as well as other sources that are difficult to quantify with the available information (wildlife, pets, etc.) were not accounted for in the Lake Calabasas nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

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F.8 El Dorado Park Lakes

The El Dorado Park lakes are located in the San Gabriel River Basin. Six lakes are located in the park. The northern four lakes are hydraulically connected and separate from the system comprised by the two southern lakes, also hydraulically connected. These lakes are listed as impaired by algae, ammonia, eutrophication, pH, copper, lead, and mercury. Dry weather contributions to these lakes include groundwater, potable water, and reclaimed water used for irrigation. No storm drains exist in the watershed that would deliver dry weather loads from areas outside of the park.

Two separate watersheds have been delineated for these separate lake systems. The subwatershed draining to the northern four lakes is comprised of 185 acres, and the subwatershed draining to the southern two lakes is comprised of 33.8 acres.

Figure F-6 shows the MS4 stormwater permittee that comprises both the northern and southern subwatersheds of the El Dorado Park lakes systems as well as the county of Los Angeles storm drain network. Although both watersheds are in the city of Long Beach incorporated area, there are no major drains that divert runoff directly to the lakes; a few small culverts pass water beneath walking paths and park roads. Because both watersheds are comprised solely of parklands that do not drain to a major storm drain system, the watershed loads to the El Dorado Park lakes are assigned load allocations in the TMDLs.

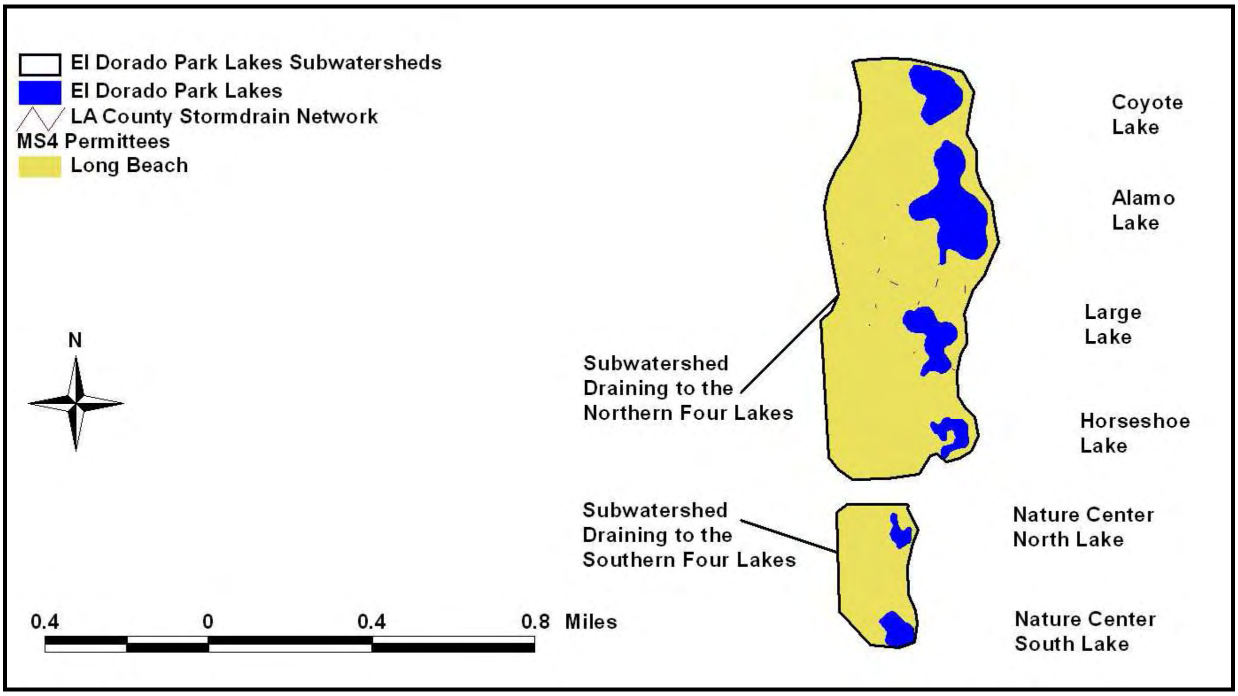

Figure F-6. MS4 Permittee and the County of Los Angeles Storm Drain Network in the El Dorado Park Lake Subwatersheds

F.8.1 Pollutant Loads from Storm Drains

The El Dorado Park lakes watersheds are isolated from upland areas; no storm drains deliver dry weather runoff from outside the park. Thus dry weather loads from storm drains are zero (wet weather loads from the watershed are discussed in Appendix D).

F.8.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

The El Dorado Park lakes are comprised of two hydraulically separate systems. The northern four lakes receive groundwater that is pumped into Coyote Lake at a rate of approximately 110 ac-ft/yr. During a typical year, 80 percent of flows are discharged during the summer season (May through September) (personal communication, Keith McDonald, Long Beach Water, August 25, 2009).

The groundwater input was sampled twice during 2009 for water quality. Table F-11 summarizes the observed water quality for this input and presents the average of the observed values used for load estimation for the northern four lakes (this source is not expected to exhibit seasonal variations of water quality).

Table F-11. Groundwater Quality Data for the El Dorado Park Lakes

Parameter	2/26/2009	7/15/2009	Average
Ammonia (mg-N/L)	0.325	0.28	0.302
Nitrate (mg-N/L)	<0.01	<0.01	<0.01
Nitrite (mg-N/L)	<0.01	<0.01	<0.01
TKN (mg-N/L)	0.805	1.1	0.952
Orthophosphate (mg-P/L)	0.072	0.071	0.0715
Total Phosphorus (mg-P/L)	0.189	0.291	0.240
Total Nitrogen (calculated) (mg-N/L)	0.815	1.11	0.962
Total Mercury (ng/L)	142	131	136.5
Methylmercury (ng/L)	0.215	0.109	0.162

The calculated total nitrogen values assume nitrate and nitrite concentrations are each equal to one-half the detection limits. Based on the average of concentrations observed and an annual average flow rate of 110 ac-ft/yr, the groundwater input delivers 287 lb-N and 71.5 lb-P per year. Total and methyl mercury loads are 18.4 g and 0.022 g, respectively. Example calculations are presented in Section F.3.

The southern lakes at El Dorado Park lakes receive supplemental flows from a potable water source. On average, 105 ac-ft are pumped annually into Nature Center North Lake. This source was sampled for water quality during the August 2009, August 2010, and September 2010 sampling events (Table F-12). Resulting average nutrient loads are 269 pounds of nitrogen and 13.7 pounds of phosphorus annually. Total and methyl mercury loads are 0.368 g and 0.00259 g, respectively. Example calculations are presented in Section F.3.

Table F-12. Potable Water Quality Data for El Dorado Park Lakes

Parameter	August 2009	August 2010	September 2010	Average
Ammonia (mg-N/L)	0.365	0.0359	0.292	0.231
Nitrate (mg-N/L)	0.37	0.173	0.173	0.239
Nitrite (mg-N/L)	<0.01	0.054	0.060	0.040
TKN (mg-N/L)	0.84	0.480	0.672	0.664
Orthophosphate (mg-P/L)	<0.0075	0.026	0.009	0.013
Total Phosphorus (mg-P/L)	0.1085	<0.0165	<0.0165	0.0478
Total Nitrogen (calculated) (mg-N/L)	1.21	0.707	0.905	0.942
Total Mercury (ng/L)	2.84	Not sampled	Not sampled	2.84
Methylmercury (ng/L)	0.020	Not sampled	Not sampled	0.020

The park area surrounding the El Dorado Park lakes is irrigated with reclaimed water. This source was sampled for water quality in December 2009. Table F-13 summarizes the water quality data relevant to the nutrient and mercury TMDLs.

Table F-13. Reclaimed Water Quality Data for El Dorado Park Lakes

Parameter	Concentration
Ammonia (mg-N/L)	0.62
Nitrate (mg-N/L)	4.45
Nitrite (mg-N/L)	0.05
TKN (mg-N/L)	1.22
Orthophosphate (mg-P/L)	0.084
Total Phosphorus (mg-P/L)	0.166
Total Nitrogen (calculated) (mg-N/L)	5.72
Total Mercury (ng/L)	1.46
Methylmercury (ng/L)	0.021

Irrigation water is applied to 221 acres surrounding Coyote and Alamo lakes (known as Area III) and 179 acres surrounding Large and Horseshoe lakes (known as Area II). At the Nature Center where the two southern lakes are located, 91.1 acres are irrigated. The applied average annual volumes to these respective areas (based on utility bills) are 244 ac-ft, 280 ac-ft, and 64.7 ac-ft; applied depths range from 8.5 inches to 18.8 inches. The annual net evapotranspiration minus precipitation depth is 34.9 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet

Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. Officials at the park state that most of the reclaimed irrigation water percolates into the ground, but some runs off into the lakes and some sprinkler heads spray across the stream. A unit area model setup in LSPC for this subbasin indicates that approximately 3.9 percent of applied irrigation water may reach the El Dorado Park lakes. No additional fertilization has occurred on the parkland. This condition is assumed to represent existing conditions. Table F-14 summarizes the pollutant loads delivered to the two separate lake systems based on this information. To estimate loading from irrigation, the results from applying the example calculation used in Section F.3 were multiplied by 0.039 (the fraction of applied flow assumed to reach the lakes).

Table F-14. Estimated Loads Resulting from Irrigation around the El Dorado Park Lakes

Pollutant	Loading to Northern Lake System	Loading to Southern Lake System
Ammonia (lb/yr)	34.7	4.29
Nitrate (lb/yr)	249	30.8
Nitrite (lb/yr)	2.80	0.35
Organic Nitrogen (lb/yr)	33.6	4.15
Total Nitrogen (lb/yr)	320	39.6
Phosphate (lb/yr)	4.70	0.58
Phosphorus (lb/yr)	9.29	1.15
Total Mercury (g/yr)	0.0371	0.00458
Methylmercury (g/yr)	0.000533	0.0000659

Note: Reclaimed water concentrations used in the loading calculations are from data collected at El Dorado Park lakes (Table F-13).

There are some additional sources of nutrient loading that may exist at El Dorado Park lakes, such as feces deposited in near lake areas by wildlife and pets. These loads are difficult to estimate with out information on wildlife populations, number of pets visiting annually, and percentage of pet owners properly disposing of pet wastes. Additionally, during sampling events at El Dorado Park lakes people were observed feeding the birds and the birds may also feed from trash cans and food litter at the park. This practice increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in or around the lake. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, etc.) were accounted for in a category called “Additional Parkland Loading.” During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations match those observed.

F.9 North, Center, and Legg Lakes

North, Center, and Legg lakes are hydraulically connected waterbodies in Whittier Narrows Regional Park located in the Los Angeles River Basin. Legg Lake is listed as impaired by odor, ammonia, pH, copper, and lead (note: trash impairment has been addressed by a previous TMDL). Dry weather contributions to these lakes include storm drains, groundwater, irrigation, fertilization, and treated groundwater from a Superfund site.

F.9.1 Pollutant Loads from Storm Drains

Five subwatersheds comprise the drainage area to these lakes. The northwestern and northeastern subwatersheds each drain to a storm drain that enters North Lake on the north side. Three separate subwatersheds areas have been delineated around the lakes to designate respective overland flow directly to each lake.

The northwestern, northeastern, and direct to north subwatersheds flow into North Lake which is basically separate from Center and Legg lakes during dry periods; North Lake discharges to Morris Creek. Legg Lake receives inputs from the direct to Legg subwatershed, from a Superfund site that discharges remediated water to the lake, and from pumped groundwater that is split between North and Legg lakes to maintain water levels. Legg Lake drains into Center Lake via a connecting channel which then discharges to Morris Creek. There are two culverts connecting Center and North lakes that allow water to flow between them when levels are sufficiently high.

Figure F-8 shows the MS4 stormwater permittees in the North, Center, and Legg lakes watershed. Loads generated from El Monte, South El Monte, Los Angeles County, and Caltrans from either the northwestern or northeastern subwatersheds are assigned wasteload allocations in the TMDLs because they drain to the storm drain network. Loads generated by South El Monte or the county of Los Angeles areas in the subwatersheds contributing directly to the lake are assigned load allocations; Caltrans areas in these subwatersheds are assigned wasteload allocations.

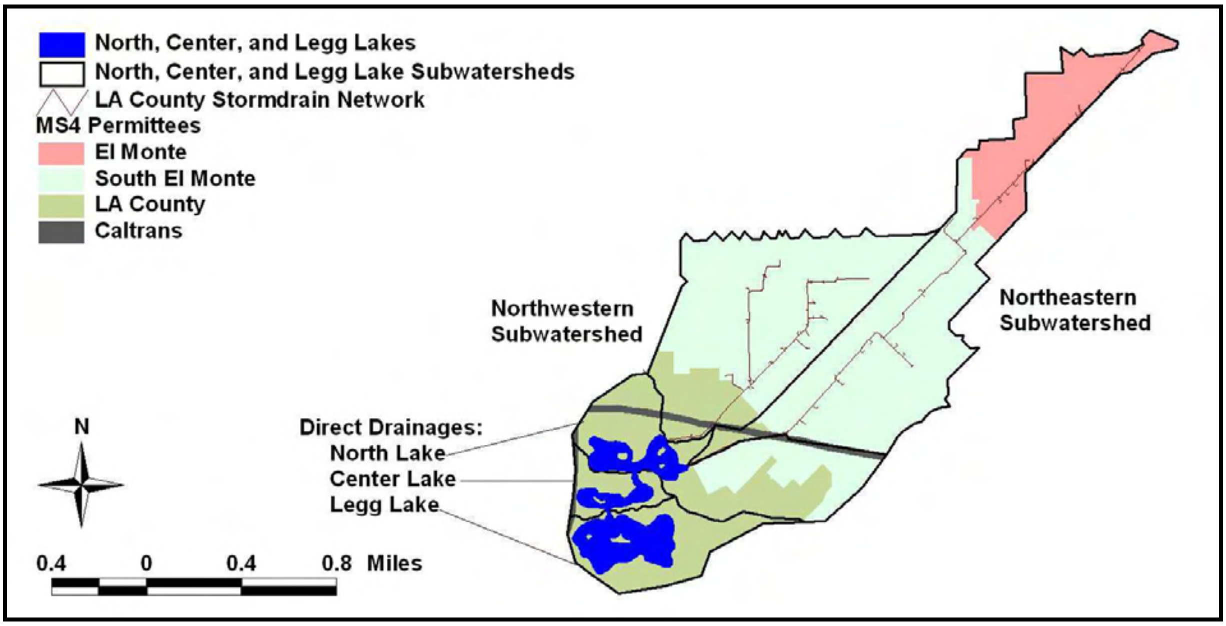

Figure F-7. MS4 Permittees and the County of Los Angeles Storm Drain Network in the North, Center, and Legg Lake Subwatersheds

Two subwatersheds contain a portion of the county of Los Angeles storm drain network that may deliver pollutant loads during dry weather. Table F-15 summarizes the upland areas draining to the Legg Lake system by subwatershed and jurisdiction. Table F-16 through Table F-18 list the estimated dry-weather flows and nutrient loads corresponding to these areas. Total nitrogen loading during dry weather is 1,478 lb/yr, and total phosphorus loading from dry weather flows is 296 lb/yr. Sample calculations are provided in Section F.2.

Table F-15. Land Use Areas (ac) Draining to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	317	60.1	5.32	**383**
Northeastern	134	305	10.0	6.18	**456**

Table F-16. Estimated Dry Weather Flows (ac-ft/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	68.6	13.0	1.15	**82.8**
Northeastern	29.0	65.9	2.17	1.34	**98.4**

Table F-17. Estimated Dry Weather Nitrogen Loads (lb/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	560	106	9.38	**675**
Northeastern	237	538	17.7	10.9	**803**

Table F-18. Estimated Dry Weather Phosphorus Loads (lb/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	112	21.2	1.88	**135**
Northeastern	47.3	108	3.53	2.18	**161**

F.9.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

To supplement water levels, treated groundwater from a Superfund site is pumped continuously into North and Legg lakes at an estimated rate of 2,534 ac-ft per year. This annual flow rate was estimated by extrapolating the flow measured by EPA for May through September 2010. Flows are split equally between these two lakes. Prior to May 2010 additional groundwater had been used to supplement water levels, but this input was discontinued.

In the summer of 2002, the city of Whittier began operating a Liquid Phase Granular Activated Carbon treatment facility associated with the San Gabriel Valley Area 1 Whittier Narrows Operable Unit Superfund site (EPA #CAD980677355) that treats groundwater contaminated with volatile organic chemicals (Stetson Engineers, 2009). In addition to monitoring levels of organic chemicals, the City is required to monitor concentrations of nitrate in the raw and treated water. Monitoring data collected over the period indicate that concentrations of nitrate did not change significantly in the treatment plant. The average nitrate concentration in the treated effluent, based on monthly samples collected in 2008, was 12 mg/L as NO_3 or 2.73 mg-N/L (Stetson Engineers, 2009).

EPA sampled the treated groundwater input during the June 8, August 11, and September 2010 sampling events. Table F-19 provides the mean observed concentrations for water quality parameters related to nutrient loading for these sampling events. Based on these concentrations, the treated Superfund discharge contributes 12,355 lb-N and 172 lb-P to the lake system, split equally between North and Legg lakes. Sample calculations are presented in Section F.3.

Table F-19. Mean Observed Concentrations for the Superfund Site at North, Center, and Legg Lakes for June, August, and September 2010 sampling events

Parameter	Mean Observed Concentration
Ammonia (mg-N/L)	0.05
Nitrate (mg-N/L)	1.60
Nitrite (mg-N/L)	0.13
TKN (mg-N/L)	0.07
Orthophosphate (mg-P/L)	0.03
Total Phosphorus (mg-P/L)	0.03
Total Nitrogen (calculated) (mg-N/L)	1.79

Runoff resulting from irrigation of 568 acres of parkland adjacent to the Legg Lake system is another potential source of nutrient loading. Water usage data for the Whittier Narrows Regional Recreation Area was provided for water years 2005 through 2009. Based on the average of the two most recent, complete water years, the total water usage at Whittier Narrows was 1,239 ac-ft. Staff at the park indicate that approximately 10 percent of this water is potable and 90 percent is reclaimed. Irrigation with the reclaimed water source began in 2006.

The usage also includes irrigation at Norman's Nursery, which is outside the watershed of the Legg Lake system. In 2006, Norman's Nursery used approximately 6.7 percent of the reclaimed water applied at Whittier Narrows. Subtracting out the usage at Norman's Nursery leaves approximately 1,040 ac-ft of reclaimed water applied around the Legg Lake system. An additional 124 ac-ft of potable water is also applied. On average, 24.6 inches of irrigation water are applied. The annual net evapotranspiration minus precipitation depth is 38.6 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model setup in LSPC for this subbasin indicates that approximately 6.3 percent of applied irrigation water reaches the lake system.

Water quality data for the reclaimed water source were provided for fiscal year 2006/2007. The total phosphorus concentration was not reported. To estimate the total phosphorus concentration, the ratio of total phosphorus to orthophosphate observed in reclaimed water at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13) was applied to the reported orthophosphate concentration. The resulting total phosphorus concentration is 1.45 mg-P/L.

The potable water source at Whittier Narrows was not sampled. Assumed concentrations for the potable water source are based on average values observed at the El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas potable water inputs. Table F-20 summarizes the reported concentrations for the reclaimed water source and the assumed concentrations for the potable water source. Only parameters relevant to the nutrient TMDLs are included.

Table F-20. Average Water Quality Data for the Irrigation Water Sources at Whittier Narrows

Parameter	Reclaimed Water	Potable Water
Ammonia (mg-N/L)	0.86	0.266
Nitrate (mg-N/L)	7.07	0.682
Nitrite (mg-N/L)	<0.03	0.011
Organic Nitrogen (mg-N/L)	1.43	0.249
Phosphate (mg-P/L)	0.733	0.015
Total Phosphorus (mg-P/L)	Not Reported	0.089
Total Nitrogen (calculated) (mg-N/L)	8.52	0.942

Note: Reclaimed water concentrations are from the reclaimed water source for fiscal year 2006/2007 (total phosphorous was not reported); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

Table F-21 summarizes the nutrient loads delivered to the Legg Lake system due to irrigation based on the volumes and water quality data described above. See Section F.3 for example calculations.

Table F-21. Estimated Annual Pollutant Loading Resulting from Irrigation at Whittier Narrows

Pollutant	Load
Ammonia (lb/yr)	158
Nitrate (lb/yr)	1,266
Nitrite (lb/yr)	2.89
Organic Nitrogen (lb/yr)	284
Total Nitrogen (lb/yr)	1,711
Phosphate (lb/yr)	130
Phosphorus (lb/yr)	258

Information regarding use of fertilizer has not been received from park staff. Even if the types and application rates were known, it would still be difficult to estimate nutrient loading from fertilization because application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at the Legg Lake system, people were observed feeding the birds and birds may feed from trash cans and food litter at the park. These practices increase nutrient loading to the lake by attracting animals that may deposit feces around the lake. Loads associated with food waste dumping, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area septic and other wastewater infrastructure systems, fertilization, etc.) were not accounted for in the North, Center, and Legg Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations in this system (see Appendix A, Nutrient TMDL Development).

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F.10 Puddingstone Reservoir

Puddingstone Reservoir is located in the San Gabriel River Basin. Impairments include low dissolved oxygen/organic enrichment, mercury, chlordane, DDT, and PCBs. Dry weather contributions to this lake include dry weather runoff to storm drains in the Northern Subwatershed and irrigation of parkland in the Southern Subwatershed.

F.10.1 Pollutant Loads from Storm Drains

Two subwatersheds comprise the drainage area to Puddingstone Reservoir. The subwatershed draining the northern part of the watershed is 6,959 acres, and the southern subwatershed is 1,169 acres. The subwatershed boundaries were chosen to separate those areas that drain to a storm drain (the northern subwatershed) and those that enter the reservoir via natural tributaries or overland flow (the southern subwatershed).

Figure F-8 shows the MS4 stormwater permittees in the Puddingstone Reservoir watershed. The northern subwatershed is primarily comprised of the county of Los Angeles, Claremont, and La Verne areas with a small amount of San Dimas, Caltrans, and Angeles National Forest areas. Loads generated from these jurisdictions in the northern subwatershed are assigned wasteload allocations because they drain to the county of Los Angeles storm drain network. The southern subwatershed is comprised of San Dimas, La Verne, and Pomona areas. Loads from these jurisdictions originating in the southern subwatershed are assigned load allocations. The small amount of Caltrans area in the Southern Subwatershed are assigned a wasteload allocation.

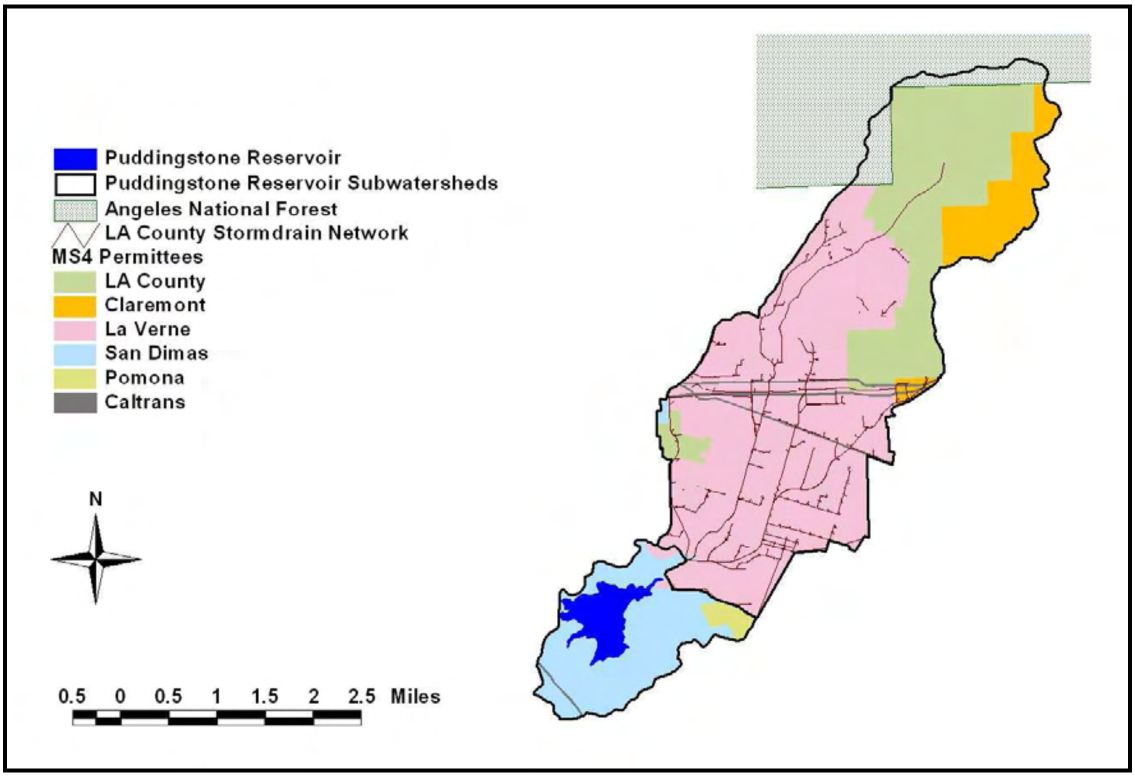

Figure F-8. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Puddingstone Reservoir Subwatersheds

The county of Los Angeles storm drain system is only present in the Northern subwatershed. The contributing areas by jurisdiction and the estimated dry-weather flows and pollutant loads are presented in Table F-22. Nutrient loads were calculated from the SCCWRP data presented in Section F.2. The relevant monitoring study did not include mercury data, so the mercury concentration observed in the summer of 2009 near the outlet of the Northern Subwatershed was used to represent dry weather concentrations of total mercury (4.24 ng/L) and methylmercury (0.553 ng/L). Dry weather loads from National Forest lands were assumed zero; wet weather loading from these areas is described in Appendix D (Wet Weather Loading). The Southern Subwatershed does not contain areas serviced by the storm drain network. Dry weather loads associated with the small pipes that drain surrounding parkland are likely due to irrigation of adjacent areas, which is discussed in the following section. Sample calculations for dry weather loads from the storm drain network are provided in Section F.2.

Table F-22. Estimated Dry Weather Storm Drain Inputs to Puddingstone Reservoir from the Northern Subwatershed

Watershed	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Area (ac)	578	1,865	4,079	5.28	28.5	110	**293**	**6,959**
Flow (ac-ft/yr)	125	403	881	1.14	6.15	23.8	**0**	**1,440**
Nitrogen (lb/yr)	1,019	3,288	7,191	9.32	50.2	194	**0**	**11,752**
Phosphorus (lb/yr)	204	658	1,438	1.86	10.0	38.8	**0**	**2,350**
Total Mercury (g/yr)	0.654	2.11	4.61	0.00597	0.0322	0.124	**0**	**7.53**
Methylmercury (g/yr)	0.085	0.275	0.601	0.000779	0.00419	0.0162	**0**	**0.983**

F.10.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

Puddingstone Reservoir does not receive supplemental flows from a potable or groundwater source. Though the Metropolitan Water District can divert water to Puddingstone Reservoir from outside the watershed, this practice is seldom used (personal communication, Adam Walden, Los Angeles County Department of Public Works, 9/16/09) and does not impact the average conditions for this reservoir.

The park area around Puddingstone Reservoir is irrigated with a combination of reclaimed and potable water. Water quality data for the reclaimed water source were provided by the city of Pomona. The potable water source at Puddingstone Reservoir was not sampled during the 2009 monitoring event and data were not available from the source. To estimate water quality for this source, concentrations were assumed equal to the average concentrations observed at the El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas potable water inputs. Table F-23 summarizes the monitoring data for the reclaimed water provided by the city of Pomona and the average concentrations assumed for the potable water source. Only those parameters relevant to the nutrient and mercury TMDLs are included in the table.

Table F-23. Average Water Quality Data for the Irrigation Water Sources at Puddingstone Reservoir

Parameter	Reclaimed Water	Potable Water
Ammonia (mg-N/L)	1.33	0.266
Nitrate (mg-N/L)	5.49	0.682
Nitrite (mg-N/L)	0.094	0.011
Organic Nitrogen (mg-N/L)	1.27	0.249
Phosphate (mg-P/L)	Not reported	0.015
Total Phosphorus (mg-P/L)	Not reported	0.089
Total Nitrogen (calculated) (mg-N/L)	8.18	1.21
Total Mercury (ng/L)	24	2.84

Note: Reclaimed water concentrations were provided by the city of Pomona (phosphate and total phosphorous were not reported); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

The monitoring data for the reclaimed water at Puddingstone Reservoir does not include phosphorus parameters. A phosphate concentration of 0.408 mg-P/L was assumed, based on averaging the phosphate concentrations reported for Legg Lake and El Dorado Park lakes. The total phosphorus concentration was estimated by applying the ratio of total phosphorus to orthophosphate observed at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13). The resulting total phosphorus concentration is 0.807 mg-P/L.

Park staff report that approximately 1,180 acres in the park are irrigated. Utility bills indicate that on average, 1,510 ac-ft of reclaimed water and 104 ac-ft of potable water are used for irrigation each year. This volume equates to a depth of 16.4 inches, which is significantly less than the net evaporation minus precipitation depth (37.7 inches) estimated from data posted on the CIMIS website for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading).

Officials at the park state that the majority of the irrigation water percolates into the ground, although areas along the shoreline do produce runoff to the reservoir during irrigation. A unit area model set up in LSPC for this subbasin indicates that approximately 10.1 percent of applied irrigation water reaches Puddingstone Reservoir. During the past three years, no additional fertilization has occurred due to budget considerations. This condition is assumed to represent existing conditions. The resulting loads due to irrigation are summarized in Table F-24. Example calculations are presented in Section F.3.

Table F-24. Estimated Annual Pollutant Loading Resulting from Irrigation at Puddingstone Reservoir

Pollutant	Load
Ammonia (lb/yr)	559
Nitrate (lb/yr)	2,294
Nitrite (lb/yr)	39.3
Organic Nitrogen (lb/yr)	533
Total Nitrogen (lb/yr)	3,425

Pollutant	Load
Phosphate (lb/yr)[1]	170
Phosphorus (lb/yr)[2]	337
Total Mercury (g/yr)	4.55

[1] The monitoring data for the reclaimed water at Puddingstone Reservoir does not include phosphorus parameters. A phosphate concentration of 0.408 mg-P/L was assumed, based on averaging the phosphate concentrations reported for Legg Lake and El Dorado Park lakes.

[2] The total phosphorus concentration was estimated by applying the ratio of total phosphorus to orthophosphate observed at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13). The resulting total phosphorus concentration is 0.807 mg-P/L.

Other sources of nutrient loading may exist at Puddingstone Reservoir such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, birds may feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems), were not accounted for in the Puddingstone Reservoir nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

F.11 Santa Fe Dam Park Lake

Santa Fe Dam Park Lake is located in the San Gabriel River Basin. Impairments of this lake include pH, copper, and lead. The waterbody is a recreational lake that was constructed within the Santa Fe Flood Control Basin, but no water from the basin is diverted into the lake (personal communication, Chris Graham, Los Angeles County Department of Parks and Recreation). Dry weather inputs include groundwater and potable water used for maintaining lake levels and runoff from irrigation.

One 362-acre subwatershed comprises the drainage area to Santa Fe Dam Park Lake. No storm drain system is present in the watershed. Figure F-9 shows the jurisdictions present in the Santa Fe Dam Park Lake watershed. Most of the drainage area is located in Irwindale, with a small portion in Azusa.

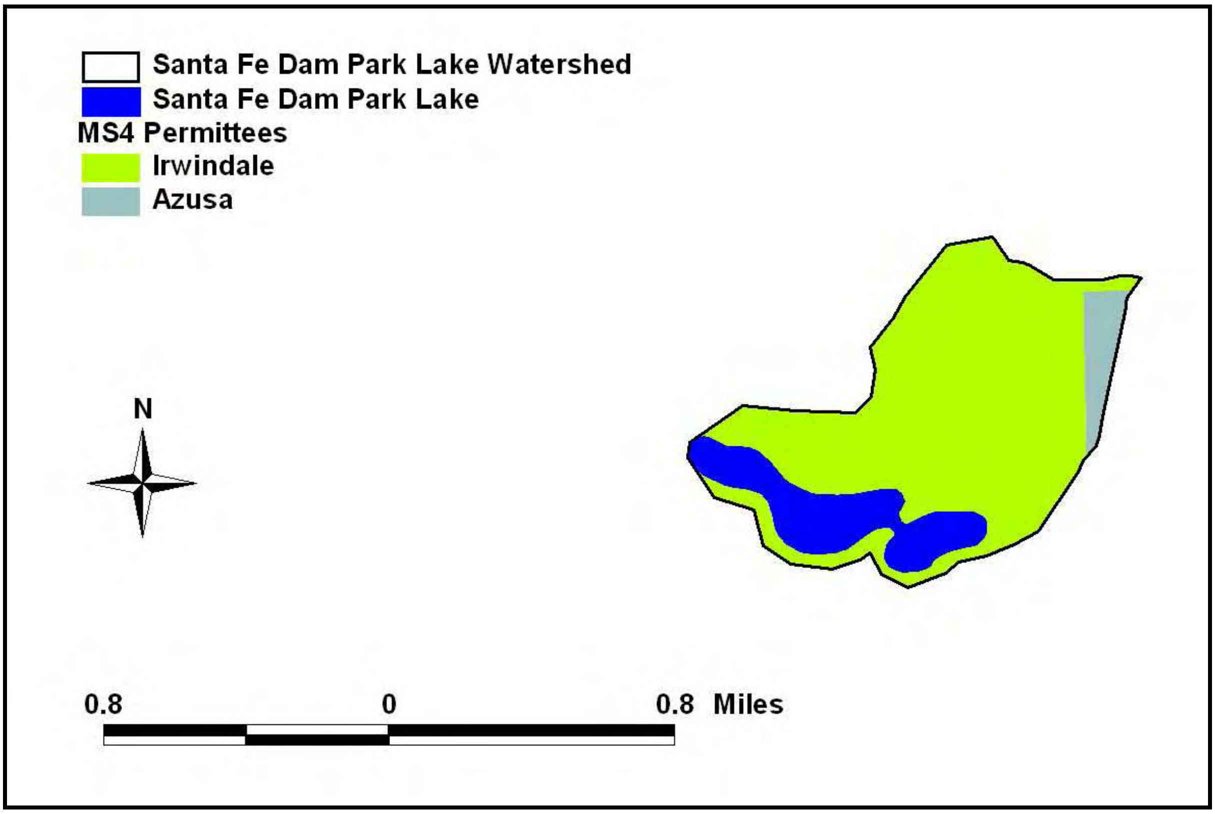

Figure F-9. Jurisdictions in the Santa Fe Dam Park Lake Subwatershed

F.11.1 Pollutant Loads from Storm Drains

There are no storm drains in the Santa Fe Dam Park Lake watershed. Thus dry weather loads from storm drains are zero for this waterbody (wet weather loads from the watershed are discussed in Appendix D).

F.11.2 Pollutant Loads from Other Dry Weather Inputs

Santa Fe Dam Park Lake receives supplemental flows from groundwater and potable water sources to maintain lake levels. Ten years of monthly usage data were used to estimate the average annual volume pumped from each source. Groundwater and potable water are pumped at average rates of 1,319 ac-ft/yr and 544 ac-ft/yr, respectively.

The groundwater input at Santa Fe Dam Park Lake was sampled on August 3, 2009 and August 12, 2010. The calculated total nitrogen value for the groundwater concentrations reported as less than the detection limit are equal to one-half the detection limits. During both sampling events, total phosphorus was analyzed as less than the detection limit of 0.016 mg/L; therefore, the phosphate concentration was used to represent the total phosphorus content of the groundwater. The potable water input at the discharge point to Santa Fe Dam Park Lake has not been sampled. The average of measurements obtained from potable water sources at other impaired lakes sampled for this TMDL study were used to estimate the nutrient concentrations for this source (El Dorado Park lakes, Echo Park Lake, Lake Calabasas, and Lincoln Park Lake). Table F-25 summarizes the average observed and estimated concentrations for the groundwater and potable water inputs at Santa Fe Dam Park Lake.

Table F-25. Water Quality Data for the Groundwater and Potable Water Inputs at Santa Fe Dam Park Lake

Parameter	Groundwater	Potable Water
Ammonia (mg-N/L)	0.03	0.266
Nitrate (mg-N/L)	2.3	0.682
Nitrite (mg-N/L)	0.02	0.011
TKN (mg-N/L)	0.67	0.516
Orthophosphate (mg-P/L)	0.026	0.015
Total Phosphorus (mg-P/L)	0.026	0.0923
Total Nitrogen (calculated) (mg-N/L)	2.99	1.21

Note: Groundwater concentrations are from data collected at Santa Fe Dam Park lake (total phosphorous was less than the detection limit, so the phosphate concentration was used to represent total phosphorous); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

Nutrient loads discharged directly to Santa Fe Dam Park Lake from these sources can be calculated from the average annual volume discharged and the water quality concentrations. Total nitrogen loads from groundwater and potable water are estimated to be 10,734 lb-N/yr and 1,790 lb-N/yr, respectively. Total phosphorus loads from these sources are 93.3 lb-P/yr and 137 lb-P/yr, respectively. Example calculations are presented in Section F.3.

In addition to inputs of potable water and groundwater, the swim beach area of Santa Fe Dam Park Lake is chlorinated during the summer months. Chlorination typically occurs seven days per week via five pumps. However, due to reduced funding available in 2009, the swim beach was closed Monday through Wednesday and only one chlorine pump was being utilized (personal communication, Chris Graham, Los Angeles County Department of Parks and Recreation, September 19, 2009). Chlorination alters the pH of the water and may be contributing to the pH impairment.

The groundwater source at Santa Fe Dam Park Lake is also used to irrigate 175 acres of parkland. Irrigation water is observed to percolate into the ground. Application volumes were not available. To estimate loading from this source, it is assumed that a depth of 1 foot of water is applied annually. A unit area model setup in LSPC for this subbasin indicates that approximately 9.6 percent of applied irrigation water reaches the lake. These assumptions yield nutrient loads to the lake of 137 lb-N/yr and

1.19 lb-P/yr. There is no fertilization schedule for this area, so loads from fertilizer are assumed zero. Example calculations are presented in Section F.3.

Other sources of nutrient loading may exist at Santa Fe Dam Park Lake such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, it is likely a recreational activity at the lake and birds may feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems) were not accounted for in the Santa Fe Dam Park Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

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F.12 Lake Sherwood

Lake Sherwood is located in the Santa Monica Bay Basin and is impaired by mercury (note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL). Dry weather contributions to this lake include dry weather runoff to storm drains in the developed subwatersheds.

F.12.1 Pollutant Loads from Storm Drains

Six subwatersheds comprise the drainage area (10,656 acres) to Lake Sherwood. Figure F-10 shows the MS4 stormwater permittees comprising each subwatershed.

Ventura County is the only stormwater permittee in the Western Subwatershed. The Hidden Valley Wash subwatershed is mostly in Ventura County with small portion in Thousand Oaks. The Northern, Near Lake Undeveloped, and Near Lake Developed subwatersheds are comprised of both Ventura County and Thousand Oaks MS4 areas. The Carlisle Canyon subwatershed contains Ventura and Los Angeles County areas as well as Thousand Oaks, California Department of Transportation (Caltrans), and California State Park areas. Neither Ventura or Los Angeles counties (the MS4 stormwater permittees in the watershed) maintain storm drain systems in the Lake Sherwood watershed. However, there are residential developments in the vicinity of the lake which drain to culverts and storm drains. These areas are generally associated with the Sherwood Valley Homeowner’s Association (SVHOA) and Sherwood Development Company. All subwatersheds will receive wasteload allocations except for the Carlisle Canyon and Near Lake Undeveloped subwatersheds. The small Caltrans area in the Carlisle Canyon subwatershed will also receive a wasteload allocation.

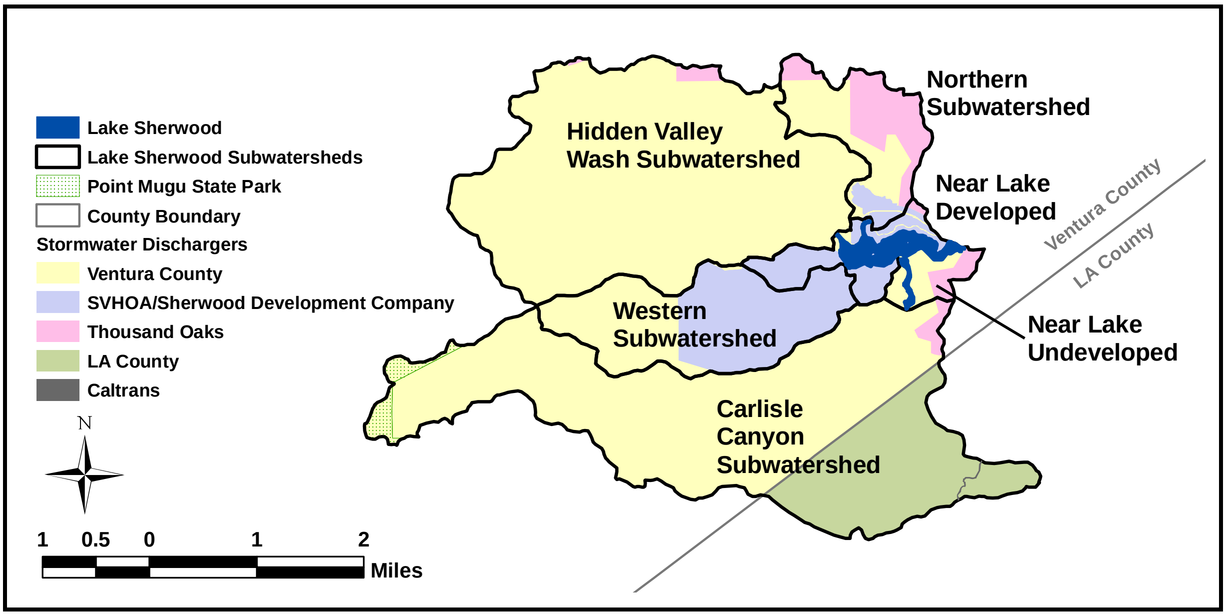

Figure F-10. MS4 Permittees in the Lake Sherwood Subwatersheds

The developed subwatersheds (Northern, Western, and Near Lake Developed) likely contribute dry weather flows and loading to Lake Sherwood via the storm drain system. [No flows were observed along Hidden Valley Wash during dry weather sampling, so dry weather loads from this subwatershed are assumed zero.] Dry weather flow volumes (described in Section F.2) were estimated for the “other urban" and residential land uses in each developed subwatershed.. All developed lands in these subwatersheds are in Ventura County. Mercury concentrations observed during the summer monitoring event (Appendix G, Monitoring Data) were assumed to represent the dry weather concentrations at the mouth of each tributary or storm drain. Table F-26 summarizes the resulting total mercury loads, and Table F-27 summarizes the methylmercury loads. Example calculations are presented in Section F.3.

Table F-26. Dry Weather Total Mercury Loading to Lake Sherwood

Subwatershed	Developed Area (ac)	Dry Weather Flows (ac-ft/yr)	Dry Weather Total Mercury Concentration (ng/L)	Dry Weather Total Mercury Load (g/yr)
Northern	43.0	9.28	54.0	0.618
Western	185	39.9	4.58	0.226
Near Lake Developed	175	37.7	54.0[1]	2.51

[1] Concentrations for this subwatershed are assumed similar to those observed in the Northern Subwatershed based on land use similarity.

Table F-27. Dry Weather Methylmercury Loading to Lake Sherwood

Subwatershed	Developed Area (ac)	Dry Weather Flows (ac-ft/yr)	Dry Weather Methylmercury Concentration (ng/L)	Dry Weather Methylmercury Load (g/yr)
Northern	43.0	9.28	0.096	0.0011
Western	185	39.9	0.536	0.0264
Near Lake Developed	175	37.7	0.096[1]	0.00447

[1] Concentrations for this subwatershed are assumed similar to those observed in the Northern Subwatershed based on land use similarity.

F.12.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

There are no additional dry weather sources of mercury loading to Lake Sherwood.

F.13 References

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Appendix G. Monitoring Data for the Los Angeles Area Lakes TMDLs

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G.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure G-1). Los Angeles Water Quality Control Board (Regional Board) assisted USEPA in this effort by compiling historic data associated with 1998 list and with collecting recent (2008-2010) monitoring results. The waterbodies are impaired by low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash. This appendix describes the monitoring data relevant to TMDL development, determinations of nonimpairment, and determinations of new impairments for these waterbodies.

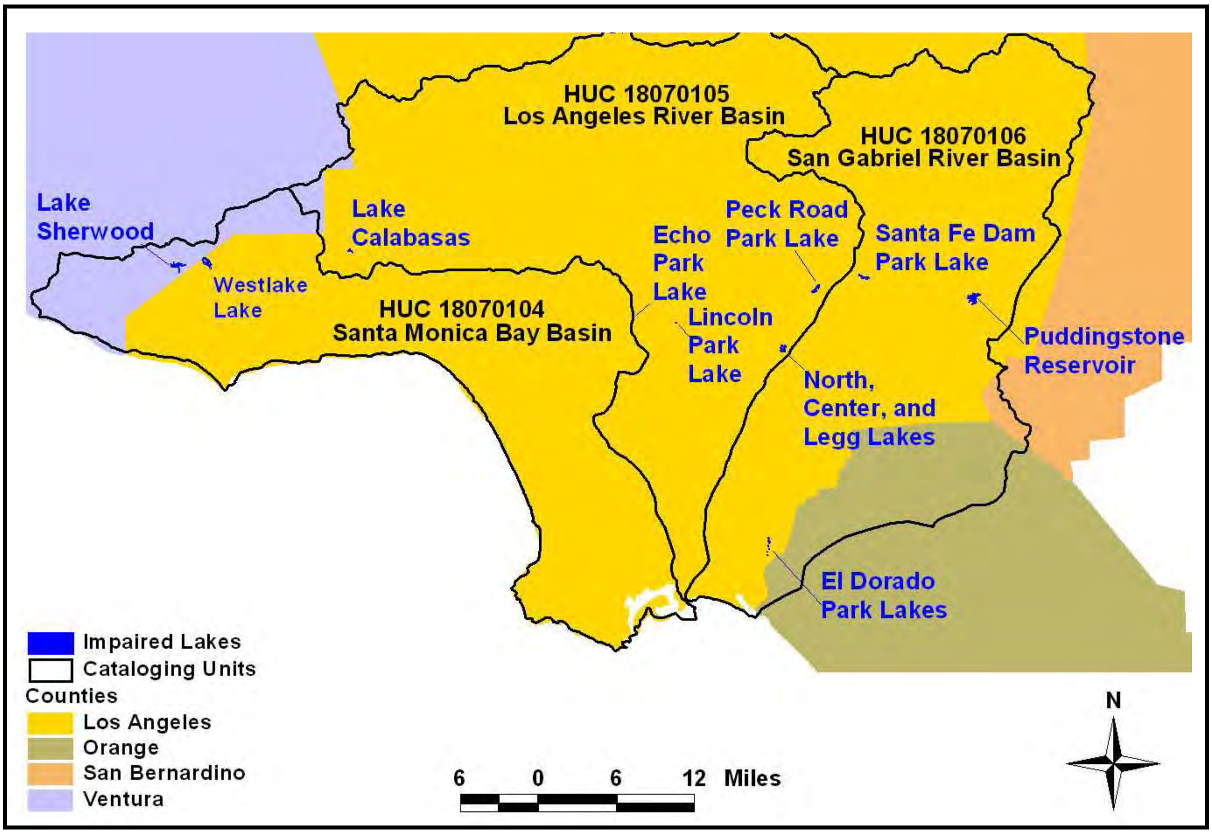

Figure G-1. Location of Impaired Lakes

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G.2 Overview of Monitoring Parameters

The impairments in the Los Angeles area lakes presented in Table 2-30 can be grouped into five categories: nutrients, mercury, metals, Organochlorine (OC) Pesticides and PCBs, and trash. Various monitoring parameters are helpful to evaluate impairments and characterize watershed and in-lake conditions for TMDL analyses. The parameters and their associated category are presented in Table G-1. This table also identifies the media and whether available data were reported by an analytical laboratory and/or measured in the field. Specifics for nutrients, mercury, metals, and OC Pesticides and PCBs data are described for each lake in Section G.4 through Section G.13. Trash monitoring measurements are described in the lake-specific chapters.

Table G-1. Monitoring Parameters by Category

Analysis Category	Monitoring Parameter	Media	Analytical Laboratory Result	Field Measurement
Nutrients	Ammonia/Ammonium	Water column	●	
	Biochemical Oxygen Demand	Water column	●	
	Chloride	Water column	●	
	Chlorophyll *a*	Water column	●	
	Depth	Water column		●
	Dissolved Organic Carbon	Water column	●	
	Dissolved Oxygen	Water column	●	●
	Electrical Conductivity	Water column		●
	Nitrate	Water column	●	
	Nitrite	Water column	●	
	Organic Nitrogen	Water column	●	
	Orthophosphate	Water column	●	
	pH	Water column	●	●
	Secchi Depth	Water column		●
	Sulfate	Water column	●	
	Suspended Solids	Water column	●	
	Temperature	Water column		●
	Total Alkalinity	Water column	●	
	Total Dissolved Solids	Water column	●	
	Total Kjeldahl Nitrogen	Water column	●	
	Total Organic Carbon	Water column	●	
	Total Phosphate	Water column	●	
	Total Phosphorous	Water column	●	
Mercury	Methylmercury (total)	Water column	●	

Analysis Category	Monitoring Parameter	Media	Analytical Laboratory Result	Field Measurement
	Methylmercury (total)	Sediment	●	
	Sulfate	Sediment	●	
	Sulfate	Water column	●	
	Total Mercury	Water column	●	
	Total Mercury	Sediment	●	
	Total Mercury	Fish tissue	●	
	Total Suspended Solids	Water column	●	
Metals	Cadmium (dissolved)	Water column	●	
	Copper (dissolved)	Water column	●	
	Copper (total)	Sediment	●	
	Lead (dissolved)	Water column	●	
	Lead (total)	Water column	●	
	Lead (total)	Sediment	●	
	Total hardness	Water column	●	
	Zinc (dissolved)	Water column	●	
OC Pesticides and PCBs	Chlordane*	Water column	●	
	Chlordane*	Porewater	●	
	Chlordane*	Fish tissue	●	
	Chlordane*	Sediment (bed and suspended sediment)	●	
	DDTs*	Water column	●	
	DDTs*	Porewater	●	
	DDTs*	Fish tissue	●	
	DDTs*	Sediment (bed and suspended sediment)	●	
	Dieldrin	Water column	●	
	Dieldrin	Porewater	●	
	Dieldrin	Fish tissue	●	
	Dieldrin	Sediment (bed and suspended sediment)	●	
	PCBs*	Water column	●	
	PCBs*	Porewater	●	
	PCBs*	Fish tissue	●	
	PCBs*	Sediment (bed and suspended sediment)	●	

* May include various chemicals that make up the total compound.

G.3 Overview of Monitoring Studies

Several studies have been conducted over the past few decades to monitor water quality in the county of Los Angeles. The University of California at Riverside conducted a study of urban lakes in the county of Los Angeles (UC Riverside, 1994). Most of the monitoring data were collected in 1992 and 1993, and the findings were summarized in 1994 as the "Evaluation of Water Quality for Selected Lakes in the Los Angeles Hydrologic Basin." Each lake was sampled at one location. Samples reported with the same date were likely replicates. Although raw data were available for the nutrient parameters, pH, total organic carbon (TOC), and total dissolved solids (TDS), only ranges and average chlorophyll *a* concentrations were provided.

The Regional Board completed a Clean Water Act Section 305(b) Water Quality Assessment and Documentation Report for the Los Angeles Region in 1996 (LARWQCB, 1996). This report identifies the impaired waters, summarizes the impairments for each lake, and provides data summaries for dissolved oxygen (DO), pH, and ammonia. A database of water quality monitoring was provided to Tetra Tech along with the Water Quality Assessment and Documentation Report. Although the database does contain limited water quality data for a few of the nutrient impaired lakes, it does not contain the raw data associated with the data summaries of DO, pH, and ammonia as listed in the report. While the data summaries are useful to explain the initial listings, they do not provide the level of detail required to directly apply the data (sampling location, depth, time, relationship to other monitored parameters, etc.).

More recently the Regional Board has collected water quality data in several of these lakes. Much of these data were collected in 2008 and 2009, with some additional metals, organics, and nutrient sampling in 2010. In addition, a few of the lakes have been studied independently over the past several years by other municipal agencies.

This appendix summarizes the water quality data collected in each lake and associated watershed through fall 2010. Where applicable, these data were used to support model development and/or TMDL calculations.

In addition to displaying the locations of the various monitoring stations for each study, the figures in this appendix also show the subwatershed boundaries and the incorporated areas comprising each watershed. In general, the areas draining to storm drain networks will receive waste load allocations for the TMDL, while the other drainage areas will receive load allocations. The areas associated with wasteload and load allocations are described in each lake chapter. Tetra Tech made slight modifications to the subwatershed boundaries that were downloaded from the county of Los Angeles GIS data depot. These minor modifications were based on aerial photographs and digital elevation models. Most changes were made to coordinate subwatershed boundaries with a sampling location, to move the boundary outside of the arms of each lake, or to aggregate subwatersheds to larger TMDL subwatersheds. Modifications are explained in the general information section for each lake.

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G.4 Monitoring Data for Peck Road Park Lake

Monitoring data relevant to the impairments of Peck Road Park Lake are available from 1992, 1993, 2008, 2009, and 2010. In addition, tributary data are available sporadically from 1977 through 1997 and fish tissue data are available from 1986 through 2007. Figure G-2 shows the historical and recent monitoring locations for Peck Road Park Lake.

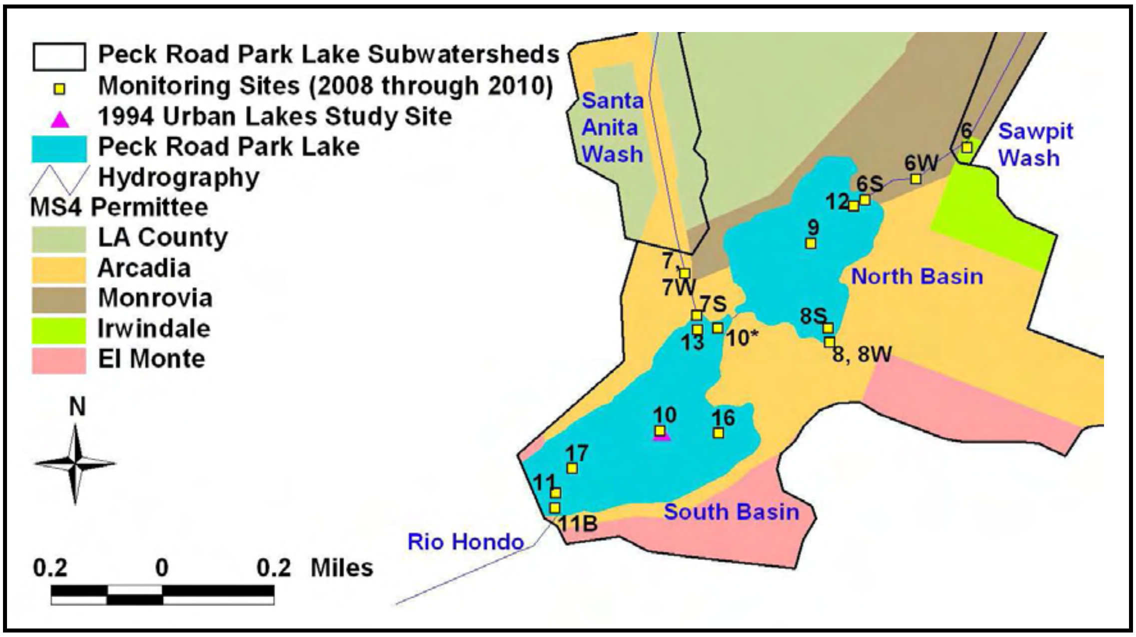

Figure G-2. Peck Road Park Lake Monitoring Sites

G.4.1 Monitoring Related to Nutrient Impairments

The nutrient and pH data collected during the 1992-93 monitoring period in support of the Urban Lakes Study are shown in Table G-2. Samples were collected from the middle of the south basin (pink triangle, Figure G-2). Unfortunately, nutrient levels were analyzed at relatively high detection limits.

Of the 90 orthophosphate samples collected, only one exceeds the reporting limit of 0.1 mg/L. This measurement was collected at a depth of 8 meters and had a value of 0.4 mg/L. Only 1 of 90 total phosphorus samples exceeded the reporting limit of 0.1 mg/L: at a depth of 5 meters the TP measurement was 0.9 mg/L.

Three nitrite samples exceeded the reporting limit for this dataset of 0.1 mg/L. All three had values of 0.2 mg/L and were located at depths ranging from 7 to 14 meters. For nitrate, 23 samples were less than the reporting limit and the maximum nitrate concentration measured was 1.1 mg/L. Twelve measurements of Total Kjeldahl Nitrogen (TKN), which includes the organic and ammonia species of nitrogen, were less than the reporting limit and the maximum TKN concentration observed was 2.0 mg/L. For ammonia, 55 out of 90 measurements were less than the reporting limit and 35 samples ranged from 0.1 mg/L to 1.2 mg/L. pH ranged from 7.3 to 8.8. Total organic carbon (TOC) concentrations ranged from 0.4 mg/L to 4.7 mg/L.

The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from <1 μg/L to 19 μg/L with an average of 8 μg/L. The graphs displaying the depth profile data for Peck Road Park Lake show that dissolved oxygen typically declines to 0 mg/L during the summer months at depths greater than 5 meters. At depths less than 5 meters, dissolved oxygen concentrations were typically around 7 mg/L during the summer months.

Table G-2. Peck Road Park Lake 1992/1993 Monitoring Data for Nutrients

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/7/1992	0	2	<0.1	<0.1	0.1	<0.1	<0.1	7.3	3	173
	3	0.2	<0.1	<0.1	0.1	<0.1	<0.1	7.8	2.2	169
	5.5	0.2	<0.1	<0.1	0.2	<0.1	<0.1	8	4	172
	7	0.3	<0.1	<0.1	0.2	<0.1	<0.1	7.9	2.3	156
	9	0.5	0.2	<0.1	0.1	<0.1	<0.1	7.8	1.8	162
7/7/1992	0	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.3	2.2	151
	3	0.2	<0.1	<0.1	0.1	<0.1	<0.1	8.3	3.1	169
	5.5	0.3	<0.1	<0.1	0.2	<0.1	<0.1	8.2	2.8	171
	6	0.1	<0.1	<0.1	0.2	<0.1	<0.1	8.2	2	171
	7.5	0.4	0.3	<0.1	0.1	<0.1	<0.1	8	3.5	170
7/23/1992	0	0.5	0.9	<0.1	0.1	<0.1	<0.1	8.1	1.2	260
	3.5	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.1	3	245
	6.5	0.2	<0.1	<0.1	0.2	<0.1	<0.1	8	1.2	242
	8.5	0.4	0.2	<0.1	<0.1	<0.1	<0.1	7.9	1.3	240
	10.5	0.5	0.2	<0.1	<0.1	<0.1	<0.1	7.8	1.1	255
	13	1	0.9	<0.1	<0.1	0.1	<0.1	7.7	2.1	223
7/23/1992	0	0.3	0.5	<0.1	<0.1	<0.1	<0.1	8.4	1.3	174
	2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	3.9	185
	4	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.2	1.2	198
7/23/1992	0	0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.5	1.5	167
	2	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.5	4.2	185
	4	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.3	1	189
	6	<0.1	0.3	<0.1	0.2	<0.1	<0.1	8.1	1.1	216
	8	0.4	0.3	<0.1	<0.1	<0.1	0.1	7.9	1.3	174
9/9/1992	0	0.2	0.3	<0.1	<0.1	<0.1	<0.1	8.4	1.9	182
	2.5	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	1.9	177
	4.5	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	1.9	175
	6.5	0.4	<0.1	<0.1	<0.1	<0.1	<0.1	8.3	3.3	174
	8.5	1.2	0.7	<0.1	<0.1	<0.1	<0.1	8.2	4.4	168

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	10	1.3	1.2	<0.1	<0.1	0.1	<0.1	8.4	4.5	167
10/8/1992	0	0.1	0.1	<0.1	<0.1	<0.1	<0.1	8.6	2.3	185
	2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.7	2.3	180
	4	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.6	2.2	180
	6	<0.1	0.1	<0.1	<0.1	<0.1	<0.1	8.3	2.8	176
	8	1.4	1.2	<0.1	<0.1	0.4	<0.1	7.8	4.4	182
	9	1.7	1.2	<0.1	<0.1	<0.1	<0.1	7.7	4.7	169
11/3/1992	0	0.7	<0.1	<0.1	0.7	<0.1	<0.1	8.4	2.9	222
	2.5	0.6	<0.1	<0.1	0.7	<0.1	<0.1	8.3	3.2	229
	5	1.1	0.1	<0.1	0.8	<0.1	<0.1	8.2	2.9	221
	7.5	0.5	0.2	<0.1	0.8	<0.1	<0.1	8.1	2.7	209
	9.5	0.6	0.1	<0.1	0.7	<0.1	<0.1	8	2.7	271
	11.6	1	0.4	<0.1	0.2	<0.1	<0.1	7.9	3.5	153
12/17/1992	0	0.6	0.3	<0.1	0.5	<0.1	<0.1	7.9	3.1	188
	2	0.9	0.2	<0.1	0.5	<0.1	<0.1	7.9	3.2	191
	4.5	1	0.2	<0.1	0.4	<0.1	<0.1	8	3	180
	7.5	1.1	0.2	<0.1	0.5	<0.1	<0.1	8	3	184
	10.5	0.7	0.3	<0.1	0.4	<0.1	<0.1	8	3.6	179
	12.5	0.8	0.3	<0.1	0.5	<0.1	<0.1	8	3	184
1/27/1993	0	0.2	<0.1	<0.1	1.1	<0.1	<0.1	8.1	2	133
	4	0.3	<0.1	<0.1	1	<0.1	<0.1	8	2.5	116
	8	0.3	0.1	<0.1	1	<0.1	0.1	8	2.3	116
	12	0.4	0.1	<0.1	1.1	<0.1	<0.1	8	2.1	133
	16	0.4	0.1	<0.1	1.1	<0.1	<0.1	8.1	2	137
	20	0.3	<0.1	<0.1	1.1	<0.1	<0.1	8.1	1.9	129
2/16/1993	0	0.8	<0.1	<0.1	1	<0.1	0.1	8.6	2.4	148
	2	0.6	<0.1	<0.1	1	<0.1	0.1	8.5	3	123
	5	0.7	<0.1	<0.1	1.1	<0.1	0.9	8.3	2	145
	8	0.3	0.1	<0.1	1.1	<0.1	<0.1	8.2	1.9	142
	11	<0.1	0.2	<0.1	1.1	<0.1	<0.1	8.2	2.3	167
	14.5	0.5	0.2	<0.1	1.1	<0.1	<0.1	8.2	1.9	151
2/25/1993	0	0.3	<0.1	<0.1	1	<0.1	<0.1	8.1	2.2	126

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	3	0.3	<0.1	<0.1	1	<0.1	<0.1	8.1	2.1	134
	6	0.2	<0.1	<0.1	0.9	0.1	<0.1	8.1	2.3	135
	9	0.1	<0.1	<0.1	0.9	0.1	<0.1	8.1	2.2	128
	12	0.3	<0.1	<0.1	0.9	<0.1	<0.1	8.2	2	131
	15.5	0.3	<0.1	<0.1	0.9	<0.1	<0.1	8.2	2	122
3/17/1993	0	0.2	<0.1	<0.1	0.8	<0.1	<0.1	8.7	1.8	163
	3	0.3	<0.1	<0.1	0.8	<0.1	<0.1	8.6	1.7	167
	6	0.2	<0.1	<0.1	0.9	<0.1	<0.1	8.3	2.3	148
	9	0.3	<0.1	<0.1	1.1	<0.1	<0.1	8.3	1.9	141
	12.5	0.1	<0.1	<0.1	1.1	<0.1	<0.1	8.1	1.9	154
	16	0.2	<0.1	<0.1	1.1	<0.1	<0.1	8.3	1.8	146
4/22/1993	0	<0.1	<0.1	<0.1	0.5	<0.1	<0.1	8.8	0.8	178
	2	<0.1	<0.1	<0.1	0.4	<0.1	<0.1	8.8	1.1	173
	5	<0.1	<0.1	<0.1	0.5	<0.1	<0.1	8.6	0.4	191
	8	<0.1	<0.1	<0.1	0.9	<0.1	<0.1	8.1	0.9	157
	11	<0.1	<0.1	<0.1	1	<0.1	<0.1	7.9	0.8	159
	14.5	<0.1	<0.1	<0.1	1.1	<0.1	<0.1	7.9	0.8	155
5/25/1993	0	0.4	<0.1	<0.1	0.3	<0.1	<0.1	8.7	2	201
	3.5	0.4	<0.1	<0.1	0.3	<0.1	<0.1	8.7	2.4	185
	6.5	0.4	<0.1	<0.1	0.6	<0.1	<0.1	8.2	2	183
	9.5	0.5	0.1	0.2	0.7	<0.1	<0.1	7.8	2	197
	12.5	0.4	<0.1	<0.1	0.6	<0.1	<0.1	8.1	1.7	190
	14	0.4	<0.1	0.2	0.7	<0.1	<0.1	7.8	1.7	162
6/23/1993	0	0.3	<0.1	<0.1	0.1	<0.1	<0.1	8.5	1.1	192
	2	0.2	<0.1	<0.1	0.1	<0.1	<0.1	7.9	1.2	167
	4	0.4	<0.1	<0.1	0.2	<0.1	<0.1	8.3	1.4	187
	7	0.4	<0.1	0.2	0.4	<0.1	<0.1	8.1	1.2	223
	9.5	0.6	0.3	<0.1	<0.1	<0.1	<0.1	8	1.3	173
	12	0.7	0.4	<0.1	<0.1	<0.1	<0.1	8.5	1.4	184

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Peck Road Park Lake states that dissolved oxygen (DO) was not supporting the aquatic life use: 195 measurements of DO were collected in the lake with concentrations ranging from 0.2 mg/L to 15.2 mg/L. The accompanying database does not contain the raw data associated with these measurements, so depth, temperature, date, and time cannot be established. The summary table also lists the odor impairment as not supporting both contact and non-contact recreation uses.

For Peck Road Park Lake, the 1996 water quality database contained eight station locations in the watershed; no stations were located within the lake for direct comparison to water quality standards. Table G-3 describes the stations contained in the database. Tetra Tech assigned labels to each site for mapping purposes. Data for Waterbody/Station ID combinations that had identical spatial coordinates were combined under one label. Table G-4 lists the nutrient data contained in the Water Quality Assessment Database for these locations.

Table G-3. Site Locations in the 1996 Water Quality Database for the Peck Road Park Lake Watershed

Waterbody	Station ID	Label
PECK ROAD SPREADING BASIN	Inlet	PRSB
Sawpit Wash	gage abv Peck Rd	SPWPR
Sawpit Wash	Peck Rd	SPWPR
Sawpit Wash	HUNTINGTON DRIVE	SPWHD
SAWPIT WASH DNS	MONROVIA CREEK	MCASPW
Santa Anita	[Blank]	SAWBEF
Santa Anita Cyn	[Blank]	SAWBEF
Santa Anita Wash	blw Live Oak Ave	SAWLO
Santa Anita Wsh	Live Oak Ave	SAWLO
Santa Anita Wsh	Colorado Blvd	SAWCB
STAFTH	SANTAANITAWASH@FOOTHILLBLVD	SAWFB

Table G-4. Water Quality Assessment Data for Tributaries in the Peck Road Park Lake Watershed

Date	Time	Station ID	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	pH
12/27/1977	10:34	PRSB	N/A	N/A	N/A	N/A	N/A	7.1
12/27/1977	07:00	MCASPW	N/A	N/A	N/A	N/A	N/A	7.6
12/28/1977	12:54	SPWHD	N/A	N/A	N/A	N/A	N/A	7.5
1/4/1978	16:15	MCASPW	N/A	N/A	N/A	N/A	N/A	7.4
1/6/1978	11:00	SPWHD	N/A	N/A	N/A	N/A	N/A	7.9
1/6/1978		SAWFB	N/A	N/A	N/A	N/A	N/A	7.2
1/10/1978	11:15	PRSB	N/A	N/A	N/A	N/A	N/A	7.5
1/16/1978	21:15	PRSB	N/A	N/A	N/A	N/A	N/A	7.8

Date	Time	Station ID	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	pH
2/7/1978	20:45	PRSB	N/A	N/A	N/A	N/A	N/A	7.0
2/9/1978	10:20	SPWHD	N/A	N/A	N/A	N/A	N/A	7.5
2/9/1978	10:09	SAWFB	N/A	N/A	N/A	N/A	N/A	7.2
2/28/1978	20:05	PRSB	N/A	N/A	N/A	N/A	N/A	7.5
3/2/1978	10:26	SPWHD	N/A	N/A	N/A	N/A	N/A	7.6
3/2/1978	10:20	SAWFB	N/A	N/A	N/A	N/A	N/A	7.4
3/4/1978	08:10	PRSB	N/A	N/A	N/A	N/A	N/A	7.3
3/18/1980		SAWBEF	0.60	N/A	N/A	N/A	<0.01	N/A
8/13/1980		SAWBEF	<0.1	N/A	N/A	N/A	0.08	N/A
12/3/1980		SAWBEF	0.10	N/A	N/A	N/A	0.05	N/A
4/13/1981		SAWBEF	N/A	N/A	N/A	1.30	<0.01	8.5
8/5/1981		SAWBEF	N/A	N/A	N/A	0.70	N/A	8.3
11/9/1981		SAWBEF	N/A	N/A	N/A	0.50	0.03	8.1
3/11/1982		SAWBEF	N/A	N/A	N/A	0.60	0.03	8.1
5/27/1982		SAWBEF	N/A	N/A	N/A	0.56	0.02	7.8
8/13/1982	14:40	SAWBEF	N/A	N/A	N/A	1.37	0.06	8.3
10/26/1982	14:00	SAWBEF	N/A	N/A	N/A	1.00	<0.03	8.1
5/6/1983		SAWBEF	N/A	N/A	N/A	0.47	0.09	8.2
9/22/1983	14:45	SAWBEF	N/A	N/A	N/A	0.70	0.05	8.0
8/21/1987		SAWBEF	N/A	N/A	N/A	0.23	0.07	7.9
9/30/1988		SAWBEF	N/A	N/A	N/A	0.20	N/A	7.8
10/2/1989		SAWBEF	N/A	N/A	N/A	0.16	<0.16	7.9
10/2/1990		SAWBEF	N/A	N/A	N/A	0.19	<0.1	7.8
4/18/1991		SPWPR	N/A	0.80	<0.03	0.80	0.06	9.0
5/8/1991		SAWBEF	N/A	N/A	N/A	0.40	0.14	8.2
5/14/1992		SAWCB	N/A	N/A	N/A	N/A	N/A	8.4
5/14/1992		SAWLO	N/A	N/A	N/A	N/A	N/A	8.4
12/22/1992		SAWLO	N/A	N/A	<0.03	4.40	N/A	N/A
12/22/1992		SPWPR	N/A	N/A	<0.03	<0.2	N/A	N/A
5/13/1997	09:55	SAWLO	3.60	0.80	<0.03	<0.2	0.10	8.7
5/13/1997	10:15	SPWPR	0.60	0.20	<0.03	<0.2	0.02	8.6

N/A = No data available.

In 2008, Regional Board collected water quality samples from several locations in Peck Road Park Lake and its inflows (Figure G-2). Site location information is listed in Table G-5.

Table G-5. Site Locations for the 2008 Peck Road Park Lake Monitoring Event

Site number	Project Site	Comment
Inflows		
6	Sawpit Wash (SPW)	Inflow
7	Santa Anita Wash (SAW)	Inflow
20	Santa Anita Wash (SAW)	Site 7-Field Duplicate
8	North Basin Outfall (NBO)	Stormwater outfall to North Basin
Mid-Lake Sites		
9	North Basin (NB)	
10	South Basin (SB)	This site was moved to the narrow section connecting the north and south basins for the December sampling event
16	South Basin (SB)	
17	South Basin (SB)	

Analytical data for the June 17, 2008 sampling event are listed in Table G-6 (sites 16 and 17 were not monitored during this event). Four of the six sites had NH_3-N concentrations less than the reporting limit of 0.1 mg/L; the maximum ammonia concentration was 0.437 mg/L. TKN ranged from 1.2 mg/L to 10 mg/L, with the higher concentrations observed at the two major inflow sites (6, 7/20). Nitrate concentrations ranged from 0.22 mg/L to 0.58 mg/L with two measurements less than the reporting limit of 0.1 mg/L. Each site had measurements of nitrite and orthophosphate less than the reporting limits of 0.1 mg/L and 0.4 mg/L, respectively. All but one site (Sawpit Wash) had total phosphate concentrations less than the reporting limit of 0.5 mg/L.

Table G-6. Analytical Data for the June 17, 2008 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total phosphate (mg/L)	Total Dissolved Solids (mg/L)
Inflows							
6	0.179	6.5	<0.1	<0.1	<0.4	0.715	303
7	<0.1	9.26	0.58	<0.1	<0.4	<0.5	1,517
20	<0.1	10	0.44	<0.1	<0.4	<0.5	1,401
8	<0.1	1.32	0.22	<0.1	<0.4	<0.5	145
Mid-lake Sites							
9	<0.1	1.2	0.24	<0.1	<0.4	<0.5	142
10	0.437	2.08	<RL	<0.1	<0.4	<0.5	134

Field data collected at these sites are summarized in Table G-7. Field data were collected in Peck Road Park Lake (sites 10 and 9) at depths ranging from the water surface to 2.5 meters. Temperature varied by approximately 1 ºC in the south basin and approximately 4 ºC in the north basin over the sampling depth. Dissolved oxygen in the lake was elevated at all depths except station 9 at a depth of 2.5 meters. Electrical conductivity was fairly constant at both sites and ranged from 0.17 mS/cm to 0.185 mS/cm. pH measurements in the lake ranged from 8.0 to 9.4, although the meter was not calibrated due to equipment malfunction and this data should not be used quantitatively. Chlorophyll *a* measurements in the lake ranged from 4.0 µg/L to 11.4 µg/L. The total depth at site 10 was approximately 8 meters and the Secchi depth was 3.2 meters; the total depth at site 9 was approximately 5.2 meters and the Secchi depth was 2.3 meters.

Site 6 (Sawpit Wash) had the highest observed temperature (33.5 ºC) and chlorophyll *a* concentration (16 µg/L). Dissolved oxygen was 9 mg/L; electrical conductivity was about two times higher than that observed in the lake. Site 7 (Santa Anita Wash) had observed temperatures slightly less than that measured in the lake; dissolved oxygen was approximately 11 mg/L. Electrical conductivity at this site was the highest (1.726 mS/cm); chlorophyll *a* was slightly higher than the majority of measurements taken in the lake (10.4 µg/L). Site 8 is a stormwater outfall at the downstream end of the north basin. Readings at this site were generally similar to those measured in the lake.

Table G-7. Field Data for the June 17, 2008 Peck Road Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	DO (mg/L)	EC (mS/cm)	pH[1]	Chl a (µg/L)	Secchi Depth (m)
10	Samples at this site collected between 10:17 and 11:05	surface	26.4	17.8	0.17	9.3	4	3.2
		0.5	26	18.7	0.17	9.4	9.4	
		1	25.7	19	0.17	9.4	4.9	
		1.5	25.6	19.2	0.17	9.4	6	
		2	25.5	19.4	0.17	9.4	6.7	
		2.5	25.4	19.5	0.17	9.4	7.5	
9	Samples at this site collected between 12:38 and 13:30	surface	28.32	18.67	0.185	9.32	4.1	2.3
		0.5	27.62	18.77	0.184	9.3	3.8	
		1	26.59	19.46	0.183	9.23	4.8	
		1.5	26.18	19.78	0.182	9.14	6.6	
		2	25.8	17	0.18	8.9	7.8	
		2.5	23.9	3	0.18	8	11.4	
8	14:57	surface	29.5	20.1	0.18	9.4	6.2	NA
6	17:18	surface	33.5	9	0.37	9.9	16	NA
7	18:40	surface	24.31	11.17	1.726	9.62	10.4	NA

[1] pH calibration was outside of accepted range. Data should not be used quantitatively.

Four sites were sampled by the Regional Board on December 11, 2008. Samples were collected from the surface at each site. Table G-8 summarizes the nutrient data collected. Measurements of TKN, nitrite, orthophosphate, and total phosphate were less than the reporting limit at each site. Ammonia concentrations ranged from 0.209 mg/L to 0.273 mg/L; nitrate ranged from 0.162 mg/L to 0.287 mg/L. Total dissolved solids ranged from 154 mg/L to 178 mg/L. Suspended solids were less than the reporting limit at each site except for site 16. Chlorophyll *a* ranged from 1.8 µg/L to 4.0 µg/L.

Table G-8. Analytical Data for the December 11, 2008 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho phosphate (mg/L)	Total phosphate (mg/L)	Total Dissolved Solids (mg/L	Suspended Solids (mg/L)	Chl *a* (µg/L)
Mid-Lake Sites									
9	0.273	<1.0	0.287	<0.1	<0.4	<0.5	154	<10	4.0
10*	0.209	<1.0	0.173	<0.1	<0.4	<0.5	178	<10	3.6
16	0.262	<1.0	0.164	<0.1	<0.4	<0.5	175	24	3.1
17	0.269	<1.0	0.162	<0.1	<0.4	<0.5	165	<10	1.8

Field data for the December 11, 2008 event are summarized in Table G-9.

Table G-9. Field Data for the December 11, 2008 Peck Road Park Lake Sampling Event

Station Number	Time	Depth (m)	Temp (C)	DO[1] (mg/L)	EC (mS/cm)	pH	Secchi Depth (m)	Total Depth (m)
9	8:15	Surface	15.61	NA	0.206	7.51	1.6	6.4
		0.5	15.6	2.44	0.204	7.48		
		1.0	15.6	2.29	0.204	7.48		
		1.5	15.59	2.23	0.204	7.47		
		2.0	15.59	2.21	0.204	7.47		
10*	10:07	Surface	16.30	6.15	0.234	7.79	1.1	2.3
		0.5	15.99	6.20	0.231	7.80		
		1.0	15.72	5.41	0.227	7.70		
		1.5	15.52	3.91	0.215	7.53		
		2.0	15.47	3.27	0.213	7.51		
16	11:45	Surface	17.29	5.77	0.237	7.74	1.6	2.1
		0.5	16.44	6.15	0.236	7.81		
		1.0	16.26	6.07	0.236	7.80		
		1.5	16.19	5.71	0.236	7.78		
		2.0	16.15	5.57	0.236	7.76		
17	12:24	Surface	16.62	4.94	0.236	7.70	1.8	11.1
		0.5	16.20	4.78	0.236	7.69		
		1.0	16.14	4.61	0.236	7.68		
		1.5	16.11	4.54	0.236	7.68		
		2.0	16.09	4.56	0.236	7.69		

[1] Field team questioned measurement of DO for this event. Meter was not calibrated prior to sampling.

Water quality monitoring was also conducted by the Regional Board on August 5, 2009 at Stations 9 and 10. The data from this event are shown in Table G-10.

Table G-10. Analytical Data for the August 5, 2009 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho phosphate (mg/L)	Total phosphorus (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)	Chl *a* (µg/L)
PRPL-9	<0.03	<0.456	<0.01	<0.01	0.0135	0.022	205	3.9	8.0
PRPL-10	<0.03	<0.456	<0.01	<0.01	0.0112	0.116	194	3.2	5.3

Profile data were also collected at stations PRPL-9 and PRPL-10 on August 5, 2009. Profiles were performed at 9:00 a.m. at station PRPL-9, and at 8:00 a.m. and 3:00 p.m. at station PRPL-10. These data are displayed in Figure G-3 through Figure G-5. At station PRPL-9, the specific conductivity was between 0.340 and 0.373 mS/cm. The pH ranged from 7.69 and 8.56. The maximum DO in the lake was 11.79 mg/L, at a depth of 1.48 meters. Below 1.48 meters, the DO steadily declines to 3.34 mg/L at 7 meters of depth. The temperature near the surface of the water was 24.5 °C and starts to decline at 1 meter of depth. The minimum temperature was 16.05 °C.

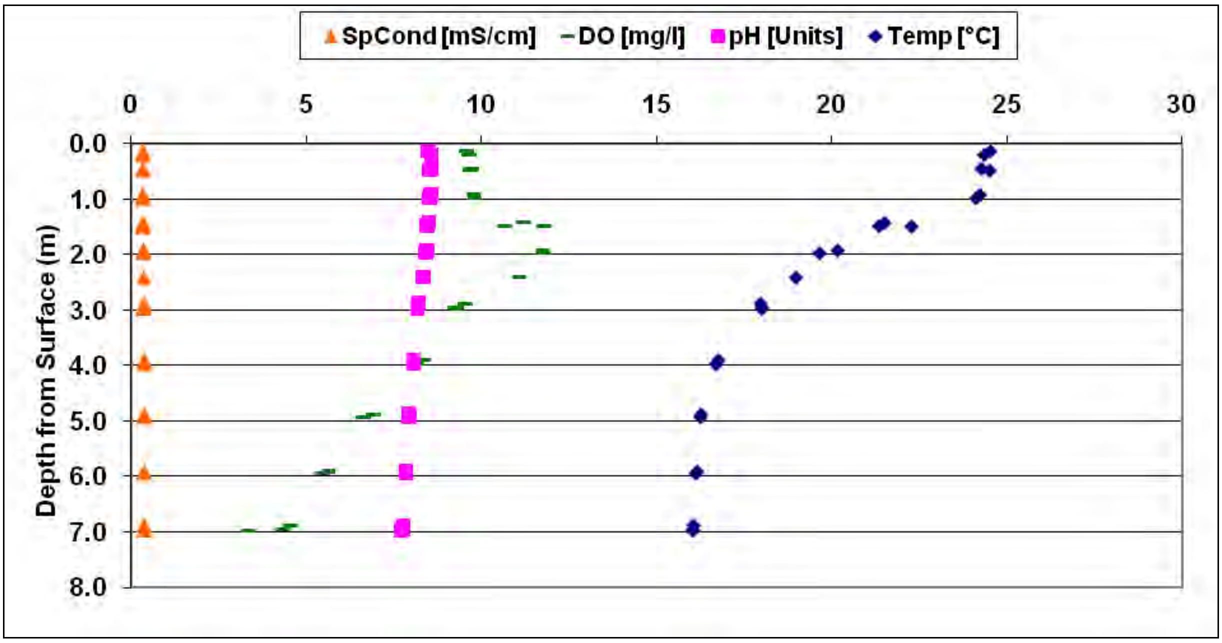

Figure G-3. Profile Data Collected at PRPL-9 in Peck Road Park Lake on August 5, 2009

Profile data were collected at the PRPL-10 at 8:00 a.m. and 3:00 p.m. The morning profile is shown in Figure G-4. The temperature in the lake ranges from 16.44 to 23.38 °C. The maximum DO is 11.78 mg/L and occurs at 2.22 meters of depth. The minimum DO was 2 mg/L at 10 meters of depth. The pH ranged from 8.45 to 7.51. The specific conductivity was constant with depth.

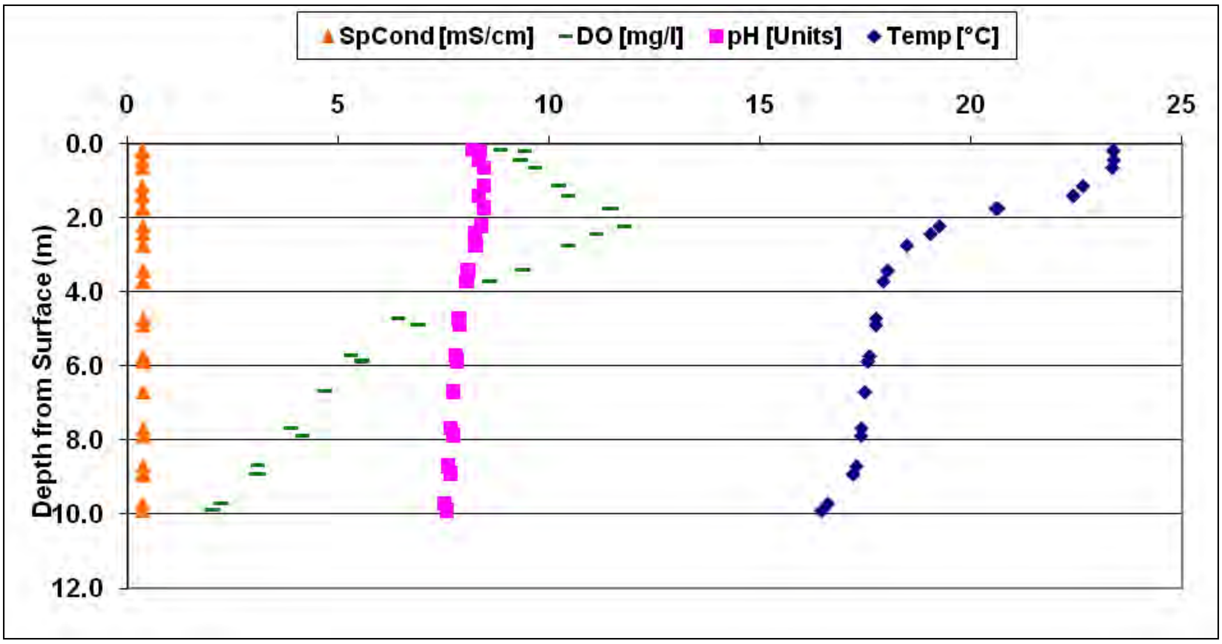

Figure G-4. Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009 at 8:00 a.m.

The profile data collected in the afternoon at Station PRPL-10 is shown in Figure G-5. The specific conductivity was constant with depth and the pH ranged from 7.53 to 8.71. The maximum DO was 12.02 mg/L at 2.03 meters. The DO decreased with depth after 2 meters, to a minimum of 1.22 mg/L. The temperature of the lake was between 24.04 and 17.07°C. Similar to the DO, temperature decreased with depth after 2.03 meters.

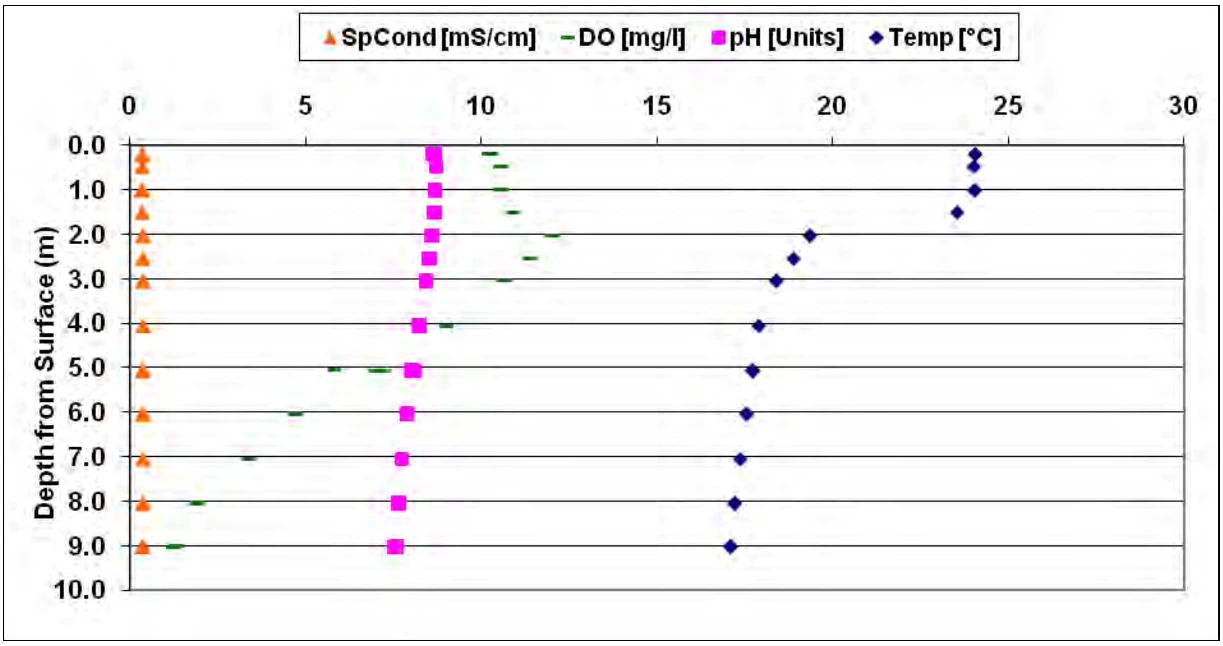

Figure G-5. Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009 at 3:00 p.m.

The DO saturation for the morning and afternoon readings at PRPL-10 are shown below in Figure G-6. The DO saturation ranges from 13 to 131 percent. DO saturation above 100 percent indicates additional oxygen input from algal productivity. The maximum DO saturation occurs at 2 meters of depth in the euphotic zone.

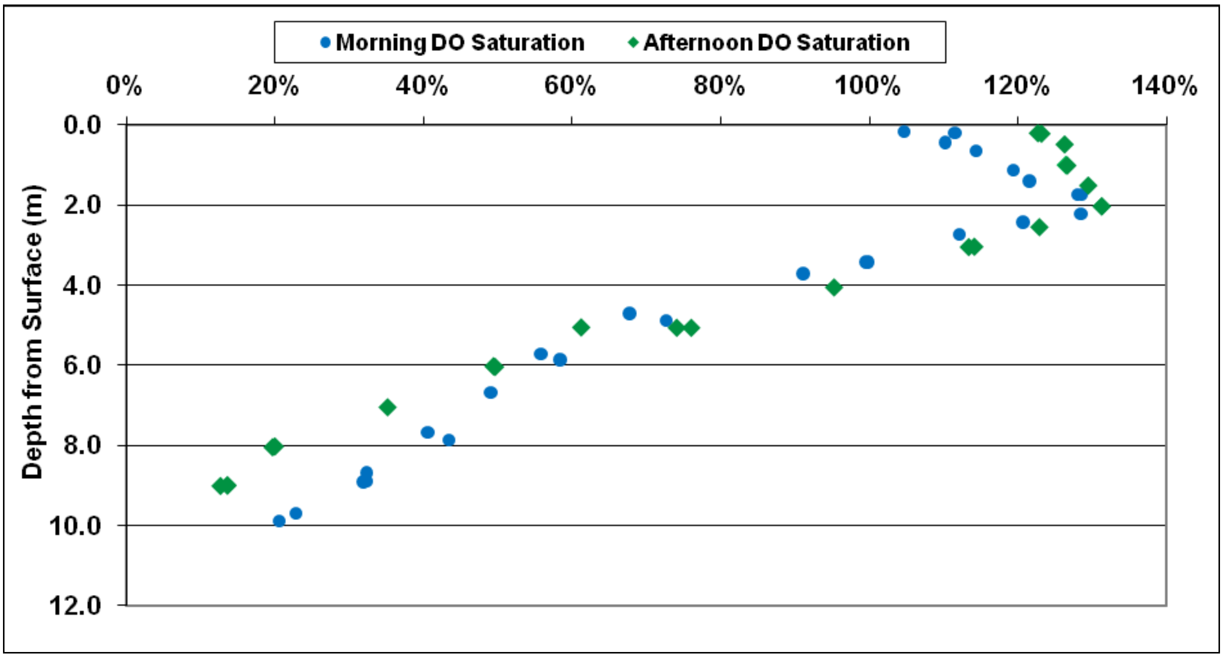

Figure G-6. DO Saturation from Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009

On September 30, 2010, additional sampling was conducted at the mid-lake sites (Table G-11). Ammonia concentrations were below the detection limit of 0.03 mg-N/L. Nitrite ranged from 0.041 to 0.043 mg-N/L, and nitrate was below the detection limit of 0.01 mg-N/L (note: nitrite values were higher than nitrate. These samples passed the laboratory QA/QC protocols, so they are considered valid). TKN ranged from 0.562 to 0.634 mg-N/L. Orthophosphate and total phosphorus ranged from 0.02 mg-P/L to 0.04 mg-P/L. Chlorophyll *a* ranged from 6.7 µg/L to 13.4 µg/L.

Table G-11. Analytical Data for the September 30, 2010 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho Phosphate (mg/L)	Total Phosphorus (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)	Chl *a* (µg/L)
PRPL-9	<0.03	0.634	<0.01	0.041	0.040	0.040	220	3.25	13.4
PRPL-10	<0.03	0.574	<0.01	0.043	0.024	0.022	200	0.75	6.68
PRPL-10 (Duplicate)	<0.03	0.562	<0.01	0.042	0.025	<0.0165	160	1.50	7.12

During the September 2010 sampling event, two continuous monitoring probes were deployed over a 24-hour period (Figure G-7 and Figure G-8). At an average depth of 0.6 meters, DO concentrations ranged from 8.6 mg/L to 10.1 mg/L. pH ranged from about 8.5 to 8.8. On September 30, 2010, depth profile measurements were also taken and are shown in Table G-12, Figure G-9, and Figure G-10. DO measurements collected from the surface of the lake ranged from 8.5 mg/L to 10.9 mg/L. At 2 meters above the bottom, DO ranged from 0.2 to 4.0 mg/L. Specific conductivity was not recorded during the profile measurements.

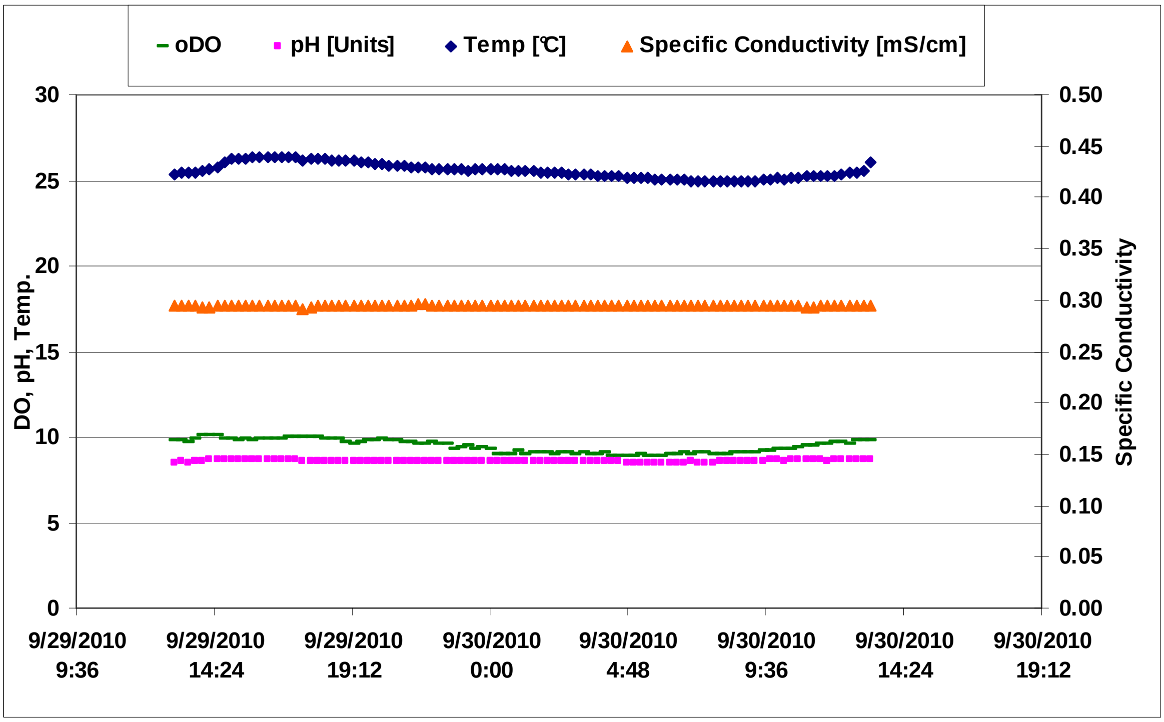

Figure G-7. 24-Hour Probe Data Collected at PRPL-9 on September 29, 2010

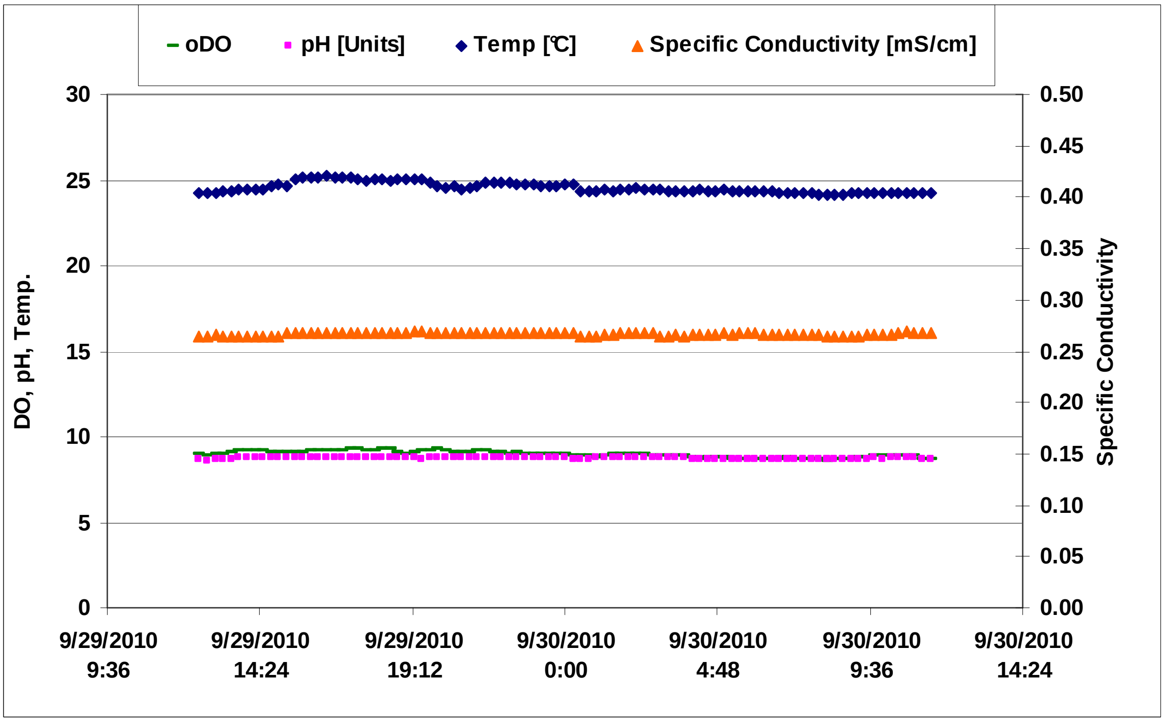

Figure G-8. 24-Hour Probe Data Collected at PRPL-10 on September 29, 2010

Table G-12. Profile Data Collected at Peck Road Park Lake on September 30, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
PRPL-9	11:15	0.5	25.61	8.49	10.87	25.2
		1	25.10	8.51	11.04	24.7
		1.5	24.76	8.52	11.20	24.9
		2	24.32	8.36	10.40	32.8
		2.5	23.37	8.18	9.10	37.8
		3	22.70	7.83	--	48.7
		3.5	22.25	7.41	2.83	50.3
		4	22.02	7.25	0.20	19.5
PRPL-10	11:33	0.5	24.85	8.34	8.54	18.0
		1	24.71	8.37	8.46	17.9
		1.5	24.57	8.41	8.73	15.6
		2	24.51	8.41	8.73	14.4
		2.5	24.40	8.39	8.50	13.1
		3	24.06	8.21	7.12	13.4
		3.5	23.88	8.18	6.92	11.2
		4	23.07	7.71	4.02	20.0

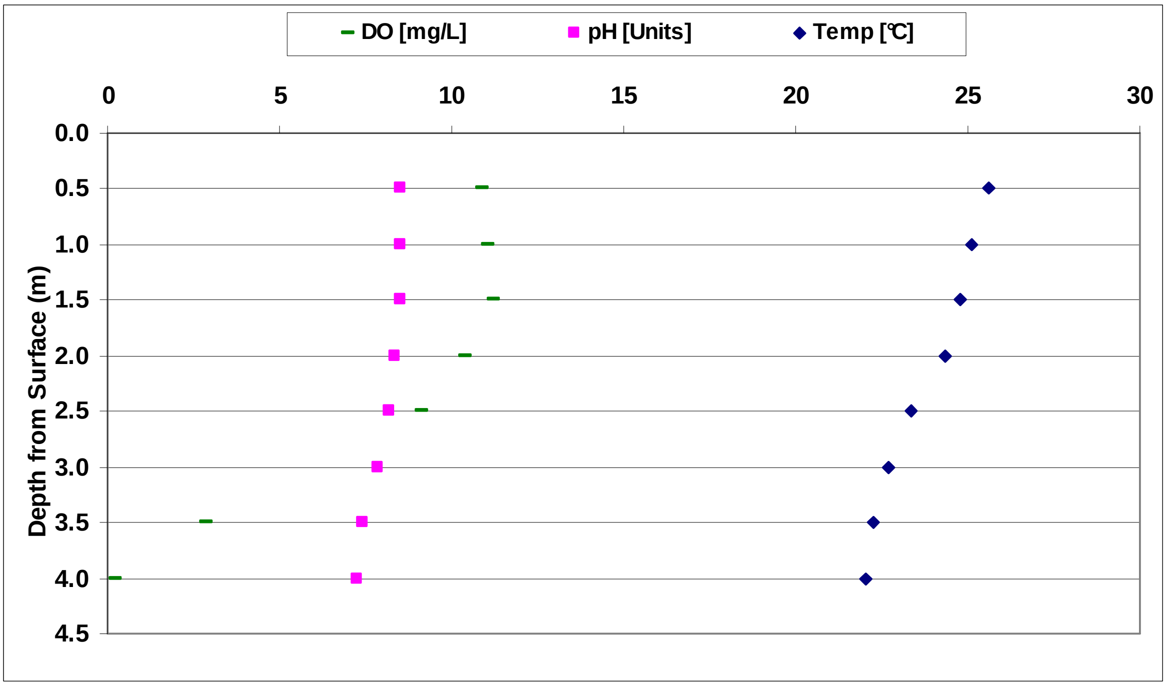

Figure G-9. Profile Data Collected at PRPL-9 on September 30, 2010

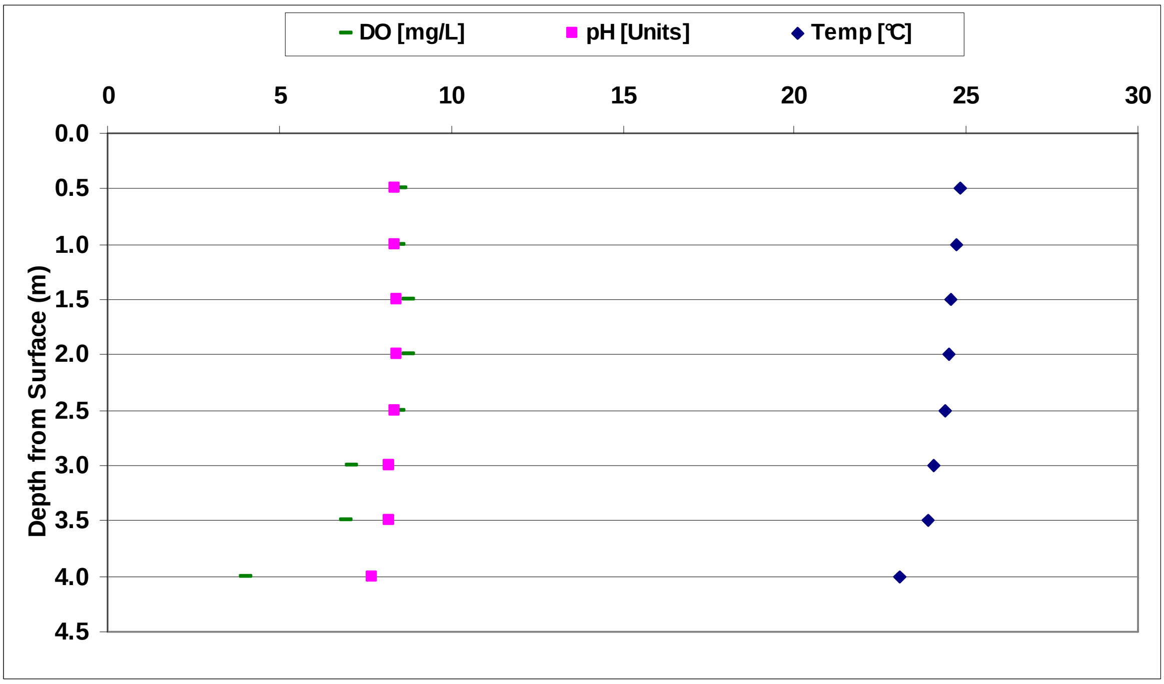

Figure G-10. Profile Data Collected at PRPL-10 on September 30, 2010

The DO saturation at PRPL-10 is shown below in Figure G-11. The DO saturation ranges from 47 to 106 percent. DO saturation above 100 percent indicates additional oxygen input from algal productivity. The maximum DO saturation occurs at 1.5 and 2 meters of depth in the euphotic zone.

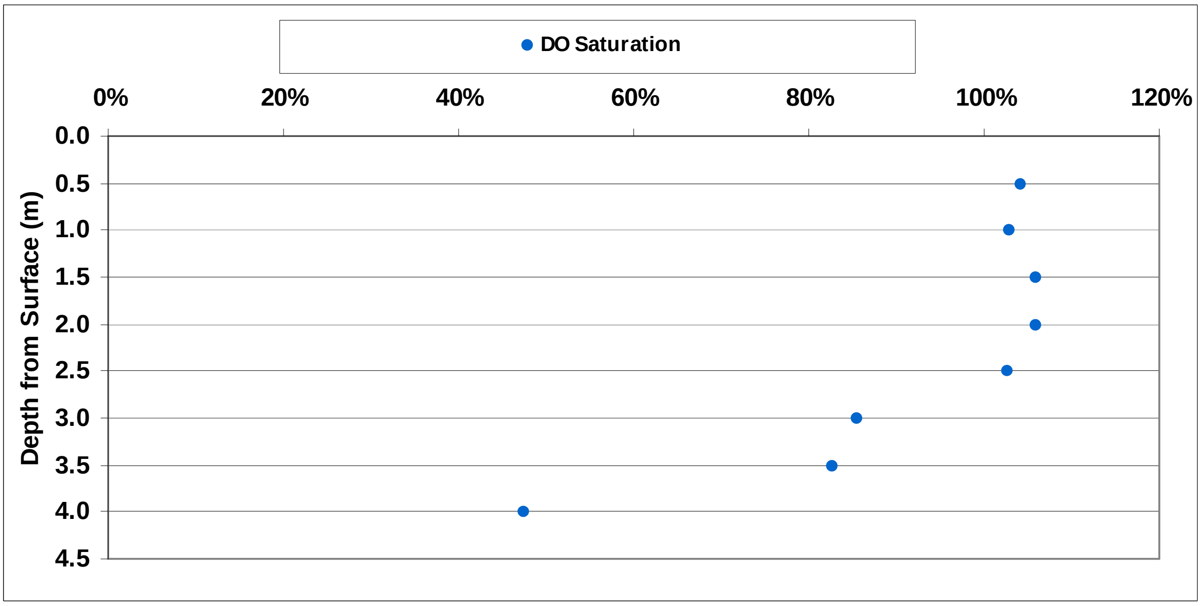

Figure G-11. DO Saturation from Profile Data Collected at PRPL-10 in Peck Road Park Lake on September 30, 2010

Sediment samples were also collected during the September 2010 monitoring event. Table G-13 summarizes these data.

Table G-13. September 30, 2010 Sediment Monitoring Data for Peck Road Park Lake

Location	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hard-ness (mg/kg)
PRPL-9	11:30	5500	36.2	1.15	1.51	0.231	35.3	4.81	30.7	36.5	38400
PRPL-10	10:45	3180	26.6	1.34	1.64	0.0446	92.3	3.98	135	33.6	27400
PRPL-10 (Duplicate)	10:45	4170	28.5	1.31	1.67	0.0337	16.2	4.07	145	33.6	28400

G.4.2 MONITORING RELATED TO LEAD IMPAIRMENT

In 1996 Peck Road Park Lake was deemed impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Peck Road Park Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Peck Road Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are presented in Table G-14. Samples were collected from the middle of the south basin (pink triangle, Figure G-2) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 20 meters. The range of the 90 dissolved copper samples was between less than 10 µg/L and 69 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 20 meters. The 90 samples collected ranged in concentration from less than 1 µg/L to 82 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The 1996 summary table for Peck Road Park Lake states that lead was not supporting the assessed uses: 90 measurements had a maximum lead concentration of 73 µg/L, a maximum copper concentration of 69 µg/L, and a maximum zinc concentration of 47 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-14. Peck Road Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/7/1992	0	<10	<1
	3	<10	<1
	5.5	<10	<1
	7	18	9
	9	47	21
7/7/1992	0	21	<1
	3	22	1
	5.5	15	<1
	6	26	15
	7.5	41	18
7/23/1992	0	<10	11
	3.5	12	<1
	6.5	<10	<1
	8.5	<10	<1
	10.5	<10	<1
	13	N/A	<1
7/23/1992	0	<10	<1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	4	<10	<1
7/23/1992	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
	8	<10	1
9/9/1992	0	12	<1
	2.5	<10	<1
	4.5	<10	1
	6.5	<10	<1
	8.5	10	2
	10	<10	2
10/8/1992	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
	8	<10	<1
	9	<10	<1
11/3/1992	0	55	1
	2.5	18	1
	5	19	1
	7.5	36	1
	9.5	53	2
	11.6	19	3
12/17/1992	0	<10	<1
	2	<10	<1
	4.5	<10	<1
	7.5	<10	<1
	10.5	<10	1
	12.5	<10	<1
1/27/1993	0	<10	<1
	4	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	8	<10	27
	12	<10	16
	16	<10	18
	20	<10	2
2/16/1993	0	<10	<1
	2	15	4
	5	16	6
	8	<10	<1
	11	<10	<1
	14.5	<10	<1
2/25/1993	0	<10	<1
	3	<10	3
	6	<10	<1
	9	<10	<1
	12	<10	<1
	15.5	<10	<1
3/17/1993	0	69	<1
	3	<10	39
	6	<10	43
	9	<10	66
	12.5	<10	53
	16	<10	73
4/22/1993	0	<10	17
	2	<10	43
	5	<10	64
	8	<10	31
	11	<10	33
	14.5	<10	12
5/25/1993	0	<10	6
	3.5	<10	3
	6.5	<10	<1
	9.5	<10	1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	12.5	<10	6
	14	<10	11
6/23/1993	0	<10	82
	2	<10	2
	4	<10	<1
	7	<10	<1
	9.5	<10	<1
	12	<10	<1

Table G-15 presents 30 additional metals samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between December 2008 and September 2010. Samples were collected at locations PRPL-8, PRPL-9, PRPL-10, and PRPL-11B in 2009 and 2010, while PRL 10*/16/17 and PRL 09 were sampled in 2008. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.2 µg/L; copper concentrations ranged from <0.4 µg/L to 10.2 µg/L; lead concentrations were between <0.05 µg/L and 1 µg/L; and zinc concentrations ranged from <0.1 µg/L to 14.8 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 sampling resulted in a hardness range of 40 mg/L to 102 mg/L. In addition, two total lead samples were collected by the Regional Board in June 2008 at PRL 09 (North Basin) and PRL 10 (South Basin). The total lead concentrations were 5.8 µg/L and 11.8 µg/L, respectively (with 96 mg/L and 88 mg/L hardness values, respectively). Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-15 were weighted more heavily in the assessment.

Table G-15. Metals Data for the 2008-2010 Peck Road Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	12/11/2008	PRL 10*/16/17	102	<0.2	1.1	<0.1	2.8	average of stations 10*, 16, and 17
RB	12/11/2008	PRL 09	84	<0.2	1.7	0.1	4.2	average of replicates
RB/EPA	8/5/2009	PRPL 8	121	<0.2	4.7	0.2	4.7	
RB/EPA	8/5/2009	PRPL 9	121	<0.2	10.2	0.3	11.2	average of replicates
RB/EPA	8/5/2009	PRPL 10	122	<0.2	5.1	0.2	7.1	
RB/EPA	8/5/2009	PRPL 11	122	<0.2	4.4	0.1	3.7	
EPA/ County	11/16/2009	PRPL-10	116	<0.2	0.4	<0.1	1.6	average of filtered samples

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
EPA/ County	11/16/2009	PRPL-11B	117	<0.2	0.4	<0.1	<0.1	average of filtered samples
EPA/ County	11/16/2009	PRPL-8	109	<0.2	1.1	0.6	1.7	average of replicates and filtered samples
EPA/ County	11/16/2009	PRPL-9	108	<0.2	0.9	0.3	8	average of duplicate and filtered samples
County	12/8/2009	PRPL-10	114	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	PRPL-11B	113	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	PRPL-8	88	<0.2	3	1	5.9	average of replicates
County	12/8/2009	PRPL-9	87	<0.2	3.3	1	9.4	average of duplicates
EPA	12/14/2009	PRPL-11B	89	<0.2	1.1	<0.1	4.1	
EPA	12/14/2009	PRPL-9	40	<0.2	2.8	0.3	14.6	
EPA	12/14/2009	PRPL-10	83	<0.2	1.2	<0.1	3.7	average of duplicates
EPA	12/14/2009	PRPL-8	40	<0.2	2.9	0.5	14.8	average of replicates
County	1/28/2010	PRPL-11B	63	<0.2	1.8	0.2	3.7	
County	1/28/2010	PRPL-9	59	<0.2	2.3	0.2	7.2	
County	1/28/2010	PRPL-10	63	<0.2	1.8	0.2	6.6	average of duplicates
County	1/28/2010	PRPL-8	59	<0.2	2.3	0.2	4.1	average of replicates
County	2/17/2010	PRPL-11B	59	<0.2	1.8	0.1	3.7	
County	2/17/2010	PRPL-9	73	<0.2	2.1	0.2	6.4	
County	2/17/2010	PRPL-10	64	<0.2	2.1	0.1	3.1	average of duplicates
County	2/17/2010	PRPL-8	59	<0.2	2.0	0.2	5.1	average of replicates
EPA / RB	9/30/2010	PRPL-8	76	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-9	75	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-10	66	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-11	66	<0.2	<0.4	<0.05	<0.1	

RB = Regional Board

EPA = USEPA

County = County of Los Angeles

USEPA also collected two sediment samples during September 2010 to further evaluate lake conditions. Table G-16 summarizes the lead concentrations measured in the samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-16. Sediment Metals Data for the September 2010 Peck Road Park Lake Sampling Event

Organization	Date	Station ID	Lead (mg/kg))	Notes
EPA	09/30/2010	PRPL9	86.8	
EPA	09/30/2010	PRPL10	82.5	Average of duplicates

G.4.3 MONITORING RELATED TO ORGANOCHLORINE PESTICIDES AND PCBS IMPAIRMENTS

The extent of Organochlorine (OC) Pesticides and PCBs in Peck Road Park Lake was assessed through Regional Board sampling and their contracted study with UCLA. Peck Road Park Lake is specifically impaired by chlordane, DDT, dieldrin, and PCBs. The collected data for these contaminants are shown below for the water column, bottom lake sediments, suspended sediment in the water column, porewater, and suspended sediments in the porewater. Fish tissue level data from research by the Toxic Substances Monitoring Program (TSMP) (TSMP, 2009) and Surface Water Ambient Monitoring Program (SWAMP) (SWAMP, 2009; Davis et al., 2008) are also presented here and used in the OC Pesticides and PCBs TMDL.

G.4.3.1 Water Column Data Observed in Peck Road Park Lake

Water column samples were collected in the summer and fall of 2008 for the UCLA study and also on December 11, 2008 by the Regional Board. All pollutants were below detection limits (ND) or quantifiable/reportable levels (DNQ). PCB-31 was detected but not quantifiable at PRPL-10 in summer 2008 and also at PRPL-6W and PRPL-7W in fall 2008. Other PBCs that were DNQ were PCB-18 at PRPL-6W in summer 2008 and PCB-44, PCB-110, and PCB-153 at PRPL-7W in fall 2008. The results from the summer 2008 and fall 2008 samples are shown in Table G-17 and Table G-18 and results from the December 11, 2008 sampling are shown in Table G-19.

Table G-17. Results from Water Column Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant	PRPL-6W			PRPL-7W			PRPL-8W			PRPL-8W (dup)			PRPL-9			PRPL-10		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)																	
Chlordane-gamma	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
Chlordane-alpha	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
4,4'-DDE	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDD	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDT	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
Dieldrin	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
PCB 5	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 18	1.58	15.79	8.64*	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 31	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	4.07*
PCB 52	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 44	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 66	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 101	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 87	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 151	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 110	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 153	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 141	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 138	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 187	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 183	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 180	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 170	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 206	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND

*Result was above the detection limit, but below the reporting limit.

Table G-18. Results from Water Column Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-6W			PRPL-7W (duplicate)			PRPL-7W		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)								
Chlordane-gamma	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
Chlordane-alpha	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
4,4'-DDE	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
4,4'-DDD	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
4,4'-DDT	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
Dieldrin	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
PCB 5	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 18	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 31	1.67	16.67	4.31*	1.67	16.67	ND	1.50	15.00	7.76*
PCB 52	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 44	1.67	16.67	ND	1.67	16.67	1.93*	1.50	15.00	ND
PCB 66	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 101	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 87	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 151	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 110	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	3.02*
PCB 153	1.67	16.67	ND	1.67	16.67	2.88*	1.50	15.00	ND
PCB 141	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 138	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 187	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 183	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 180	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 170	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 206	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND

*Result was above the detection limit, but below the reporting limit.

Table G-19. Results from Water Column Samples Collected at Peck Road Park Lake on December 11, 2008

Contaminant (ng/L)	PRPL-9	PRPL-10*	PRPL-16	PRPL-17	MDL
Chlordane-alpha	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	1.0
2-4’DDD	ND	ND	ND	ND	1.0
2-4’DDE	ND	ND	ND	ND	1.0
2-4’DDT	ND	ND	ND	ND	1.0
4-4’DDD	ND	ND	ND	ND	1.0
4-4’DDE	ND	ND	ND	ND	1.0
4-4’DDT	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	1.0
PCB003	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PRPL-9	PRPL-10*	PRPL-16	PRPL-17	MDL
PCB119	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	1.0

G.4.3.2 Porewater Data Observed in Peck Road Park Lake

Analysis of porewater and porewater suspended solids were performed for PRPL-6S, PRPL-7S, PRPL-9, and PRPL-10 in summer 2008. None of the contaminants were found in the porewater or associated solids at PRPL-6S. PCBs were detected below reporting limits (DNQ) in the water and suspended solids in porewater samples from PRPL-7S and PRPL-9. Three different PCB congeners were detected in the porewater suspended sediment from PRPL-10. No pollutants were detected in the porewater at PRPL-10. The analysis of porewater and suspended solids in porewater are shown in Table G-20, and Table G-21, respectively (see Stenstrom et al., 2009 for raw data).

Table G-20. Results from Porewater Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant (ng/L)	PRPL-6S	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	ND	15
DDT	ND	ND	ND	ND	30
Dieldrin	ND	ND	ND	ND	30
Total PCBs	ND	DNQ[1]	DNQ[2]	ND	15

[1] PCB-31 was detected below reporting limit (150 ng/L)

[2] PCB-5 was detected below reporting limit (150 ng/L).

Table G-21. Results of Porewater Suspended Sediments Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant (µg/kg dry weight)	PRPL- 6S	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	ND	2.26 – 9.25
DDT	ND	ND	ND	ND	4.51 – 18.50
Dieldrin	ND	ND	ND	ND	4.51 – 18.50
Total PCBs	ND	DNQ[1]	DNQ[2]	DNQ[3]	2.26 – 9.25

[1] PCB-52 was detected below reporting limit (22.55 µg/kg dry weight).

[2] PCB-87, PCB-153, PCB-180 were detected below reporting limit (66.03 µg/kg dry weight for each congener).

[3] PCB-160, PCB-145, PCB-187 were detected below reporting limit (59.72 µg/kg dry weight for each congener).

In fall 2008 samples from PRPL-7S, PRPL-9 and PRPL-10 were analyzed for contaminants in porewater. None of the organic chemicals of interest were detected in the samples. The porewater had insufficient TSS for analysis. The results from the fall 2008 analysis are shown in Table G-22.

Table G-22. Results of porewater sampling collected at Peck Road Park Lake in Fall 2008

Contaminant (ng/L)	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	15
DDT	ND	ND	ND	30
Dieldrin	ND	ND	ND	30
Total PCBs	ND	ND	ND	15

G.4.3.3 Fish Tissue Data Observed in Peck Road Park Lake

Concentrations of Aroclor PCBs, chlordane, DDTs, dieldrin, and PCBs in fish tissue are shown for Peck Road Park Lake in Table G-23. Largemouth bass were the only fish species collected from Peck Road Park Lake. Aroclor PCBs were not detected in the fish samples. The average chlordane and DDT concentrations (17.2 ppb chlordane and 21.8 ppb DDTs) are both above OEHHA 2008 Fish Contaminant Goals (FCGs) for these contaminants (5.6 ppb for chlordane and 21 ppb for DDTs). The average PCBs concentration was 34.4 ppb, higher than the 3.6 ppb FCG for PCBs. The average dieldrin concentrations (1.06 ppb) are higher than the 0.45 ppb FCG for dieldrin.

Table G-23. Compiled Fish Tissue Analytical Data for Peck Road Park Lake

Program	Pollutant	Sample Date	Common Name	Concentration (ppb, wet wt)	Mean Length (mm)	Mean Weight (g)
TSMP	Aroclor PCBs	7/21/1986	Largemouth Bass	ND	332	788
TSMP	Aroclor PCBs	7/21/1986	Largemouth Bass	ND	175	90
TSMP	Aroclor PCBs	4/17/1991	Largemouth Bass	ND	126	29.6
TSMP	Aroclor PCBs	4/27/1992	Largemouth Bass	ND	160	68.5
SWAMP	Total PCBs	Summer 2007	Largemouth Bass	55.307	361.4	526.2
SWAMP	Total PCBs	Summer 2007	Largemouth Bass	22.651	360.4	499.2
SWAMP	Total PCBs	4/19/2010	Largemouth Bass	25.345	359.6	846
TSMP	Chlordane	7/21/1986	Largemouth Bass	42	332	788
TSMP	Chlordane	7/21/1986	Largemouth Bass	7	175	90
TSMP	Chlordane	4/17/1991	Largemouth Bass	14.1	126	29.6
TSMP	Chlordane	4/27/1992	Largemouth Bass	ND	160	68.5
SWAMP	Chlordane	Summer 2007	Largemouth Bass	19.212	361.4	526.2
SWAMP	Chlordane	Summer 2007	Largemouth Bass	8.637	360.4	499.2
SWAMP	Chlordane	4/19/2010	Largemouth Bass	12.465	359.6	846
TSMP	DDTs	7/21/1986	Largemouth Bass	35	332	788
TSMP	DDTs	7/21/1986	Largemouth Bass	18	175	90
TSMP	DDTs	4/17/1991	Largemouth Bass	39	126	29.6
TSMP	DDTs	4/27/1992	Largemouth Bass	14	160	68.5
SWAMP	DDTs	Summer 2007	Largemouth Bass	24.416	361.4	526.2
SWAMP	DDTs	Summer 2007	Largemouth Bass	8.982	360.4	499.2
SWAMP	DDTs	4/19/2010	Largemouth Bass	13.109	359.6	846
TSMP	Dieldrin	4/17/1991	Largemouth Bass	N/A	126	29.6
TSMP	Dieldrin	4/27/1992	Largemouth Bass	N/A	160	68.5
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.965	361.4	526.2
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.542	360.4	499.2
SWAMP	Dieldrin	4/19/2010	Largemouth Bass	1.66	359.6	846

ND = Non-detect
N/A = Not applicable

G.4.3.4 Sediment Data Observed in Peck Road Park Lake

Sediment samples for Peck Road Park Lake were collected by USEPA and the county of Los Angeles on November 16, 2009, and in the summer and fall of 2008 by UCLA. UCLA collected sediment samples at PRPL-6S, PRPL-7S, PRPL-9 and PRPL-10 in the summer 2008. Each sample also had laboratory duplicates and PRPL-7S had a field duplicate. At PRPL-6S (laboratory duplicate), DDE was detected at 20 µg/kg dry weight and PCB-180 was detected at 11 µg/kg dry weight. PCB-18 was detected at PRPL-7S (laboratory duplicate of the field duplicate sample) with a sediment concentration of 17 µg/kg dry weight. Chlordane-gamma was detected in PRPL-9S (laboratory duplicate) sediment samples at 7 µg/kg dry weight. The chlordane-gamma level at PRPL-9 was the only detected contaminant above the CBSQG for TEC and PEC levels. No contaminants were above reporting levels at PRPL-10S. The results of the sampling are shown in Table G-24.

The results of the UCLA fall 2008 sediment analysis are shown in Table G-25. Sediments from PRPL-7S, PRPL-9 and PRPL-10 were collected. Each sample also had laboratory duplicates and PRPL-9 had a field duplicate. PCB-31 at PRPL-7S and PCB-66 at PRPL-9 were the only pollutants detected above reportable levels. At PRPL-9 PCB-66 was detected at 8.60 µg/kg dry weight. At PRPL-7S, PCB-31 was quantified at 276.41 µg/kg dry weight. No contaminants were above reporting levels at PRPL-10.

Chlordane-gamma was detected at all four stations on November 16, 2009, in concentrations ranging from 1.0 to 6.6 µg/kg. Chlordane-alpha was detected at PRPL-9, PRPL-10 and PRPL-13 with concentrations ranging from 3.4 to 6.5 µg/kg. The DDT compound was not detected at any of the sites, but DDT-associated degradation products (DDD and DDE) were detected at three of the four stations (PRPL-9, PRPL-10 and PRPL-13). Several PCB congeners were also detected; however, dieldrin was not detected in any of the sediment samples. The raw data for these samples are reported in Table G-26. The detection limit for all samples was 1 µg/kg dry sediment.

Table G-24. Results from Sediment Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant	PRPL-6S			PRPL-6S (lab dup)			PRPL-7S			PRPL-7S (lab dup)			PRPL-7SB (field dup)			PRPL-7SB (lab dup of field dup)			PRPL-9			PRPL-9 (lab dup)			PRPL-10			PRPL-10 (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																													
Chlordane-gamma	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	7.14	0.72	7.20	ND	0.65	6.48	ND
Chlordane-alpha	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
4,4'-DDE	0.76	7.62	ND	0.72	7.20	20.07	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
4,4'-DDD	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
4,4'-DDT	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	0.90*	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
Dieldrin	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
PCB 5	0.38	3.81	0.40*	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 18	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	17.09	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 31	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 52	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 44	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 66	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	4.19*	0.65	6.48	ND
PCB 101	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 87	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 151	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 110	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	0.24*	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 153	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 141	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	0.85*	0.72	7.20	ND	0.65	6.48	ND
PCB 138	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	0.87	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 187	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 183	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 180	0.38	3.81	ND	0.36	3.60	11.38	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 170	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 206	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND

*Results were above the detection level, but below the reporting level.

Table G-25. Results from Sediment Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-7S			PRPL-7S (lab dup)			PRPL-9			PRPL-9 (lab dup)			PRPL-9 (field dup)			PRPL-9 (lab dup of field dup)			PRPL-10			PRPL-10 (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																							
Chlordane-gamma	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
Chlordane-alpha	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
4,4'-DDE	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	ND	0.97	9.69	ND	0.98	9.82	ND
4,4'-DDD	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	4.26*	0.97	9.69	ND	0.98	9.82	ND
4,4'-DDT	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	3.81*	0.97	9.69	ND	0.98	9.82	ND
Dieldrin	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	ND	0.97	9.69	ND	0.98	9.82	ND
PCB 5	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 18	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 31	0.59	5.91	ND	0.59	5.91	276.41	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 52	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 44	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 66	0.59	5.91	2.38*	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	8.60	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 101	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 87	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 151	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 110	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 153	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 141	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	3.55*	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 138	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 187	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 183	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 180	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	1.83*	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 170	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 206	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND

*Results were above the detection level but below the reporting level.

Table G-26. Results from Sediment Samples Collected at Peck Road Park Lake on November 16, 2009

Contaminant (µg/kg dry weight)	PRPL-9			PRPL-10	PRPL-12	PRPL-13	MDL
	Results	Field Dup	Field and Lab Dup				
Chlordane-gamma	5.3	6.2	6.6	5.6	1	3.1	1
Chlordane-alpha	5.4	6.4	6.5	5.6	ND	3.4	1
cis-Nonachlor	2.5	2.7	3.4	3	ND	1.5	1
trans-Nonachlor	6.3	6.3	6.4	4.1	ND	3.2	1
Oxychlordane	ND	ND	ND	ND	ND	ND	1
2,4` - DDD	ND	ND	ND	ND	ND	ND	1
2,4` - DDE	ND	ND	ND	ND	ND	ND	1
2,4` - DDT	ND	ND	ND	ND	ND	ND	1
4,4` - DDD	3	ND	ND	4.1	ND	2.8	1
4,4` - DDE	7.3	9.7	8.4	7.7	ND	8.2	1
4,4` - DDT	ND	ND	ND	ND	ND	ND	1
Dieldrin	ND	ND	ND	ND	ND	ND	1
PCB037	ND	2.1	ND	ND	ND	ND	1
PCB074	2.9	2.3	2	ND	ND	ND	1
PCB095	1.2	1.7	1.4	1.6	ND	ND	1
PCB099	ND	ND	ND	1.0	ND	ND	1
PCB101	1.4	2.2	1.1	1.4	ND	1.0	1
PCB110	1.8	ND	1.1	ND	ND	1.2	1
PCB118	ND	1.6	1.4	ND	ND	ND	1
PCB138	5.1	3.1	ND	ND	ND	ND	1
PCB149	1.3	2	2.2	1.6	ND	1.3	1
PCB151	ND	1	1	ND	ND	ND	1
PCB153	2.1	ND	ND	1.8	ND	1.6	1
PCB174	1.8	2	2.5	1.1	ND	ND	1
PCB177	ND	1.4	1	ND	ND	ND	1
PCB180	1.1	1.6	2.5	1.8	ND	ND	1
PCB187	1.8	2.4	1.6	1.1	ND	ND	1
PCB194	ND	ND	ND	ND	1.0	ND	1
PCB206	ND	2.3	ND	1.3	ND	ND	1

G.4.3.5 Suspended Sediment Data Observed in Peck Road Park Lake

Suspended solids (TSS) from Peck Road Park Lake were collected in the summer and fall of 2008. Summer samples were taken at PRPL-6S, PRPL-6W, PRPL-7S, PRPL-9 and PRPL-10. PRPL-6W was the only sample that had enough suspended matter to perform the analysis. None of the pesticides were detected in the sample. PCB-110 was detected, but not quantifiable. The results of the summer sampling are shown in Table G-27.

Table G-27. Results from Suspended Sediment Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant	PRPL-6W		
	DL	RL	Result
	µg/kg dry suspended solids		
Chlordane-gamma	5.14	51.35	ND
Chlordane-alpha	5.14	51.35	ND
4,4'-DDE	10.27	102.71	ND
Dieldrin	10.27	102.71	ND
4,4'-DDD	10.27	102.71	ND
4,4'-DDT	10.27	102.71	ND
PCB 5	5.14	51.35	ND
PCB 18	5.14	51.35	ND
PCB 31	5.14	51.35	ND
PCB 52	5.14	51.35	ND
PCB 44	5.14	51.35	ND
PCB 66	5.14	51.35	ND
PCB 101	5.14	51.35	ND
PCB 87	5.14	51.35	ND
PCB 151	5.14	51.35	ND
PCB 110	5.14	51.35	27.15*
PCB 153	5.14	51.35	ND
PCB 141	5.14	51.35	ND
PCB 138	5.14	51.35	ND
PCB 187	5.14	51.35	ND
PCB 183	5.14	51.35	ND
PCB 180	5.14	51.35	ND
PCB 170	5.14	51.35	ND
PCB 206	5.14	51.35	ND

*Result was above detection limit, but below reporting limits.
Note: Samples were collected at PRPL-7S, PRPL-9 and PRPL-10, but had insufficient sample for analysis.

In fall 2008, TSS from PRPL-6W and PRPL-7W were analyzed for the contaminants. The only chemicals detected were PCB-138 at PRPL-6W and PCB-180 at PRPL-7S, both below reportable limits. These results are shown in Table G-28.

Table G-28. Results from Suspended Sediment Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-6W			PRPL-7W			PRPL-7W (field duplicate)		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
	µg/kg dry suspended solids								
Chlordane-gamma	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
Chlordane-alpha	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
4,4'-DDE	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
Dieldrin	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
4,4'-DDD	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
4,4'-DDT	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
PCB 5	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 18	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 31	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 52	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 44	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 66	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 101	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 87	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 151	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 110	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 153	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 141	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 138	2.36	23.63	3.56*	20.41	204.08	ND	14.42	144.23	ND
PCB 187	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 183	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 180	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	48.23*
PCB 170	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 206	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND

*Results are above the detection limits but below the reporting limits.

G.5 Monitoring Data for Lincoln Park Lake

Monitoring data relevant to the impairments of Lincoln Park Lake are available from 1992, 1993, 2008, and 2009. Figure G-12 shows the historical and recent monitoring locations for Lincoln Park Lake.

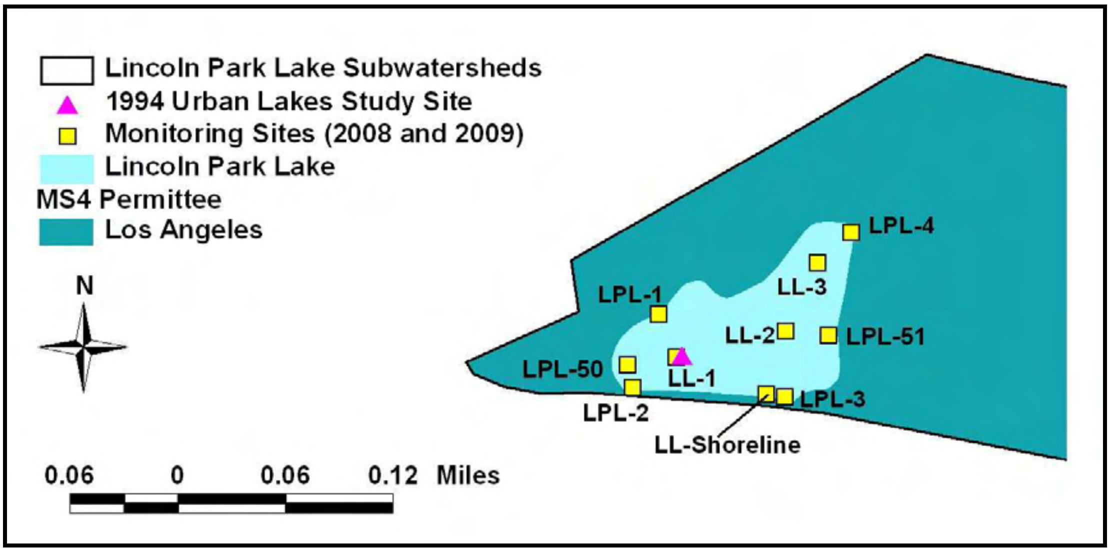

Figure G-12. Lincoln Park Lake Monitoring Sites

G.5.1 Monitoring Related to Nutrient Impairments

Water quality sampling was conducted in Lincoln Park Lake in 1992 and 1993 for the Urban Lakes Study (Table G-29) from a station located in the western half of the lake (UC Riverside, 1994) (pink triangle, Figure G-12). Sampling occurred over 2 meters of depth on 12 sampling days. TKN ranged from 0.3 mg/L to 2.8 mg/L; eight of 28 samples for ammonia were less than detection and the maximum observed ammonia concentration was 1.1 mg/L. All nitrite samples were less than the reporting limit, and 17 of 28 nitrate samples were less than the reporting limit. The maximum nitrate concentration was 0.3 mg/L. Orthophosphate concentrations in 1992 were less than or equivalent to the reporting limit, while concentrations in 1993 ranged from 0.2 mg/L to 0.3 mg/L. Total phosphorus was also higher in 1993 with concentrations ranging from 0.2 mg/L to 0.5 mg/L compared to concentrations in 1992 of which nine samples were less than the reporting limit and the maximum observed concentration was 0.2 mg/L. pH measurements ranged from 7.7 to 9.1. TOC ranged from 6.0 mg/L to 14.5 mg/L, with one outlier of 132 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from <1 μg/L to 97 μg/L with an average of 33 μg/L.

Table G-29. Lincoln Park Lake 1992/1993 Monitoring Data for Nutrients

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/13/1992	0	1.4	0.4	<0.01	<0.01	0.1	<0.01	7.8	9	672
	2	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.8	9.1	653
7/13/1992	0	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.9	9.4	671
	1.5	1.4	0.3	<0.01	<0.01	<0.01	<0.01	7.9	9.1	668
7/13/1992	0	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.9	8.9	667
	1.5	1.4	0.3	<0.01	<0.01	<0.01	<0.01	8.1	8.6	649
8/19/1992	0	2.8	0.3	<0.01	0.1	<0.01	0.1	8.4	8.5	701
	2	2.2	0.3	<0.01	<0.01	<0.01	0.1	8.5	8.6	697
9/17/1992	0	1.9	0.2	<0.01	0.1	<0.01	0.2	8.2	8.4	631
	2	1.6	0.1	<0.01	<0.01	<0.01	0.1	8.4	7.9	629
10/15/1992	0	1.4	0.3	<0.01	<0.01	<0.01	<0.01	8.2	6.6	645
	2	1.1	0.4	<0.01	0.1	<0.01	<0.01	8.2	6.6	638
11/5/1992	0	1.9	0.7	<0.01	0.2	<0.01	0.1	8.1	6.3	602
	1.7	1.7	0.8	<0.01	0.2	<0.01	0.1	8.2	7.1	581
12/8/1992	0	1.7	0.5	<0.01	0.3	<0.01	<0.01	7.7	6	575
	1.5	1.9	0.5	<0.01	0.3	<0.01	0.1	7.7	6.1	568
1/14/1993	0	2.7	0.6	<0.01	0.3	0.2	0.4	7.8	7	419
	2	2.3	0.7	<0.01	0.3	0.2	0.3	8	6.5	446
2/2/1993	0	2.4	1.1	<0.01	0.3	0.2	0.2	8.1	6.2	539
	2	2.1	1.1	<0.01	0.3	0.2	0.2	8.1	6.1	598
3/24/1993	0	1.9	<0.01	<0.01	<0.01	0.3	0.5	8.8	9.5	634
	2	1.8	<0.01	<0.01	<0.01	0.3	0.4	8.8	9.1	617
4/6/1993	0	1.6	<0.01	<0.01	<0.01	0.3	0.4	8.9	7.9	594
	1.5	1.6	<0.01	<0.01	<0.01	0.3	0.5	8.9	8.6	604
5/3/1993	0	0.3	<0.01	<0.01	<0.01	0.3	0.4	9.1	11.1	640
	2	0.7	<0.01	<0.01	<0.01	0.3	0.4	9.0	11.2	650
6/7/1993	0	1.6	<0.01	<0.01	<0.01	0.3	0.2	8.8	132	674
	2	1.9	<0.01	<0.01	<0.01	0.3	0.3	8.7	14.5	674

There are no stations in Lincoln Park Lake or its drainage area listed in the Regional Board Water Quality Assessment Database. The Water Quality Assessment Report, however, states that DO was partially supporting the aquatic life use with 78 measurements of dissolved oxygen ranging from 0.1 mg/L to

13.7 mg/L. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonium samples were collected ranging from non-detect to 1.14 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples.

The Regional Board sampled water quality at four stations around the shoreline of Lincoln Park Lake in 2008. All samples were collected from the edge of the lake using a 6-ft extension pole. Samples were collected approximately 4 inches below the water surface.

During the October 29, 2008 sampling event, concentrations of total phosphate and ammonia at each station were less than the reporting limits of 0.5 mg/L and 0.1 mg/L, respectively. TKN at each site ranged from 1.49 mg/L to 2.32 mg/L. Total dissolved solids ranged from 847 mg/L to 868 mg/L. Suspended solids ranged from less than the reporting limit of 10 mg/L to 12 mg/L. Chlorophyll *a* ranged from 44 µg/L to 123 µg/L.

During the November 6, 2008 sampling event, concentrations of orthophosphate, nitrate, and nitrite at each site were less than the reporting limits of 0.4 mg/L, 0.1 mg/L, and 0.1 mg/L, respectively. No other parameters were measured during this event.

Field data for these two sampling events are summarized in Table G-30.

Table G-30. Field Data for 2008 Monitoring Events at Lincoln Park Lake

Site	Date	Time	Temperature	pH	Total Depth (m)
LPL-1	10/29/2008	15:15	22	8.9	0.6
	11/6/2008	9:34	17	8.5	0.6
LPL-2	10/29/2008	14:05	20	8.9	0.9
	11/6/2008	10:05	17	8.5	0.5
LPL-3	10/29/2008	15:45	20	9.0	0.4
	11/6/2008	10:30	17	8.7	0.3
LPL-4	10/29/2008	16:50	22	9.0	0.5
	11/6/2008	10:45	17	8.5	0.4

In 2009, the City of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples at three locations in Lincoln Park Lake. Table G-31 summarizes the analyses for samples collected on February 18 through July 28, 2009. The nitrate in the lake at all locations and sampling times was below the detection level. After February, all nitrite levels were also below detection level; and after March, all ammonia samples were also below detection. The fraction of organic nitrogen was between 0.8 and 1.8 throughout the sampling period. The chlorophyll *a* was lowest in February; at LL-3 it was 13 µg/L. The maximum amount of chlorophyll (47 µg/L) was sampled in July at LL-2. Suspended solids were also higher in the summer months. In July, the average TSS was 18.2 mg/L and only 11.2 mg/L in February.

Table G-31. 2009 City of Los Angeles Bureau of Sanitation 2009 Lincoln Park Lake Monitoring Data

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TP (mg/L)	Chlorophyll *a* (µg/L)	BOD (mg/L)	TSS (mg/L)
2/18/2009	LL-1	0.24	1.3	<0.02	0.11	0.14	35	6	10.5
	LL-2	0.24	1.0	<0.02	0.11	0.09	44	5	12.0
	LL-3	0.27	1.7	<0.02	0.13	0.09	13	4	11.0
3/26/2009	LL-1	0.09	1.3	<0.02	<0.02	0.13	26	4	14.0
	LL-2	0.08	1.3	<0.02	<0.02	0.12	32	5	13.3
	LL-3	0.10	1.8	<0.02	<0.02	0.11	22	5	10.0
4/27/2009	LL-1	<0.05	1.5	<0.02	<0.02	0.18	24	6	12.0
	LL-2	<0.05	1.0	<0.02	<0.02	0.17	28	3	12.0
	LL-3	<0.05	1.3	<0.02	<0.02	0.18	23	4	13.2
5/28/2009	LL-1	<0.05	1.0	<0.02	<0.02	0.13	26	<3	13.0
	LL-2	<0.05	1.2	<0.02	<0.02	0.16	21	<3	16.0
	LL-3	<0.05	1.2	<0.02	<0.02	0.16	25	<3	8.0
7/28/2009	LL-1	<0.05	0.9	<0.02	<0.02	0.15	40	4	16.0
	LL-2	<0.05	1.4	<0.02	<0.02	0.16	47	4	19.5
	LL-3	<0.05	0.8	<0.02	<0.02	0.14	44	4	19.0

Sonde data were also collected by the City of Los Angeles Bureau of Sanitation. Table G-32 presents the mean daily values measured at stations LL-1, LL-2, and LL-3 for temperature, specific conductivity, dissolved oxygen, and pH at three depths. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification.

Table G-32. Mean Values of Sonde Data Collected in Lincoln Park Lake at Stations 1, 2, and 3

	Temperature (°C)			Specific Conductivity (mS/cm)			Dissolved Oxygen (mg/L)			pH		
Date	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m
7/28/2008	27.30	27.47	27.48	1.20	1.19	1.19	8.35	7.69	8.05	8.65	8.67	8.71
2/18/2009	12.77	12.35	11.96	1.04	1.04	1.04	8.79	8.74	8.42	8.24	8.22	8.17
3/26/2009	17.90	17.74	N/A	1.09	1.09	N/A	8.61	8.48	N/A	8.34	8.31	N/A
4/27/2009	20.54	20.55	20.76	1.14	1.14	1.14	7.42	7.05	6.49	8.36	8.34	8.31
5/28/2009	23.67	23.77	23.76	1.21	1.21	1.21	7.94	7.74	7.75	8.43	8.45	8.46

N/A = no data available

On March 10, 2009, the Regional Board and USEPA sampled water quality in Lincoln Park Lake. Two sites were accessed by wading in from boat access ramps located on either side of the lake. Samples were collected from 1 foot at each site and the total depth at each site was approximately 2.2 feet. Table G-33 summarizes the nutrient and chlorophyll *a* measurements for these two stations. Ammonia concentrations were relatively high and ranged from 1.2 mg/L to 1.26 mg/L; TKN was 2.2 mg/L at both stations. Nitrate and nitrite were both relatively low with concentrations averaging 0.07 mg/L and 0.04 mg/L, respectively. Orthophosphate concentrations were approximately 0.08 mg/L and total phosphorus concentrations were approximately 0.126 mg/L. Chlorophyll *a* concentrations at both sites were less than the detection limit of 1 µg/L.

Table G-33. In-lake and Shoreline Water Column Measurements for Lincoln Park Lake

Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	TP (mg/L)	TDS (mg/L)	TSS (mg/L)	Chloro-phyll *a* (µg/L)
LPL-50	1.20	2.2	0.07	0.04	0.0762	0.125	703	4	<1 µg/L
LPL-50 (duplicate)	1.24	NA	0.07	0.04	0.0835	0.125	NA	NA	<1 µg/L
LPL-51	1.26	2.2	0.06	0.04	0.0802	0.127	664	5.2	<1 µg/L

Profile data collected in Lincoln Park Lake on March 10, 2009 are summarized in Table G-34. DO concentrations in the lake generally ranged from 5.9 mg/L to 6.2 mg/L with one reading of 7.0 mg/L from a surface sample. pH ranged from 6.7 to 7.0. Profile depths listed in the field notes (ranging from surface to 1.3 meters) were multiplied by the ratio of total depth reported in the field notes to the depth measured on the probe cable at each monitoring station because the probe was drifting and indicating depths greater than actual (Anna Sofranko, USEPA Region IX, personal communication, May 12, 2009).

Table G-34. Field Data for the March 10, 2009 Lincoln Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
LPL-50	13:45	Surface	18.9	7.0	7.6	Greater than total depth	0.69
		0.26	18.8	6.8	6.0		
		0.53	18.6	6.8	6.2		
		~0.69 (bottom)	17.2	6.7	6.0		
LPL-51	15:10	Surface	19.1	6.7	6.2	Greater than total depth	0.66
		0.26	19.2	6.7	6.1		
		0.53	19.2	6.7	6.2		
		~0.66 (bottom)	19.2	6.7	5.9		

On August 4, 2009, USEPA and the Regional Board collected additional nutrient samples from Lincoln Park Lake. Ammonia, TKN, nitrite, and nitrate were all less than the detection limits of 0.03 mg-N/L, 0.456 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively. Orthophosphate was less than the detection limit (0.0075 mg-P/L), and total phosphorus was 0.182 mg-P/L. The chlorophyll *a* concentration was 27.3 μg/L. The potable water input was also sampled during this event. Ammonia and nitrate were both 0.33 mg-N/L; nitrite was 0.03 mg-N/L. TKN measured 0.531 mg-N/L. Orthophosphate and total phosphorus were 0.017 mg-P/L and 0.118 mg-P/L, respectively.

Profile data associated with this event were collected at LL-1, shown in Table G-35. The DO concentration ranged from 8.32 to 10.19 mg/L. The total depth at this station was 1.7 meters, and the Secchi depth was 0.66 meters. The pH was approximately 9.1 at all depths.

Table G-35. Field Data for the August 4, 2009 Lincoln Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
LL-1	14:40	0.1	28.7	9.1	9.14	0.66	1.70
		0.52	28.4	9.1	9.51		
		1.01	26.7	9.2	10.19		
		1.5	26.1	9.0	8.32		

Field data were collected for the potable water source during the August sampling event. After purging the line for approximately ten minutes, the pH was 7.82, the DO was 6.62 mg/L, and the temperature was 26.8 ºC.

Additional supplemental water quality samples were collected from Lincoln Park Lake. Table G-36 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data measured in the lake. Temperature and pH measurements reported in the field notes are also shown in this table. Both temperature and pH significantly increased between March and August. The average temperature in March was 19.0 °C and the temperature in August was 28.7 °C. The pH ranged from 6.7 in the winter to 9.1 in the summer. Chloride, sulfate, TDS, DOC, and TOC all significantly increased in August. The alkalinity in the summer was 61 mg/L lower than the level measured in March.

Table G-36. Supplemental Water Quality Monitoring for Lincoln Park Lake

Date	Location	Time	Temperature (℃)	pH	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
3/10/2009	LPL-50	13:40	18.9	7.0	97	247	142	262	703	5.6	6.1
	LPL-51	14:30	19.1	6.7	99	250	142	257	664	5.4	5.3
8/4/2009	LL-1[1]	14:05	28.7	9.1	134	305	81	281	826	9.8	10.5

[1] These data were averages of laboratory replicates, except for temperature and pH data (which were surface samples collected at 14:40).

The city of Los Angeles provided water quality monitoring data for the Glendale Water Reclamation Plant, which may be used to supplement lake levels and irrigate parkland at Lincoln Park in the future.

Table G-37 summarizes the average water quality for this source based on monthly averages reported for 2008 and 2009.

Table G-37. Average Water Quality for the Glendale Water Reclamation Plant

NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Orthophosphate (mg/L)	TP (mg/L)
1.3	3.17	5.64	0.009	1.76	1.93

G.5.2 MONITORING RELATED TO LEAD IMPAIRMENT

In 1996, Lincoln Park Lake was deemed impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Lincoln Park Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Lincoln Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-38. Specifically, samples were collected from a station located in the western half of the lake (UC Riverside, 1994) (pink triangle, Figure G-12) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to two meters. The range of the 28 dissolved copper samples was between less than 10 µg/L and 81 µg/L. Similarly, dissolved lead samples were also collected throughout the water column at depths from the surface to two meters. The 28 samples collected ranged in concentration from less than 1 µg/L to 94 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Lincoln Park Lake states that lead was not supporting the assessed uses: 28 measurements reported a maximum lead concentration of 94 µg/L, a maximum copper concentration of 61 µg/L, a maximum cadmium concentration of 1.6 µg/L, and a maximum zinc concentration of 13 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-38. Lincoln Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/13/1992	0	16	<1
	2	<10	<1
7/13/1992	0	13	<1
	1.5	<10	N/A
7/13/1992	0	20	21
	1.5	11	<1
8/19/1992	0	22	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	2	23	<1
9/17/1992	0	17	N/A
	2	16	1
10/15/1992	0	<10	<1
	2	<10	<1
11/5/1992	0	<10	2
	1.7	<10	2
12/8/1992	0	<10	7
	1.5	<10	7
1/14/1993	0	11	9
	2	<10	1
2/2/1993	0	<10	<1
	2	<10	<1
3/24/1993	0	<10	<1
	2	<10	<1
4/6/1993	0	81	2
	1.5	47	<1
5/3/1993	0	<10	<1
	2	<10	3
6/7/1993	0	17	94
	2	16	33

Table G-39 presents 40 additional metal samples that were collected by the USEPA, Regional Board, and/or the city of Los Angeles between October 2008 and December 2010 at Lincoln Park Lake. Samples were collected at locations LPL 1/2/3/4, LPL 50/51, LL-1, LL-2, LL-3, LPL-2/4, and LL-Shoreline. Sites were analyzed for dissolved cadmium, copper, lead, and/or zinc (only lead data are reported for the city of Los Angeles samples).

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.4 µg/L; copper concentrations were between 2.1 µg/L and 8.12 µg/L; lead concentrations ranged from <0.05 µg/L to 2.0 µg/L; and zinc concentrations were 0.3 µg/L to 1.3 µg/L. Metal toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 sampling resulted in a hardness range of 166 mg/L to 356 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-39 were weighted more heavily in the assessment.

Table G-39. Metals Data for the 2008-2010 Lincoln Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	10/29/2008	LPL 1/2/3/4	247.8	0.4	4.4	<0.1	0.5	average of replicates; average of sites 1-4
RB	3/10/2009	LPL 50/51	258.9	<0.2	2.1	0.2	1.3	average of replicates & duplicates; average of sites 50 and 51
City LA	2/18/2009	LL-1/2/3	292.0	N/A	N/A	<2	N/A	average of sites 1, 2, and 3
City LA	3/26/2009	LL-1/2/3	257.0	N/A	N/A	0.1	N/A	average of sites 1, 2, and 3
City LA	4/27/2009	LL-1/2/3	311.3	N/A	N/A	0.2	N/A	average of sites 1, 2, and 3
City LA	5/28/2009	LL-1/2/3	316.7	N/A	N/A	0.1	N/A	average of sites 1, 2, and 3
City LA	7/28/2009	LL-1/2/3	279.3	N/A	N/A	0.4	N/A	average of sites 1, 2, and 3
RB/EPA	8/4/2009	LL 1	281.0	<0.2	4	0.3	0.8	average of replicates
RB/EPA	8/4/2009	LPL 2 / 4	282.3	<0.2	2.1	0.1	0.3	average of shore sites 2 and 6
City LA	8/28/2009	LL-Shoreline	324	N/A	N/A	0.1	N/A	
City LA	9/4/2009	LL-Shoreline	312	N/A	N/A	<0.1	N/A	
City LA	9/11/2009	LL-Shoreline	328	N/A	N/A	<0.1	N/A	
City LA	9/18/2009	LL-Shoreline	320	N/A	N/A	0.2	N/A	
City LA	9/25/2009	LL-Shoreline	331	N/A	N/A	0.2	N/A	
City LA	10/2/2009	LL-Shoreline	315	N/A	N/A	<0.1	N/A	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	10/9/2009	LL-Shoreline	316	N/A	N/A	0.2	N/A	
City LA	10/16/2009	LL-Shoreline	356	N/A	N/A	0.2	N/A	
City LA	10/23/2009	LL-Shoreline	331	N/A	N/A	0.3	N/A	
City LA	10/30/2009	LL-Shoreline	332	N/A	N/A	0.3	N/A	
City LA	11/6/2009	LL-Shoreline	330	N/A	N/A	0.3	N/A	
City LA	11/13/2009	LL-Shoreline	349	N/A	N/A	0.2	N/A	
City LA	11/20/2009	LL-Shoreline	307	N/A	N/A	0.4	N/A	
City LA	12/4/2009	LL-Shoreline	323	N/A	N/A	0.3	N/A	
City LA	12/11/2009	LL-Shoreline	321	N/A	N/A	0.4	N/A	
City LA	12/18/2009	LL-Shoreline	318	N/A	N/A	0.2	N/A	
City LA	1/8/2010	LL-Shoreline	333	N/A	N/A	0.5	N/A	
City LA	1/15/2010	LL-Shoreline	315	N/A	N/A	1.6	N/A	
City LA	1/22/2010	LL-Shoreline	271	N/A	N/A	0.4	N/A	
City LA	2/5/2010	LL-Shoreline	286	N/A	N/A	0.3	N/A	
City LA	2/12/2010	LL-Shoreline	265	N/A	N/A	0.2	N/A	
City LA	2/19/2010	LL-Shoreline	236	N/A	N/A	<0.1	N/A	
City LA	2/26/2010	LL-Shoreline	260	N/A	N/A	<0.1	N/A	
EPA / RB	9/28/2010	LL-1	166	<0.4	8.12	<0.1	<0.5	
EPA / RB	9/28/2010	LPL-4	167	<0.2	3.73	<0.05	<0.1	
City LA	10/8/2010	LL-Shoreline	256	N/A	N/A	0.17	N/A	
City LA	10/22/2010	LL-Shoreline	269	N/A	N/A	0.17	N/A	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	11/5/2010	LL-Shoreline	253	N/A	N/A	0.19	N/A	
City LA	11/19/2010	LL-Shoreline	266	N/A	N/A	0.19	N/A	
City LA	12/3/2010	LL-Shoreline	253	N/A	N/A	0.34	N/A	
City LA	12/17/2010	LL-Shoreline	238	N/A	N/A	0.16	N/A	

N/A = No data available
RB = Regional Board
EPA = USEPA
City LA = City of Los Angeles

USEPA also collected one sediment sample in September 2010 to further evaluate lake conditions. Table G-40 summarizes the lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-40. Sediment Metals Data for the September 2010 Lincoln Park Lake Sampling Event

Organi zation	Date	Station ID	Lead (mg/kg)	Notes
EPA	09/28/2010	LL1	105	Average of duplicates

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G.6 Monitoring Data for Echo Park Lake

Echo Park Lake has been monitored more frequently than many other lakes addressed in this memo. Sampling has occurred in 1992, 1993, and 2003 through 2009. In addition, fish tissue data are available for 1987 to 2007. Figure G-13 shows the location of historic and recent monitoring locations in Echo Park Lake.

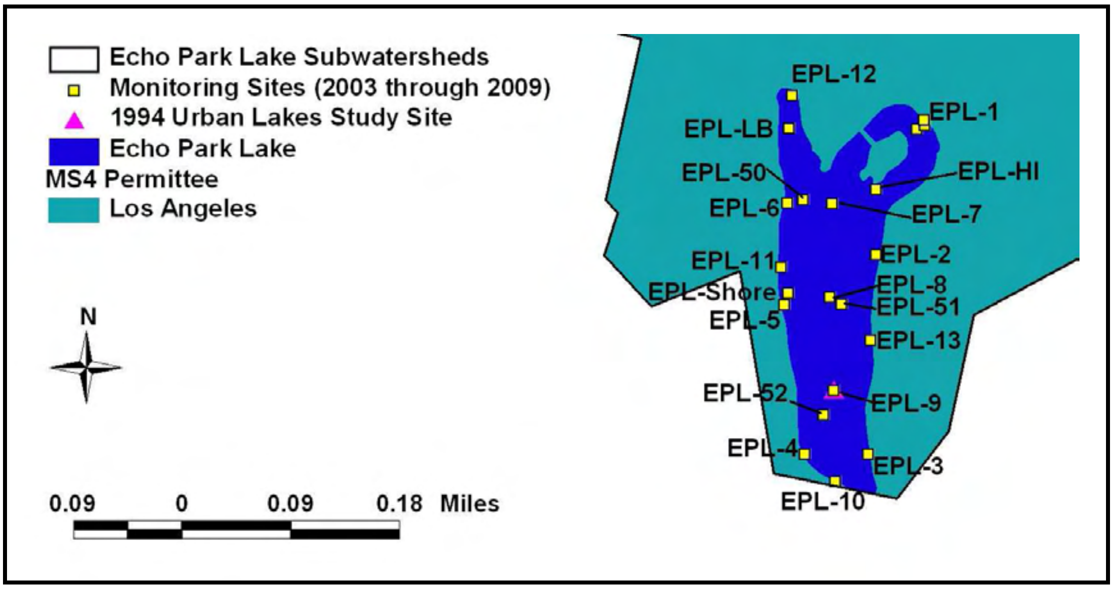

Figure G-13. Historic and Recent Sampling Sites at Echo Park Lake

G.6.1 Monitoring Related to Nutrient Impairments

Results of the 1992/1993 Urban Lakes Study sampling are summarized in Table G-41. Sampling occurred near the center of the lower half of the lake (UC Riverside, 1994) (pink triangle, Figure G-13). TKN concentrations during this sampling period ranged from 0.9 mg/L to 1.9 mg/L. Ammonium concentrations were less than the reporting limit for 22 of 31 samples, and the maximum observed ammonium concentration was 0.7 mg/L. Nitrite concentrations were less than the reporting limit in all samples; 24 of 31 nitrate samples were less than the reporting limit. The maximum observed nitrate concentration was 0.2 mg/L. Orthophosphate concentrations were generally less than or equivalent to the reporting limit with some observations of 0.2 mg/L. Total phosphorus concentrations ranged from less than the reporting limit to 0.3 mg/L. pH measurements ranged from 7.7 to 9.4, and TOC ranged from 4.8 mg/L to 7.6 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 6 µg/L to 66 µg/L with an average of 24 µg/L.

Table G-41. Echo Park Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/14/1992	0	1	0.3	<0.1	<0.1	0.1	<0.1	8.8	6.9	333
	1.5	1	<0.1	<0.1	<0.1	0.1	<0.1	8.9	7.1	321
7/14/1992	0	1.1	<0.1	<0.1	<0.1	0.1	<0.1	9.0	7.4	319
	1.5	1.2	<0.1	<0.1	<0.1	0.1	<0.1	9.0	7.4	317
7/14/1992	0	1.4	0.1	<0.1	<0.1	0.1	0.1	9.0	7.6	316
	1.3	1.4	<0.1	<0.1	<0.1	0.1	0.1	8.9	6.8	322
8/13/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	<0.1	9.3	6.5	322
	1.5	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.7	323
8/13/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.4	322
	2	1.2	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.6	319
	0	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.1	6.5	334
9/17/1992	0	1.1	<0.1	<0.1	<0.1	<0.1	0.2	8.3	6.9	364
	1.7	1.1	<0.1	<0.1	<0.1	<0.1	0.1	8.2	7.2	359
10/15/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	<0.1	8.1	6.7	447
	1.7	1.3	0.1	<0.1	<0.1	<0.1	<0.1	8.1	6.8	450
11/5/1992	0	1.9	<0.1	<0.1	<0.1	<0.1	0.2	8.7	6.8	411
	1.5	1.9	<0.1	<0.1	<0.1	<0.1	0.2	8.7	7.2	428
12/8/1992	0	1.3	0.2	<0.1	0.2	<0.1	<0.1	7.7	6.1	443
	1.5	1.3	0.2	<0.1	0.1	<0.1	<0.1	7.8	5.9	453
1/12/1993	0	1.8	0.7	<0.1	0.2	0.2	<0.1	7.8	5.5	350
	1.5	1.7	0.7	<0.1	0.1	0.2	0.1	7.8	5.4	357
2/2/1993	0	1.7	0.5	<0.1	0.2		<0.1	8.5	4.8	323
	1.5	1.6	0.6	<0.1	0.2	<0.1	<0.1	8.5	4.8	299
3/17/1993	0	0.9	<0.1	<0.1	0.1	<0.1	<0.1	8.8	5.4	252
	1.5	1	<0.1	<0.1	<0.1	<0.1	<0.1	8.8	5.2	251
4/7/1993	0	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.4	5.4	249
	1.5	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.4	4.8	251
5/3/1993	0	1.2	<0.1	<0.1	<0.1	0.2	0.3	8.9	5.3	352
	2	1.2	<0.1	<0.1	<0.1	0.2	0.3	8.9	5	321
6/8/1993	0	1.1	<0.1	<0.1	<0.1	0.2	0.1	8.9	7.1	386
	1.5	1.1	<0.1	<0.1	<0.1	0.2	0.1	8.6	7.1	411

There were no stations in Echo Park Lake or its drainage area in the Regional Board Water Quality Assessment Database. The Water Quality Assessment Report, however, states pH was not supporting the contact recreation use and partially supporting the aquatic life use: 69 measurements of pH were collected which ranged from 7.0 to 9.4. Thirty-one ammonium samples were collected with values ranging from non-detect to 0.71 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium); ammonia was listed as not supporting the aquatic life and contact recreation uses. Raw data are not available to assess location, date, time, depth, temperature, or pH with regards to these samples. Odor and algae were both listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

In 2003, the City of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples from Echo Park Lake. Stations EPL-1 through EPL-6 are perimeter stations that were only sampled for bacterial parameters and EPL-7, EPL-8, and EPL-9 are mid-lake stations. Table G-42 lists the nutrient data collected through February 2010 for the three in-lake stations. Of the 84 samples collected during this period, 38 were non-detect for ammonia; the maximum ammonia concentration was 0.93 mg/L. Organic nitrogen concentrations ranged from 0.28 mg/L to 3.14 mg/L. Thirty-five nitrate samples were non detect, and the maximum observed concentration was 1.0 mg/L. Fifty-five of the nitrite samples were non detect; the other two samples had concentrations of 0.02 and 0.09 mg/L. Total nitrogen concentrations, calculated from the sum of ammonia, organic nitrogen, nitrate, and nitrite, ranged from 0.28 mg/L to 3.48 mg/L. Total phosphate measurements generally ranged from 0.06 mg/L to 0.51 mg/L with three measurements less than detection. Biochemical oxygen demand (BOD) ranged from 4 mg/L to 18 mg/L with 25 measurements less than the detection limit; the length and type of the BOD test was not specified in the data set received. TSS measurements ranged from 3 mg/L to 31 mg/L. No chlorophyll *a* data were reported.

Table G-42. City of Los Angeles Bureau of Sanitation Echo Park Lake Monitoring Data

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
5/14/2003	EPL-7	<0.10	0.3	<0.02	<0.02	0.28	0.12	<4	8
	EPL-8	<0.10	0.3	<0.02	<0.02	0.28	0.14	<4	11
	EPL-9	<0.10	0.3	<0.02	<0.02	0.28	0.15	4	16
8/12/2003	EPL-7	0.34	3.1	<0.02	<0.10	3.48	0.28	17	29
	EPL-8	0.56	2.0	<0.02	<0.10	2.58	0.08	18	25
	EPL-9	0.34	1.9	<0.02	<0.10	2.24	0.23	16	25
11/20/2003	EPL-7	0.30	1.0	0.16	<0.02	1.46	0.09	<4	5
	EPL-8	0.30	1.3	0.15	<0.02	1.75	0.08	<4	5
	EPL-9	0.60	1.0	0.16	<0.02	1.76	0.07	<4	4
2/18/2004	EPL-7	<0.10	< 0.1	1.00	0.09	1.09	0.08	10	6
	EPL-8	<0.10	0.6	0.08	<0.02	0.68	0.08	9	3
	EPL-9	<0.10	1.2	0.10	<0.02	1.30	0.16	9	5
5/18/2004	EPL-7	0.10	1.1	<0.02	<0.02	1.20	<0.05	<4	8
	EPL-8	0.10	1.0	<0.02	<0.02	1.10	<0.05	<4	11
	EPL-9	0.20	1.0	<0.02	<0.02	1.20	<0.05	<4	8

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
8/25/2004	EPL-7	0.10	1.2	<0.02	<0.02	1.30	0.09	5	19
	EPL-8	<0.10	1.1	<0.02	<0.02	1.10	0.07	6	14
	EPL-9	<0.10	1.0	<0.02	<0.02	1.00	0.08	6	13
11/17/2004	EPL-7	0.50	1.3	0.17	<0.02	1.97	0.48	16	16
	EPL-8	0.57	1.0	0.18	<0.02	1.75	0.40	7	7
	EPL-9	0.49	1.3	0.18	<0.02	1.97	0.51	8	8
2/17/2005	EPL-7	<0.05	1.5	0.18	<0.02	1.65	0.13	4	16
	EPL-8	<0.05	0.8	0.06	<0.02	0.85	0.12	<4	9
	EPL-9	<0.05	0.7	0.06	0.02	0.80	0.09	<4	8
5/19/2005	EPL-7	<0.05	1.0	0.07	<0.02	1.07	0.15	<4	31
	EPL-8	<0.05	0.8	0.02	<0.02	0.82	0.11	<4	18
	EPL-9	<0.05	1.0	<0.02	<0.02	1.00	0.13	<4	21
8/18/2005	EPL-7	<0.05	1.4	<0.02	<0.02	1.40	0.06	7	23
	EPL-8	<0.05	1.2	<0.02	<0.02	1.20	0.06	7	20
	EPL-9	<0.05	1.2	<0.02	<0.02	1.20	0.07	6	20
11/17/2005	EPL-7	0.21	0.7	<0.02	<0.02	0.91	0.25	4	16
	EPL-8	0.24	2.2	<0.02	<0.02	2.44	0.26	4	20
	EPL-9	0.69	1.4	<0.02	<0.02	2.09	0.26	4	16
2/9/2006	EPL-7	<0.05	1.0	0.04	<0.02	1.04	0.16	AE[1]	AE[1]
	EPL-8	<0.05	1.0	<0.02	<0.02	1.00	0.21	AE[1]	AE[1]
	EPL-9	<0.05	1.1	<0.02	<0.02	1.10	0.25	AE[1]	AE[1]
5/11/2006	EPL-7	0.45	0.3	<0.02	<0.02	0.75	0.07	<3	18
	EPL-8	0.30	0.4	<0.02	<0.02	0.70	0.18	<3	21
	EPL-9	0.35	0.3	<0.02	<0.02	0.65	0.15	<3	16
8/17/2006	EPL-7	0.09	1.2	<0.02	<0.02	1.29	0.15	8	28
	EPL-8	0.10	0.9	<0.02	<0.02	1.00	0.16	4	10
	EPL-9	0.17	1.1	<0.02	<0.02	1.27	0.19	6	16
11/16/2006	EPL-7	0.10	0.9	0.07	<0.02	1.07	0.07	<3	18
	EPL-8	0.14	1.8	0.06	<0.02	2.00	0.08	<3	13
	EPL-9	0.19	1.5	0.04	<0.02	1.73	0.08	<3	14
2/8/2007	EPL-7	0.59	0.7	0.06	<0.02	1.35	0.21	<3	8
	EPL-8	0.67	0.4	0.06	<0.02	1.13	0.21	<3	9
	EPL-9	0.93	0.4	0.06	<0.02	1.39	0.22	<3	9

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
5/17/2007	EPL-7	0.10	1.2	<0.20	<0.20	1.30	0.14	<3	24
	EPL-8	0.09	1.4	<0.20	<0.20	1.49	0.31	<3	27
	EPL-9	0.08	1.6	<0.20	<0.20	1.68	0.24	<3	22
8/16/2007	EPL-7	<0.05	0.9	<0.02	<0.02	0.90	0.11	4	19
	EPL-8	<0.05	1.0	<0.02	<0.02	1.00	0.12	5	11
	EPL-9	<0.05	0.8	0.02	<0.02	0.82	0.12	5	12
11/7/2007	EPL-7	<0.05	0.9	<0.02	<0.02	1.00	0.13	5	12
	EPL-8	0.10	0.9	<0.02	<0.02	1.00	0.14	5	12
	EPL-9	0.10	0.7	<0.02	<0.02	0.80	0.12	4	12
2/14/2008	EPL-7	0.18	0.2	0.13	< 0.02	0.51	0.11	< 3	8
	EPL-8	0.28	0.5	0.12	< 0.02	0.90	0.12	< 3	11
	EPL-9	0.27	0.3	0.12	< 0.02	0.69	0.12	< 3	10
5/8/2008	EPL-7	< 0.05	1.1	0.08	< 0.02	1.18	0.13	5	18
	EPL-8	0.09	1.1	0.08	< 0.02	1.27	0.14	6	16
	EPL-9	0.17	0.6	0.09	< 0.02	0.86	0.12	< 3	14
8/7/2008	EPL-7	< 0.05	0.8	< 0.02	< 0.02	0.80	0.11	< 3	19
	EPL-8	< 0.05	0.9	< 0.02	< 0.02	0.90	0.13	< 3	19
	EPL-9	< 0.05	1.0	< 0.02	< 0.02	1.00	0.12	3	16
11/20/2008	EPL-7	0.31	1.0	0.38	0.03	1.72	0.08	3	10
	EPL-8	0.32	1.1	0.33	< 0.02	1.75	0.10	3	14
	EPL-9	0.28	1.0	0.31	< 0.02	1.59	< 0.05	3	12
2/19/2009	EPL-7	< 0.05	0.5	0.28	0.10	0.88	0.13	< 3	4
	EPL-8	< 0.05	0.4	0.29	0.11	0.80	0.10	< 3	4
	EPL-9	0.05	0.5	0.30	0.10	0.95	0.11	< 3	6
5/21/2009	EPL-7	< 0.05	1	< 0.02	< 0.02	1.00	0.15	3	17
	EPL-8	< 0.05	0.8	< 0.02	< 0.02	0.80	0.15	4	19
	EPL-9	< 0.05	1.1	< 0.02	< 0.02	1.10	0.16	5	15
8/18/2009	EPL-7	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
12/22/2009	EPL-7	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
2/16/2010	EPL-7	0.10	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	0.11	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	0.10	n/a	n/a	n/a	n/a	n/a	n/a	n/a

[1] AE indicates analyst error: no value reported.

Sonde data were also collected by the City of Los Angeles Bureau of Sanitation. Table G-43 presents the mean daily values measured at stations EPL-7, EPL-8, and EPL-9 for temperature, specific conductivity, dissolved oxygen, and pH at four depths. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification.

Table G-43. Mean Values of Sonde Data Collected in Echo Park Lake at Stations 7, 8, and 9

Date	Temperature (℃)				Specific Conductivity (mS/cm)				Dissolved Oxygen (mg/L)				pH			
	Surface	0.5 - 1.0 m	1.0 - 1.5m	>1.5m	Surface	0.5 - 1.0 m	1.0 - 1.5m	>1.5m	Surface	0.5 - 1.0 m	1.0 - 1.5 m	>1.5m	Surface	0.5 - 1.0 m	1.0 - 1.5 m	>1.5m
8/25/2004	23.8	23.4	23.1	22.9	0.750	0.749	0.750	0.750	8.9	8.8	8.6	7.9	8.5	8.4	8.4	7.9
11/17/2004	15.9	15.7	15.5	15.3	0.656	0.653	0.654	0.656	9.1	8.9	8.9	8.8	7.9	7.9	7.9	7.9
2/17/2005	16.0	15.9	15.7	15.5	0.417	0.417	0.418	0.418	10.5	10.5	10.4	9.8	8.4	8.4	8.3	8.3
5/19/2005	22.5	22.0	21.9	21.9	0.448	0.448	0.447	0.451	9.8	9.8	9.7	9.6	8.6	8.6	8.6	8.5
8/18/2005	23.6	23.6	23.3	23.5	0.460	0.459	0.459	0.467	9.3	9.4	9.4	8.8	8.9	8.9	8.9	8.1
11/17/2005	16.2	16.0	15.9	15.8	0.537	0.537	0.538	0.538	11.9	12.0	11.9	11.3	8.6	8.6	8.5	8.3
2/9/2006	13.7	13.7	13.7	13.8	0.540	0.541	0.541	0.541	10.9	10.9	10.8	10.7	8.4	8.4	8.4	8.3
5/11/2006	20.5	20.6	20.6	20.3	0.499	0.498	0.499	0.500	10.5	10.6	10.5	9.8	8.6	8.6	8.6	8.5
8/17/2006	24.9	24.8	24.6	24.5	0.485	0.485	0.486	0.492	7.7	7.5	6.7	6.1	8.4	8.3	8.2	8.2
11/16/2006	16.5	16.4	16.2	16.1	0.591	0.590	0.591	0.592	10.3	10.2	10.0	9.7	8.2	8.2	8.2	8.2
2/8/2007	13.6	13.6	13.5	13.5	0.589	0.588	0.588	0.589	8.7	9.0	9.0	8.9	8.0	8.1	8.1	8.1
5/17/2007	20.8	20.7	20.6	20.4	0.638	0.635	0.633	0.633	8.8	8.8	8.7	8.2	8.4	8.3	8.3	8.3
8/16/2007	25.8	25.7	25.5	25.3	0.671	0.669	0.671	0.672	9.3	9.6	9.4	8.7	9.0	9.0	9.0	8.9
11/7/2007	17.6	17.6	17.5	17.5	0.724	0.724	0.724	0.724	8.5	8.5	8.4	8.2	8.39	8.39	8.37	8.34
2/14/2008	14.49	14.48	14.45	14.40	0.64	0.64	0.64	0.64	9.60	9.58	9.60	9.61	8.26	8.28	8.29	8.30
5/8/2008	18.83	18.80	18.70	18.97	0.78	0.78	0.78	0.79	7.65	7.64	7.52	6.41	8.09	8.17	8.20	8.26
8/7/2008	26.66	26.44	26.33	26.06	0.90	0.90	0.89	0.89	6.58	6.24	6.00	5.71	8.21	8.18	8.14	8.11
11/20/2008	16.69	16.46	16.36	16.34	1.04	1.04	1.04	1.04	9.90	9.82	9.61	9.46	8.09	8.07	8.04	8.02
2/19/2009	12.44	12.30	12.17	12.13	0.80	0.80	0.80	0.80	11.03	11.02	10.98	10.99	8.31	8.30	8.29	8.29
5/21/2009	23.91	23.71	23.62	23.51	0.85	0.85	0.85	0.85	8.60	8.44	8.29	8.14	8.28	8.28	8.27	8.26

In 2008, the Regional Board sampled nine locations in Echo Park Lake. Site location descriptions are listed in Table G-44. As the lake is relatively shallow and well mixed by wind action and aerators, the sampling team collected analytical samples from the lake surface only. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label.

Table G-44. Site Locations for the 2008 Echo Park Lake Monitoring Event

Sampling Event	Regional Board Site Number	Project Site	Alternate Label
June	1	Below City of LA storm drain	EPL-1
	2	Below County of LA storm drain	EPL-6
	3	Lake mid-point	EPL-8
	4	Lotus Beds	EPL-LB
	5	Hydroponic Island	EPL-HI
December	1	Shoreline sample at northwest segment of lake	EPL-12
	2	Shoreline sample at western side of lake	EPL-11
	3	Shoreline sample on southern edge of lake	EPL-10
	4	Shoreline sample on eastern side of lake	EPL-13
	5	Shoreline sample near City of LA storm drain	EPL-1 (shoreline)

Ammonia concentrations in Echo Park Lake were fairly similar at all three sampled locations (Sites EPL-8, EPL-LB, and EPL-HI) on June 25, 2008 and ranged from 0.131 mg/L to 0.136 mg/L (Table G-45). TKN at the lake midpoint and near the hydroponic island ranged from 1.38 mg/L to 1.49 mg/L; the concentration was higher in the lotus beds at 4.72 mg/L. Concentrations of nitrate, nitrite, orthophosphate, and total phosphate were all less than the reporting limits of 0.1, 0.1, 0.4, and 0.5 mg/L, respectively. Total dissolved solids ranged from 565 mg/L to 651 mg/L, and suspended solids ranged from 11.2 mg/L to 96 mg/L.

Table G-45. Analytical Data for the June 25, 2008 Echo Park Lake Sampling Event

Alternate Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total Phosphate (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)
EPL-8	0.136	1.41	<0.1	<0.1	<0.4	<0.5	603	13.2
EPL-LP	0.131	4.72	<0.1	<0.1	<0.4	<0.5	651	96.0
EPL-HI	0.133	1.38	<0.1	<0.1	<0.4	<0.5	565	11.6
EPL-8 (dup.)	0.129	1.49	<0.1	<0.1	<0.4	<0.5	582	11.2

Field data were collected at five sites in Echo Park Lake by the Regional Board (Table G-46). Temperatures at sites EPL-6, EPL-8, and EPL-HI ranged from 25.46 ºC to 27.76 ºC. Temperatures on the north end of the lake were higher (28.95 and 32.0 ºC) but were also collected later in the day. Dissolved oxygen concentrations ranged from 4.95 mg/L to 9.82 mg/L, and pH ranged from 8.21 to 8.56 (note that the pH meter was not producing calibration results within the acceptable range). Electrical conductivity did not vary significantly at any location and ranged from 0.680 mS/cm to 0.688 mS/cm. The Secchi depth readings at sites EPL-6, EPL-8, and EPL-HI ranged from 0.66 m to 0.68 m. Chlorophyll *a* samples collected at depths less than the Secchi depth at each site ranged from 10.9 µg/L to 15 µg/L, with the exception of Site 4 in the lotus beds where the concentration was 26.7 µg/L. At depths greater than the Secchi depth at each site, chlorophyll *a* concentrations were generally higher with concentrations ranging from 16.1 µg/L to 53.6 µg/L. These higher numbers may reflect chlorophyll *a* contained in decaying algae that has settled to the bottom of the lake. A description of the methodology or equipment used to measure chlorophyll *a* concentrations in the field was not provided.

Table G-46. Field Data for the June 25, 2008 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	DO (mg/L)	EC (mS/cm)	pH[1]	Chl a (ug/L)	Secchi Depth (m)
EPL-8	Begin time: 10:06	Surface	26.63	6.57	0.685	8.27	12.7	0.66
		0.5	25.80	6.51	0.685	8.30	15.0	
		1.0	25.54	6.26	0.686	8.26	15.8	
		1.5	25.46	4.95	0.686	8.21	53.6	
		2	25.52	5.33	0.684	8.22	0.8	
EPL-HI	Begin time: 11:05	Surface	26.69	8.11	0.686	8.46	11.9	0.68
		0.5	26.48	7.44	0.685	8.46	11.8	
		1.0 (bottom)	25.83	6.68	0.687	8.43	16.1	
EPL-6	Begin time: 12:23	Surface	27.76	8.3	0.685	8.53	10.9	0.68
		0.5	26.94	7.6	0.686	8.52	12.0	
		1.0 (bottom)	26.76	7.46	0.685	8.50	20.6	
EPL-LB	14:10	Surface	32.0	6.25	0.688	8.29	26.7	NA
EPL-1	14:49	Surface	28.95	9.82	0.680	8.56	14.0	NA

[1] pH calibration was outside of accepted range. Data should not be used quantitatively.

Samples were also collected on December 18, 2008 from five shoreline locations at a depth of approximately 4 inches (Table G-47). Ammonia ranged from 0.206 mg/L to 0.344 mg/L. TKN ranged from 1.1 mg/L to 1.55 mg/L with one measurement near the lotus pond that was less than the reporting limit of 1 mg/L. Nitrate ranged from 0.215 mg/L to 0.325 mg/L. All samples of nitrite, orthophosphate, and total phosphate were less than the reporting limits of 0.1 mg/L, 0.4 mg/L, and 0.5 mg/L, respectively. Total dissolved solids ranged from 549 mg/L to 576 mg/L. All measurements of suspended solids were less than the reporting limit of 10 mg/L except at EPL-1. Chlorophyll *a* ranged from 8.5 µg/L to 20.2 µg/L.

Table G-47. Analytical Data for the December 18, 2008 Echo Park Lake Sampling Event

Alternate Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	Total phos-phate (mg/L)	Total Dissolved Solids (mg/L)	Suspen ded Solids (mg/L)	Chloro-phyll *a* (µg/L)
EPL-12	0.344	1.55	0.215	<0.1	<0.1	<0.4	549	46.5	20.2
EPL-11	0.209	1.1	0.325	<0.1	<0.1	<0.4	576	<RL	10.4
EPL-10	0.239	1.1	0.3	<0.1	<0.1	<0.4	576	<RL	15.1
EPL-13	0.215	<RL	0.317	<0.1	<0.1	<0.4	576	<RL	8.9
EPL-1 (shoreline)	0.234	1.24	0.309	<0.1	<0.1	<0.4	567	<RL	10.7
EPL-11 (dup.)	0.206	1.16	0.312	<0.1	<0.1	<0.4	576	<RL	8.5

Field data from the December 2008 sampling event are summarized in Table G-48. Temperature in the lake ranged from 8 ºC to 10.5 ºC; pH ranged from 7.7 to 8.1.

Table G-48. Field Data for the December 18, 2008 Echo Park Lake Sampling Event

Site	Time	Temp (C)	pH	Total Depth (m)
EPL-12	9:00	8	7.7	0.2
EPL-11	10:15	10	8.0	0.7
EPL-10	11:20	10	8.0	0.9
EPL-13	12:10	10.5	8.0	0.6
EPL-1 (shoreline)	13:50	10.5	8.1	Not reported

On March 10, 2009, USEPA and the Regional Board sampled Echo Park Lake at three locations (Table G-49). Samples were collected at Site EPL-50 at 9:50 from a depth of 0.61 m. Site EPL-51 was also sampled from a depth of 0.61 m at 10:30. Site EPL-52 was sampled at 11:00 from a depth of 0.46 m. Ammonia concentrations ranged from 0.04 mg/L to 0.06 mg/L, and TKN ranged from 0.7 mg/L to 1.3 mg/L. Nitrate was approximately 0.15 mg/L at each station and nitrite was less than detection. Orthophosphate was less than detection at each station and total phosphorus generally ranged from 0.033 mg/L to 0.071 mg/L. The total phosphorus measured at EPL-52 was 0.762 mg/L, though the field duplicate had a value of 0.071 mg/L. Chlorophyll *a* measurements in the lake ranged from 14.2 µg/L to 15.2 µg/L.

Sites EPL-51 and 52 and the potable water input (PW) were sampled again on August 4th, 2009 (also in Table G-49). Site-51 was sampled at 8:15 and had a total depth of 1.7 meters and a Secchi depth of 0.3 meters. Site-52 was sampled at 9:00, had a depth of 1.8 meters, and a Secchi depth of 0.6 meters. All nitrogen parameters (ammonia, TKN, nitrate, and nitrite) were below detection limits at both in-lake sites. Total phosphorus was 0.196 mg/L at EPL-51 and 0.195 mg/L at EPL-52. The orthophosphate concentrations were 0.0850 and 0.0917 at sites EPL-51 and EPL-52, respectively. The TSS average at the stations was 15.2 mg/L and the chlorophyll *a* average was 15.3µg/L. The TDS was 505 mg/L at EPL-51 and 494 mg/L at EPL-52. The lab noted that these samples were analyzed after the allowable holding time limit, but the data are included here because the TDS amounts were in the expected range and the extended holding time does not appear to have greatly affected these measurements.

Table G-49. In-lake Water Column Measurements for Echo Park Lake

Date	Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	TP (mg/L)	TDS (mg/L)	TSS (mg/L)	Chl *a* (µg/L)
3/10/2009	EPL-50	0.04	0.97	0.15	<0.01	<0.008	0.033	427	4.6	15.2
	EPL-51	0.05	1.30	0.14	<0.01	<0.008	0.046	438	7.0	14.2
	EPL-52	0.06	0.70	0.15	<0.01	<0.008	0.762	442	7.3	14.6
	EPL-52 (duplicate)	0.06	NA	NA	NA	NA	0.071	NA	NA	NA
8/4/2009	EPL-51	<0.03	<0.456	<0.01	<0.01	0.0850	0.196	505	18.0	15.5
	EPL-52	<0.03	<0.456	<0.01	<0.01	0.0917	0.195	494	12.3	15.0
	EPL-PW	<0.03	<0.456	0.9	<0.01	0.0202	0.122	348	<0.5	NA

Additional data taken during the sampling events on March 10 and August 4, 2009, are shown in Table G-50. The chloride concentrations increased significantly between the winter and summer events. The winter average of chloride was 76.2 mg/L and the summer concentration was 92.9 mg/L. The sulfate concentrations were lower in the summer, at an average of 91 mg/L compared to the winter average of 131 mg/L. There were not significant changes between the March and August measurements of alkalinity, DOC, and TOC.

Table G-50. Supplemental Water Quality Monitoring for In-lake Samples in the Echo Park Lake

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	DOC (mg/L)	TOC (mg/L)
3/10/2009	EPL-50	9:50	75.8	130	133	181	3.2	2.6
	EPL-51	10:30	76.7	132	126	178	3.1	4.8
	EPL-52	11:00	76.1	130	126	178	3.0	3.4
8/4/2009	EPL-51	8:15	92.7	91	136	191.4	3.6	5.3
	EPL-52	9:00	93.1	91	140	187.6	3.6	5.5
	EPL-PW	10:40	66.4	65.4	141	128.6	0.95	0.85

Profile data collected in Echo Park Lake are summarized in Table G-51. Based on this data the lake appears well mixed both vertically and spatially. DO concentrations in the lake generally ranged from 7.0 mg/L to 8.6 mg/L with one reading of 10.0 mg/L from a surface sample. pH ranged from 7.5 to 7.9. Profile depths listed in the field notes (ranging from surface to 2.5 meters) were multiplied by the ratio of total depth reported in the field notes to the depth measured on the probe cable at each monitoring station because the probe was drifting and indicating depths greater than actual (Anna Sofranko, USEPA Region IX, personal communication, May 12, 2009).

Table G-51. Field Data for the March 10, 2009 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-50	9:40	Surface	15.2	7.1	10.0	1.27	1.40
		0.28	15.3	7.5	8.4		
		0.56	15.3	7.5	8.3		
		0.84	15.2	7.6	8.3		
		1.12	15.1	7.6	8.1		
		1.40	15.1	7.7	7.9		
EPL-51	10:30	Surface	15.5	7.7	8.6	1.14	1.67
		0.34	15.4	7.8	8.2		
		0.67	15.3	7.8	8.2		
		1.00	15.0	7.8	8.3		
		1.34	15.0	7.9	8.2		
		1.68	15.0	7.9	8.0		
EPL-52	11:00	Surface	16.2	7.9	7.5	0.91	1.83
		0.36	16.1	7.9	7.5		
		0.73	16.0	7.9	7.2		
		1.10	15.7	7.9	7.4		
		1.46	15.5	7.9	7.3		
		1.82	15.3	7.8	7.0		

Profile data collected in Echo Park Lake in the summer of 2009 are summarized in Table G-52. DO concentrations in the lake ranged from 6.4 mg/L to 7.6 mg/L. The pH ranged from 8.3 to 8.6. Based on this data the lake appears well mixed vertically in the summer as well as in the spring. The Secchi depths at EPL-51 and EPL-52 were much less during the August sampling. The Secchi depth at EPL-51 is 0.84 meters less than the spring Secchi depth. At station EPL-52, the Secchi depth is 0.35 meters less in August than in March. The temperature of the lake ranged from 26.1 to 26.5°C, approximately 10°C higher than the March temperatures. The increased temperature and decreased lake clarity illustrate an increase in algal productivity during this summer season.

Table G-52. Field Data for the August 4, 2009 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-51	9:05	Surface	26.1	8.3	6.7	0.30	1.73
		0.5	26.1	8.3	6.6		
		1.0	26.1	8.3	6.5		
		1.5	26.1	8.4	6.4		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-52	9:45	Surface	26.5	8.6	7.5	0.56	1.78
		0.1	26.5	8.6	7.5		
		0.5	26.4	8.6	7.5		
		1.0	26.3	8.6	7.6		
		1.5	26.3	8.6	7.5		

Field data were collected for the potable water source during the August sampling event. After purging the line for approximately five minutes, the pH was 7.48, the DO was 8.73 mg/L, and the temperature was 21.95 ºC.

The city of Los Angeles provided water quality monitoring data for the Glendale Water Reclamation Plant, which may be used to supplement lake levels and irrigate parkland at Echo Park in the future. Table G-53 summarizes the average water quality for this source based on monthly averages reported for 2008 and 2009.

Table G-53. Average Water Quality for the Glendale Water Reclamation Plant

NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Orthophosphate (mg/L)	TP (mg/L)
1.3	3.17	5.64	0.009	1.76	1.93

G.6.2 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 Echo Park Lake was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Echo Park Lake is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Echo Park Lake (pink triangle, Figure G-13), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-54. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to two meters. The range of the 31 dissolved copper samples was between less than 10 µg/L and 105 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to two meters. The 31 samples collected ranged in concentration from less than 1 µg/L to 105 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Echo Park Lake states that copper and lead were not supporting the assessed uses: 31 measurements had a maximum lead concentration of 105 µg/L, a maximum copper concentration of 105 µg/L, and a maximum zinc concentration of 14 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-54. Echo Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/14/1992	0	14	<1
	1.5	15	<1
7/14/1992	0	20	<1
	1.5	19	<1
7/14/1992	0	N/A	<1
	1.3	20	<1
8/13/1992	0	87	2
	1.5	N/A	<1
8/13/1992	0	42	<1
	2	33	<1
	0	31	2
9/17/1992	0	105	1
	1.7	95	2
10/15/1992	0	40	6
	1.7	16	<1
11/5/1992	0	<10	1
	1.5	<10	1
12/8/1992	0	<10	6
	1.5	<10	6
1/12/1993	0	<10	4
	1.5	15	1
2/2/1993	0	15	7
	1.5	<10	2
3/17/1993	0	<10	37
	1.5	13	97
4/7/1993	0	<10	5
	1.5	<10	18
5/3/1993	0	11	1
	2	10	1
6/8/1993	0	<10	105
	1.5	11	60

Table G-55 presents 61 additional metal samples that were collected by the USEPA, Regional Board, and/or the city of Los Angeles between October 2008 and March 2010. Samples were collected at locations: EPL-1, EPL-2, EPL-3, EPL-4, EPL-5, EPL-7, EPL-8, EPL-9, EPL-12, EPL-50, EPL-51, EPL-52 and EPL-Shore. Sites were analyzed for dissolved cadmium, copper, lead and/or zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.1 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.2 µg/L; copper concentrations were between 0.7 µg/L and 26.3 µg/L; lead concentrations ranged from 0.1 µg/L to 5.5 µg/L; and zinc concentrations were <0.1 µg/L to 2.7 µg/L. In addition, three total lead samples were collected by the Regional Board in June 2008 at EPL3, EPL4, and EPL5. The total lead concentrations ranged from 4.1µg/L to 14.3 µg/L (with a hardness range of 265 mg/L to 267 mg/L). Metal toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 dissolved metals sampling resulted in a hardness range of 106 mg/L to 283 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-55 were weighted more heavily in the assessment.

Table G-55. Metals Data for the 2008-2010 Echo Park Lake Sampling Events

Organiz-ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	11/17/2004	EPL 7/8/9	153	N/A	7.3	4.7	N/A	average of 7, 8, 9
City LA	2/17/2005	EPL 7/8/9	106	N/A	11.0	3.0	N/A	average of 7, 8, 9
City LA	5/19/2005	EPL 7/8/9	133	N/A	7.7	1.0	N/A	average of 7, 8, 9
City LA	8/18/2005	EPL 7/8/9	130	N/A	26.3	4.0	N/A	average of 7, 8, 9
City LA	11/17/2005	EPL 7/8/9	154	N/A	4.7	5.5	N/A	average of 7, 8, 9
City LA	2/9/2006	EPL 7/8/9	167	N/A	4.0	1.0	N/A	average of 7, 8, 9
City LA	5/11/2006	EPL 7/8/9	147	N/A	9.0	1.2	N/A	average of 7, 8, 9
City LA	8/17/2006	EPL 7/8/9	111	N/A	13.0	2.3	N/A	average of 7, 8, 9
City LA	11/16/2006	EPL 7/8/9	166	N/A	10.3	1.6	N/A	average of 7, 8, 9
City LA	2/8/2007	EPL 7/8/9	168	N/A	6.3	1.6	N/A	average of 7, 8, 9
City LA	5/17/2007	EPL 7/8/9	179	N/A	17.7	1.8	N/A	average of 7, 8, 9
City LA	8/16/2007	EPL 7/8/9	167	N/A	5.0	1.1	N/A	average of 7, 8, 9
City LA	11/7/2007	EPL 7/8/9	184	N/A	1.0	1.1	N/A	average of 7, 8, 9
City LA	2/14/2008	EPL 7/8/9	186	N/A	2.7	1.1	N/A	average of 7, 8, 9
City LA	5/8/2008	EPL 7/8/9	231	N/A	5.6	3.1	N/A	average of 7, 8, 9
City LA	8/7/2008	EPL 7/8/9	214	N/A	6.5	2.0	N/A	average of 7, 8, 9
City LA	11/20/2008	EPL 7/8/9	283	N/A	3.1	2.0	N/A	average of 7, 8, 9
RB	12/18/2008	EPL 1 R1	208	<0.2	1.6	0.5	2.3	average of replicates; lotus bed location
RB	12/18/2008	EPL 2/3/4	216	<0.2	2.5	0.2	2.4	average of duplicates; average of site 2, 3, &

Organiz -ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
								4 (shoreline)
RB	12/18/2008	EPL 5 R1	209	<0.2	2.6	0.2	2.7	oxbow location
City LA	2/19/2009	EPL 7/8/9	226	N/A	3.1	2.0	N/A	average of 7, 8, 9
RB/EPA	3/10/2009	EPL 50/51/52	178	<0.2	1.8	1.0	1.5	average of duplicates & MS; average of 50, 51 & 52
City LA	3/26/2009	EPL-shore	234	N/A	4.9	0.5	N/A	
City LA	5/21/2009	EPL 7/8/9	230	N/A	2.7	2.0	N/A	average of 7, 8, 9
RB/EPA	8/4/2009	EPL 1	186	<0.2	1.7	0.7	<0.1	
RB/EPA	8/4/2009	EPL 12	192	<0.2	0.7	0.5	2.1	
RB/EPA	8/4/2009	EPL 5	190	<0.2	1.5	0.6	0.4	
RB/EPA	8/4/2009	EPL 51/52	190	0.2	1.5	0.6	0.1	average of replicates and 51/52
City LA	8/18/2009	EPL 7/8/9	189	N/A	2.9	0.4	N/A	average of replicates and sites
City LA	8/28/2009	EPL 8	199	N/A	3.8	0.7	N/A	
City LA	8/28/2009	EPL-shore	204	N/A	4.8	0.6	N/A	
City LA	9/4/2009	EPL 8	198	N/A	1.4	0.2	N/A	
City LA	9/4/2009	EPL-shore	209	N/A	1.7	0.2	N/A	
City LA	9/11/2009	EPL-shore	206	N/A	2.8	0.5	N/A	
City LA	9/25/2009	EPL-shore	207	N/A	3.8	0.5	N/A	
City LA	10/2/2009	EPL-shore	203	N/A	3.0	0.2	N/A	
City LA	10/9/2009	EPL-shore	196	N/A	4.0	0.7	N/A	
City LA	10/16/2009	EPL-shore	222	N/A	4.8	0.5	N/A	sampling after rainy weather
City LA	10/23/2009	EPL-shore	213	N/A	4.8	0.8	N/A	
City LA	10/30/2009	EPL-shore	229	N/A	1.6	0.4	N/A	
City LA	11/6/2009	EPL-shore	231	N/A	3.7	0.7	N/A	
City LA	11/13/2009	EPL-shore	216	N/A	2.8	0.3	N/A	
City LA	11/20/2009	EPL-shore	208	N/A	3.4	0.5	N/A	
RB/EPA	12/1/2009	EPL 1	193	0.2	1.9	0.2	0.1	average of replicates
RB/EPA	12/1/2009	EPL 12	193	<0.2	1.5	0.7	0.4	
RB/EPA	12/1/2009	EPL 51/52	191	<0.2	1.4	0.4	0.3	average of replicates and 51/52

Organiz-ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	12/4/2009	EPL-shore	216	N/A	4.0	0.4	N/A	
City LA	12/11/2009	EPL-shore	214	N/A	4.2	0.3	N/A	
City LA	12/18/2009	EPL-shore	215	N/A	N/A	0.3	N/A	
City LA	12/18/09 & 12/22/09	Mid Lake EPL 7/8/9	231	N/A	3.3	0.4	N/A	average of dry mid-lake samples
City LA	1/8/2010	EPL-shore	250	N/A	4.5	0.5	N/A	
City LA	1/15/2010	EPL-shore	227	N/A	4.6	0.4	N/A	
City LA	1/22/2010	EPL-shore	197	N/A	3.0	0.4	N/A	
City LA	2/5/2010	EPL-shore	218	N/A	4.0	0.4	N/A	
City LA	2/12/2010	EPL-shore	212	N/A	4.9	0.2	N/A	
City LA	2/16/2010	Mid Lake EPL 7/8/9	215	N/A	2.9	0.2	N/A	average of mid-lake samples
City LA	2/19/2010	EPL-shore	226	N/A	2.7	0.3	N/A	
City LA	2/26/2010	EPL-shore	212	N/A	2.2	0.1	N/A	
City LA	3/5/2010	EPL-shore	236	N/A	2.4	0.2	N/A	
City LA	3/12/2010	EPL-shore	234	N/A	3.6	0.4	N/A	
City LA	3/19/2010	EPL-shore	236	N/A	6.6	2.5	N/A	

N/A = No data available.
RB = Regional Board
EPA = USEPA
City LA = City of Los Angeles

G.6.3 MONITORING RELATED TO ORGANOCHLORINE PESTICIDES AND PCBS IMPAIRMENT

Echo Park Lake is impaired by chlordane, dieldrin, and PCBs. Monitoring data for chlordane, DDT, dieldrin, and PCBs in Echo Park Lake are reviewed in this section. Echo Park Lake is not listed for DDT but those data are presented here for completeness because other lakes in the region (Peck Road Park Lake and Puddingstone Reservoir) are affected by this contaminant.

In 2008, UCLA conducted organics measurements at Echo Park Lake at five locations. Site location descriptions are listed in Table G-56. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label.

Table G-56. Site Locations for the 2008 UCLA Echo Park Lake Monitoring Event

UCLA Site Number	Project Site	Alternate Label
1	Below City of LA storm drain	EPL-1
2	Below County of LA storm drain	EPL-6
3	Lower Lake	EPL-9
4	Lotus Beds	EPL-LB
5	Hydroponic Island	EPL-HI

The Regional Board conducted organics monitoring in Echo Park Lake in December 2008. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label (Table G-57).

Table G-57. Site Locations for the 2008 Regional Board Echo Park Lake Monitoring Event

Regional Board Site Number	Project Site	Alternate Label
1	Shoreline sample at northwest segment of lake	EPL-12
2	Shoreline sample at western side of lake	EPL-11
3	Shoreline sample on southern edge of lake	EPL-10
4	Shoreline sample on eastern side of lake	EPL-13
5	Shoreline sample near City of LA storm drain	EPL-1 (shoreline)

Additional samples were collected by the USEPA and the Regional Board in December 2009 at EPL-12, EPL-51, and EPL-52. Alternative labels were not needed for these locations.

G.6.3.1 Water Column Data Observed in Echo Park Lake

Lake water samples were collected from EPL-9 and EPL-LB in the summer of 2008 as part of an organics study performed by UCLA and funded by a grant managed by the Regional Board.

The samples were analyzed for chlordane, DDT, dieldrin, and PCBs. All contaminants were below reportable levels. Table G-58 shows the results and detection limits for each constituent. PCB-31 was detected at EPL-9 and PCB-5 was detected at EPL-LB, but not at reportable levels .

Table G-58. Results from Water Column Samples Collected at Echo Park Lake in Summer 2008

	EPL-9			EPL-9 (field duplicate)			EPL-LB		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	(ng/L)								
Chlordane-gamma	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
Chlordane-alpha	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
4,4'-DDE	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND

Contaminant	EPL-9			EPL-9 (field duplicate)			EPL-LB		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)								
4,4'-DDD	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDT	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
Dieldrin	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
PCB 5	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	8.43*
PCB 18	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 31	1.50	15.00	3.50*	1.50	15.00	4.15*	1.50	15.00	ND
PCB 52	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 44	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 66	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 101	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 87	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 151	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 110	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 153	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 141	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 138	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 187	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 183	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 180	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 170	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 206	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND

*Result was above detection limit but below the reporting limit

Water samples from Echo Park Lake were also collected by the Regional Board on December 18, 2008 at EPL-12, EPL-11, EPL-10, EPL-13, and EPL-1. The samples were only analyzed for PCB congeners, no organochloride pesticides were analyzed. PCBs at all stations were not detected. Each congener had a detection limit of 1 ng/L (Table G-59).

Table G-59. Results from Water Column Samples Collected at Echo Park Lake on December 18th, 2008

Contaminant (ng/L)	EPL-12	EPL-11	EPL-10	EPL-13	EPL-1	MDL
PCB003	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	EPL-12	EPL-11	EPL-10	EPL-13	EPL-1	MDL
PCB028	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	1.0
PCB119	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	EPL-12	EPL-11	EPL-10	EPL-13	EPL-1	MDL
PCB167	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	1.0

G.6.3.2 Porewater Data Observed in Echo Park Lake

Samples of porewater from summer 2008 were analyzed for EPL-1, EPL-6, EPL-9, EPL-LB and EPL-HI. PCB-5 was detected in the porewater at EPL-6 and EPL-9, and PCB-31 was detected at EPL-HI. None of the organic pollutants were detected at EPL-LB. EPL-1 was not reported due to a laboratory error during analysis. The porewater from EPL-6 was the only sample with sufficient TSS for analysis. PCB-66 was detected in the TSS, but not at reportable levels. The results of the porewater analysis are shown in Table G-60 (see Stenstrom et al., 2009 for raw data).

Table G-60. Results from Porewater Samples Collected at Echo Park Lake in Summer 2008

Contaminant	Porewater (ng/L)					TSS in Porewater (µg/kg)	
	EPL-6	EPL-9	EPL-LB	EPL-HI	MDL	EPL-6	MDL
Chlordane	ND	ND	ND	ND	15	ND	3.65
DDT	ND	ND	ND	ND	30	ND	7.31
Dieldrin	ND	ND	ND	ND	30	ND	7.31
Total PCBs	DNQ[1]	DNQ[1]	ND	DNQ[2]	15	DNQ[3]	3.65

[1] PCB-5 was detected in these samples at less than the reporting level (150 ng/L)

[2] PCB-31 was detected in this sample at less than the reporting level (150 ng/L)

[3] PCB- 66 was detected less than the reporting level (36.54 µg/kg)

Porewater from EPL-1 and EPL-LB in fall 2008 was also analyzed as part of the UCLA study. No contaminants were detected in the porewater from either site. The results of this analysis are shown in Table G-61 (see Stenstrom et al., 2009 for raw data).

Table G-61. Results from Porewater Samples Collected at Echo Park Lake in Fall 2008

Contaminant (ng/L)	EPL-1	EPL-LB	MDL
Chlordane	ND	ND	15 – 1,500
DDT	ND	ND	30 – 3,000
Dieldrin	ND	ND	30 – 3,000
Total PCBs	ND	ND	15 – 1,500

G.6.3.3 Fish Tissue Levels Observed in Echo Park Lake

Concentrations of Aroclor PCBs, chlordane, DDT, dieldrin, and PCBs in largemouth bass, common carp, and bullhead species were reported by SWAMP (Davis et al., 2008) and TSMP (2009), shown in Table G-62. Concentrations of chlordane were highest in bullhead fish; 66.0 ppb on average. Bullhead fish also had higher average concentrations of DDTs and dieldrin than the other species (60.0 ppb and 7.0 ppb, respectively). Largemouth bass had the lowest average concentrations of the organochlorine pesticides. PCBs were only tested in common carp and largemouth bass. The average concentrations of PCBs in common carp was 81.8 ppb and 49.0 ppb in largemouth bass. The average dieldrin concentrations (1.13 ppb) are higher than the 0.45 ppb FCG for dieldrin.

Table G-62. Compiled Fish Tissue Analytical Data for Echo Park Lake

Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
Aroclor PCBs	6/17/1987	Bullhead	50	236	205.6
Aroclor PCBs	6/17/1987	Largemouth Bass	84	145	42.6
Aroclor PCBs	4/19/1991	Largemouth Bass	ND	244	271.3
Aroclor PCBs	4/24/1992	Largemouth Bass	60	315	581.8
Total PCBs	Summer 2007	Common Carp	119.01	501	1714.4
Total PCBs	Summer 2007	Common Carp	82.618	380	807.4
Total PCBs	Summer 2007	Largemouth Bass	64.716	498	1823.6
Total PCBs	Summer 2007	Largemouth Bass	31.478	380	916
Total PCBs	4/13/2010	Largemouth Bass	50.863	377.2	901
Total PCBs	4/13/2010	Common Carp	43.861	377.2	928
Chlordane	6/17/1987	Bullhead	66	236	205.6
Chlordane	6/17/1987	Largemouth Bass	17.8	145	42.6
Chlordane	4/19/1991	Largemouth Bass	ND	244	271.3
Chlordane	4/24/1992	Largemouth Bass	ND	315	581.8
Chlordane	Summer 2007	Common Carp	32.19	501	1714.4
Chlordane	Summer 2007	Common Carp	21.96	380	807.4

Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
Chlordane	Summer 2007	Largemouth Bass	15.484	498	1823.6
Chlordane	Summer 2007	Largemouth Bass	0.844	380	916
Chlordane	4/13/2010	Largemouth Bass	2.517	377.2	901
Chlordane	4/13/2010	Common Carp	4.216	377.2	928
DDTs	6/17/1987	Bullhead	60	236	205.6
DDTs	6/17/1987	Largemouth Bass	30	145	42.6
DDTs	4/19/1991	Largemouth Bass	ND	244	271.3
DDTs	4/24/1992	Largemouth Bass	11	315	581.8
DDTs	Summer 2007	Common Carp	23.458	501	1714.4
DDTs	Summer 2007	Common Carp	14.87	380	807.4
DDTs	Summer 2007	Largemouth Bass	13.029	498	1823.6
DDTs	Summer 2007	Largemouth Bass	6.35	380	916
DDTs	4/13/2010	Largemouth Bass	7.448	377.2	901
DDTs	4/13/2010	Common Carp	7.3	377.2	928
Dieldrin	6/17/1987	Bullhead	7	236	205.6
Dieldrin	6/17/1987	Largemouth Bass	ND	145	42.6
Dieldrin	4/19/1991	Largemouth Bass	ND	244	271.3
Dieldrin	4/24/1992	Largemouth Bass	ND	315	581.8
Dieldrin	Summer 2007	Common Carp	1.08	501	1714.4
Dieldrin	Summer 2007	Common Carp	0.79	380	807.4
Dieldrin	Summer 2007	Largemouth Bass	0.848	498	1823.6
Dieldrin	Summer 2007	Largemouth Bass	0.585	380	916
Dieldrin	4/13/2010	Largemouth Bass	[0.45]*	377.2	901
Dieldrin	4/13/2010	Common Carp	0.538	377.2	928

ND = Non-detect

* Values in square brackets are reported concentrations below the practical reporting limit and are included in the averages.

G.6.3.4 Sediment Data Observed in Echo Park Lake

Sediment samples from Echo Park Lake were collected for the UCLA study in the fall and summer of 2008 and then by USEPA and the Regional Board in December 2009. The results from the UCLA study are shown in Table G-63 and Table G-64 for fall and summer, respectively. All samples had laboratory duplicates and a field duplicate was also collected during each event. The only contaminant above reportable limits in the fall was PCB-5 at EPL-1. PCB-66 and PCB-153 were also detected but not quantified at EPL-1. DDT was detected at EPL-LB, but not quantifiable (Table G-63).

Table G-63. Results from Sediment Samples Collected at Echo Park Lake in Fall 2008

Contaminant	EPL-1			EPL-1 (lab dup)			EPL-1 (field dup)			EPL-1 (lab dup of field dup)			EPL-LB			EPL-LB (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																	
Chlordane-gamma	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
Chlordane-alpha	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
4,4'-DDE	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
4,4'-DDD	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
4,4'-DDT	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	15.79*	1.85	18.49	ND
Dieldrin	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
PCB 5	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	9.13	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 18	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 31	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 52	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 44	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 66	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	4.60*	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 101	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 87	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 151	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 110	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 153	1.04	10.35	1.44*	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 141	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 138	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 187	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 183	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 180	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 170	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 206	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND

Chlordane and several PCB congeners were detected in the summer 2008 sediment samples and concentrations of each detected congener are shown in Table G-64. Chlordane-gamma was detected at EPL-6 with a concentration of 8 µg/kg dry weight. EPL-6 was also found to have concentration of PCBs over the reporting limits for the following congeners: PCB-5, PCB-44, PCB-52 and PCB-66. EPL-9 had concentration of PCBs over the reporting limit for PCB-5 and PCB-52, while PCB-5 and PCB-138 were reportable at EPL-LB. Dieldrin and DDT were not detected at any of the sampled locations.

Table G-64. Results from Sediment Samples Collected at Echo Park Lake in Summer 2008

Contaminant	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																	
	EPL-1			EPL-1 (lab dup)			EPL-6			EPL-6 (lab dup)			EPL-6 (field dup)			EPL-6 (lab dup of field dup)		
Chlordane-gamma	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	8.02	0.83	5.37	ND	0.60	6.04	ND
Chlordane-alpha	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
4,4'-DDE	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
4,4'-DDD	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
4,4'-DDT	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
Dieldrin	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
PCB 5	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	12.68
PCB 18	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 31	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 52	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	50.32	0.60	6.04	5.49
PCB 44	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	41.98	0.60	6.04	2.56
PCB 66	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	12.11	0.60	6.04	ND
PCB 101	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 87	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 151	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 110	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 153	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 141	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 138	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 187	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 183	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 180	0.83	8.31	1.32*	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 170	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 206	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
Contaminant	**EPL-9**			**EPL-9 (lab dup)**			**EPL-LB**			**EPL-LB (lab dup)**			**EPL-HI**			**EPL-HI (lab dup)**		
Chlordane-gamma	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
Chlordane-alpha	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
4,4'-DDE	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
4,4'-DDD	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
4,4'-DDT	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
Dieldrin	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
PCB 5	0.89	8.93	ND	0.93	9.33	36.49	1.23	12.29	164.7	1.23	12.30	94.62	0.63	6.29	ND	0.57	5.66	ND
PCB 18	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 31	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 52	0.89	8.93	10.84	0.93	9.33	12.37	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 44	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 66	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 101	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 87	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND

Contaminant	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																	
PCB 151	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 110	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 153	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 141	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 138	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	19.84	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 187	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 183	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 180	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 170	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 206	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND

*Results were above detection limit but lower than reporting limits

Sediment sampling was conducted by USEPA and the Regional Board at EPL-12, EPL-51, and EPL-52 on December 1, 2009. Similar to the fall and summer 2008 sampling, DDT and dieldrin were not detected in any of the sediment samples. However, 4,4 - DDE was detected at all three sites. Chlordane-gamma was detected at all three locations between 2 and 5 µg/kg dry weight and chlordane-alpha was detected at all three locations between 2 and 4.8 µg/kg dry weight. PCBs were detected at all three stations. Many of the same congeners were found at different EPL stations; PCB018, PCB095, PCB101, and PCB110 were detected at all locations. Other congeners detected and the results of the sediment sampling are shown in Table G-65.

Table G-65. Results from Sediment Samples Collected at Echo Park Lake on December 1, 2009

Contaminant (µg/kg dry weight)	EPL-12	EPL-51		EPL-52	MDL
		Result	Field Dup		
Chlordane-gamma	2.5	5	ND	2	1
Chlordane-alpha	2	4.8	ND	2.1	1
cis-Nonachlor	ND	ND	ND	ND	1
trans-Nonachlor	ND	ND	ND	ND	1
Oxychlordane	ND	ND	ND	ND	1
2,4 - DDD	ND	ND	ND	ND	1
2,4 - DDE	ND	ND	ND	ND	1
2,4 - DDT	ND	ND	ND	ND	1
4,4 - DDD	ND	ND	ND	ND	1
4,4 - DDE	18.6	21.1	20	5.8	1
4,4 - DDT	ND	ND	ND	ND	1
Dieldrin	ND	ND	ND	ND	1
PCB008	ND	ND	ND	2.1	1
PCB018	7.0	ND	7.4	3.5	1
PCB044	ND	ND	ND	6.7	1
PCB049	ND	ND	ND	3.0	1

Contaminant (µg/kg dry weight)	EPL-12	EPL-51		EPL-52	MDL
		Result	Field Dup		
PCB052	ND	ND	ND	9.0	1
PCB066	ND	ND	ND	8.1	1
PCB087	ND	30.5	ND	ND	1
PCB095	5.0	9.1	8.2	3.5	1
PCB097	ND	ND	ND	5.0	1
PCB099	ND	10.3	8.4	3.0	1
PCB101	7.3	11.4	9.1	4.7	1
PCB105	ND	ND	ND	12.3	1
PCB110	18.8	7.6	19.9	10.3	1
PCB119	ND	9.6	ND	1.3	1
PCB132	ND	ND	ND	4.6	1
PCB149	ND	13.6	ND	ND	1

G.6.3.5 Suspended Sediment Data Observed in Echo Park Lake

Echo Park Lake samples from summer 2008 were analyzed for pollutant concentrations associated with suspended sediments in the lake. Samples were collected at EPL-9, EPL-LB and EPL-HI. At EPL-9, PCB-31 was detected at 117 µg/kg dry weight and PCB-153 was also detected, but not within reportable limits. All other PCB congeners were less than the detection limits. No contaminants were detected at EPL-LB. The sample at EPL-HI did not have sufficient suspended solids for analysis. The results of the sampling are shown in Table G-66.

Table G-66. Results from Suspended Sediment Samples Collected at Echo Park Lake in Summer 2008

	EPL-9			EPL-LB		
	DL	RL	Result	DL	RL	Result
Contaminant	µg/kg dry suspended solids					
Chlordane-gamma	10.05	100.50	ND	3.19	31.95	ND
Chlordane-alpha	10.05	100.50	ND	3.19	31.95	ND
4,4'-DDE	20.10	201.01	ND	6.39	63.90	ND
4,4'-DDD	20.10	201.01	ND	6.39	63.90	ND
4,4'-DDT	20.10	201.01	ND	6.39	63.90	ND
Dieldrin	20.10	201.01	ND	6.39	63.90	ND
PCB 5	10.05	100.50	ND	3.19	31.95	ND
PCB 18	10.05	100.50	ND	3.19	31.95	ND
PCB 31	10.05	100.50	116.74	3.19	31.95	ND
PCB 52	10.05	100.50	ND	3.19	31.95	ND
PCB 44	10.05	100.50	ND	3.19	31.95	ND
PCB 66	10.05	100.50	ND	3.19	31.95	ND
PCB 101	10.05	100.50	ND	3.19	31.95	ND
PCB 87	10.05	100.50	ND	3.19	31.95	ND

PCB 151	10.05	100.50	ND	3.19	31.95	ND
PCB 110	10.05	100.50	ND	3.19	31.95	ND
PCB 153	10.05	100.50	37.43*	3.19	31.95	ND
PCB 141	10.05	100.50	ND	3.19	31.95	ND
PCB 138	10.05	100.50	ND	3.19	31.95	ND
PCB 187	10.05	100.50	ND	3.19	31.95	ND
PCB 183	10.05	100.50	ND	3.19	31.95	ND
PCB 180	10.05	100.50	ND	3.19	31.95	ND
PCB 170	10.05	100.50	ND	3.19	31.95	ND
PCB 206	10.05	100.50	ND	3.19	31.95	ND

*Results were above detection limits, but below reporting limits.
Note: EPL-HI was sampled but could not be analyzed.

G.7 Monitoring Data for Lake Calabasas

Monitoring data relevant to the impairments of Lake Calabasas are available from 1992, 1993, and 2004 through 2009. Figure G-14 shows the historical and recent monitoring locations for Lake Calabasas.

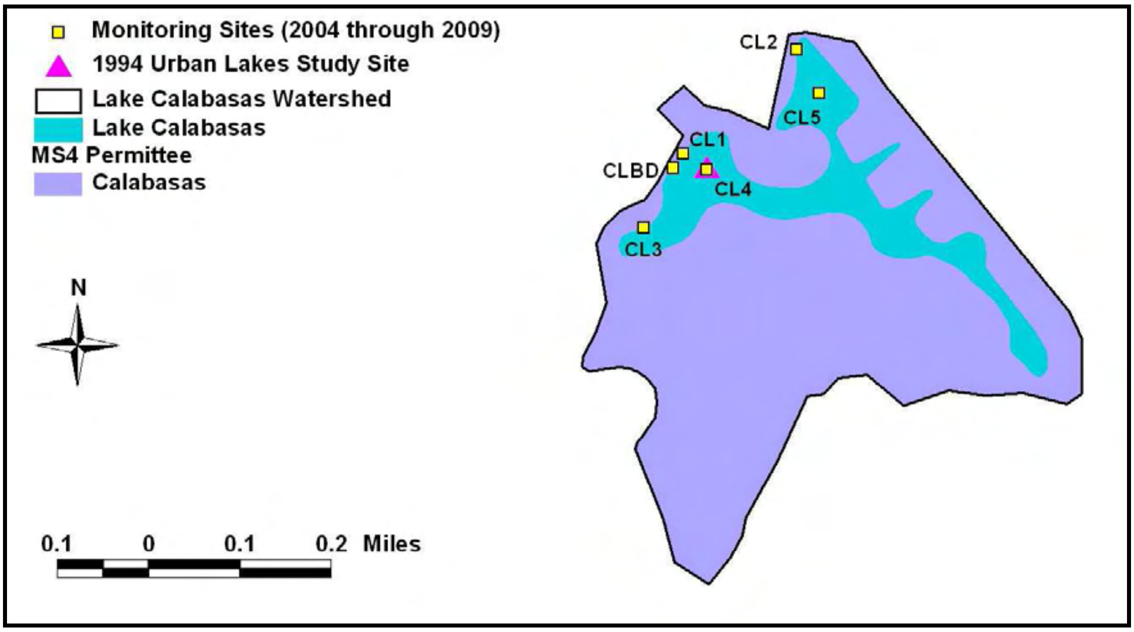

Figure G-14. Lake Calabasas Monitoring Sites

G.7.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Lake Calabasas was monitored from the western side of lake (pink triangle, Figure G-14) in 1992/1993 for water quality (Table G-67) as part of the Urban Lakes Study (UC Riverside, 1994). TKN ranged from 1.0 mg/L to 1.8 mg/L with two samples less than the reporting limit. Ammonium concentrations were usually less than or equivalent to the reporting limit although four samples collected in February and March 1993 ranged from 0.3 mg/L to 0.5 mg/L , the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All of the nitrite and nitrate samples were less than the reporting limit except one nitrate sample of 0.1 mg/L. Five of 28 phosphate samples measured 0.1 mg/L; the others were less than the reporting limit. Total phosphorus concentrations ranged from 0.1 mg/L to 0.2 mg/L with seven samples less than detection. pH in the lake ranged from 8.3 to 9.3. TOC ranged from 5.3 mg/L to 11.5 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 µg/L to 172 µg/L with an average of 39 µg/L.

Table G-67. Lake Calabasas 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/30/1992	0	1.4	<0.01	<0.01	<0.01	<0.01	0.1	8.8	8.2	479
	1.5	1.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	10	445
7/30/1992	0	1.6	<0.01	<0.01	<0.01	0.1	<0.01	9.1	8.8	430
	1.5	1.8	<0.01	<0.01	<0.01	0.1	<0.01	9.0	9.9	451
7/30/1992	0	1.3	<0.01	<0.01	<0.01	<0.01	0.1	9.2	11.2	444
	1.5	1.2	<0.01	<0.01	<0.01	0.1	0.1	9.2	9.9	455
8/17/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	9.2	9.5	547
	1.5	1.5	0.1	<0.01	<0.01	<0.01	0.1	9.2	9.5	516
9/23/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.8	9.4	495
	1.5	1.6	<0.01	<0.01	<0.01	<0.01	0.1	9.0	10	490
10/21/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.9	11.5	550
	2	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.9	10.4	561
11/9/1992	0	-	<0.01	<0.01	0.1	<0.01	-	8.3	10.2	481
	1.5	1.7	<0.01	<0.01	<0.01	<0.01	0.1	8.3	10.4	491
12/14/1992	0	1.8	<0.01	<0.01	<0.01	<0.01	0.1	8.8	9.1	539
	1.5	1.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	9	541
1/20/1993	0	1.4	<0.01	-	<0.01	<0.01	0.1	8.9	5.3	363
	1.5	1.1	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	6.8	362
2/17/1993	0	1.2	0.5	<0.01	<0.01	0.1	<0.01	8.3	5.7	385
	1.5	1.2	0.4	<0.01	<0.01	0.1	<0.01	8.3	5.4	361
3/18/1993	0	1	0.3	<0.01	<0.01	<0.01	0.1	8.4	6.7	344
	1.5	1.4	0.3	<0.01	<0.01	<0.01	0.2	8.4	6.5	348
4/20/1993	0	<0.01	<0.01	<0.01	<0.01	<0.01	0.2	8.9	6.2	346
	2	<0.01	<0.01	<0.01	<0.01	<0.01	0.1	8.9	5.9	328
5/20/1993	0	1.8	<0.01	<0.01	<0.01	<0.01	0.2	9.2	5.9	322
	1.5	1.8	<0.01	<0.01	<0.01	<0.01	0.1	9.3	5.4	329
6/16/1993	0	1.2	<0.01	<0.01	<0.01	<0.01	0.1	9.0	7.7	356
	1.5	1.3	<0.01	<0.01	<0.01	<0.01	0.1	9.0	8	357

There were no stations in Lake Calabasas or its watershed in the Regional Board Water Quality Assessment Database. The Report (LARWQCB, 1996), however, states that DO was partially supporting the aquatic life use and that 92 measurements of dissolved oxygen were collected which ranged from

0.2 mg/L to 15.7 mg/L. pH was partially supporting the aquatic life use and not supporting the secondary drinking water standards. pH was measured 85 times, and values ranged from 7.4 to 9.3. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonia samples were collected ranging from non-detect to 0.45 mg/L. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Odor was listed as not supporting the contact and non-contact recreation uses. Eutrophication was not supporting the aquatic life use.

The city of Calabasas has been monitoring water quality in Lake Calabasas from a boat dock since 2004 at station CLBD. Table G-68 presents the monthly monitoring data collected through 2008. Nitrate concentrations have ranged from 0.04 mg/L to 1.6 mg/L; phosphate concentrations ranged from 0.03 mg/L to 0.77 mg/L. Secchi depths range from 0.5 m to greater than 2.7 m, and pH ranged from 7.91 to 9.69. Dissolved oxygen has been observed ranging from 4.8 mg/L to 15.82 mg/L with water temperatures ranging from 48.5 ºF to 90.8 ºF.

Table G-68. City of Calabasas 2004 to 2008 Monitoring Data

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (°F)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
1/20/2004	0.34	0.08	1.5	8.13	8.2	66.46	140	625	0.23
2/17/2004	0.30	0.11	1.8	8.01	8.12	65.23	135	640	0.28
3/24/2004	0.20	0.16	2.1	7.96	7.94	69.5	145	646	0.31
4/20/2004	0.15	0.20	2.4	7.91	7.71	70.5	145	652	0.45
5/19/2004	0.18	0.13	1.2	8.5	8.33	74.9	140	658	0.47
6/29/2004	0.23	0.11	0.5	9.07	9.8	78.08	140	661	0.5
7/28/2004	0.25	0.06	0.6	8.99	9.18	83.7	145	674	0.51
8/6/2004	0.23	0.19	1.1	8.12	9.8	81.18	145	694	0.52
9/28/2004	0.09	0.08	0.9	9.29	8.79	73.64	150	699	0.53
10/26/2004	0.16	0.03	0.9	9.29	11.29	64.79	145	633	0.48
11/24/2004	0.14	0.08	1.5	8.92	7.79	55.96	140	616	0.47
12/8/2004	0.48	0.09	1.8	7.91	8.74	49.69	140	717	0.49
1/11/2005	0.36	0.29	1.5	8.14	10.15	53.54	145	438	0.33
2/10/2005	0.36	0.07	2.1	8.83	11.26	56.22	145	445	0.34
3/31/2005	0.45	0.38	1.7	8.85	8.67	64.4	135	362	0.27
4/19/2005	0.50	0.20	1.2	8.98	8.59	68.03	140	402	0.3
5/3/2005	0.18	0.27	1.1	9.02	9.54	73.67	140	411	0.32
6/21/2005	0.48	0.07	0.8	8.85	7.95	79.77	145	451	0.34
7/27/2005	0.50	0.08	0.9	8.8	7.21	86.42	150	485	0.36
8/24/2005	0.34	0.05	0.8	8.81	11.45	81.82	155	521	0.39
9/19/2005	0.55	0.12	1.1	8.08	8.59	74.03	155	574	0.4

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (°F)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
10/10/2005	0.57	0.17	1.1	8.24	9.74	68	155	550	0.42
11/17/2005	0.59	0.09	0.8	8.64	14.2	62.96	155	551	0.42
12/22/2005	0.36	0.04	0.9	8.91	14.14	55.62	155	551	0.42
1/18/2006	0.66	0.05	0.9	9.1	10.86	52.9	155	501	0.39
2/6/2006	0.68	0.08	0.7	9.69	11.43	59.68	155	529	0.4
3/1/2006	0.52	0.15	0.8	9.32	8.48	56.31	145	539	0.41
4/26/2006	0.25	0.12	1.8	8.65	6.22	65.51	130	527	0.4
5/22/2006	0.66	0.06	1.8	9.03	6.18	74.46	130	546	0.41
6/22/2006	0.41	0.07	1.2	9.03	10.76	85.13	140	579	0.43
7/24/2006	1.61	0.12	1.1	8.95	7.38	90.82	140	615	0.46
8/16/2006	0.43	0.06	1.2	8.83	9.77	81.51	145	619	0.47
9/25/2006	0.48	0.06	1.2	9.57	10.41	76.44	145	647	0.49
10/18/2006	0.23	0.07	1.5	8.5	13.35	67.27	150	654	0.5
11/28/2006	0.50	0.06	0.6	9.38	6.69	58.46	150	653	0.5
12/27/2006	0.52	0.09	0.6	9.16	7.96	54.43	155	658	0.51
1/9/2007	0.61	0.12	0.8	8.92	6.57	50.01	160	661	0.51
2/22/2007	0.48	0.09	0.9	8.23	7.86	55.21	150	623	0.49
3/15/2007	0.18	0.12	1.2	8.03	4.96	68.6	145	593	0.45
4/26/2007	0.25	0.06	1.5	9.03	14.62	71.32	145	588	0.44
5/18/2007	0.30	0.77	1.4	8.84	8.13	71.56	150	610	0.46
6/22/2007	0.20	0.12	0.9	8.95	8.19	79.56	155	643	0.49
7/27/2007	0.27	0.05	0.9	9.17	7	82.78	160	661	0.5
8/17/2007	0.41	0.07	0.9	9.02	7.59	84.23	165	672	0.51
9/21/2007	0.39	0.07	0.9	8.96	8.67	76.7	155	643	0.49
10/5/2007	0.45	0.04	0.9	9.18	12	71.15	155	653	0.5
11/1/2007	0.41	0.04	1.5	8.79	6.06	67.72	155	660	0.5
12/19/2007	0.39	0.04	1.5	8.97	4.8	52.79	155	653	0.5
1/31/2008	0.43	0.03	1.8	9.53	11.78	54.35	140	527	0.4
2/29/2008	0.27	0.08	1.8	9.2	10.1	58.7	135	526	0.4
3/7/2008	0.32	0.47	1.5	9.07	15.82	62.02	130	524	0.4
4/8/2008	0.25	0.12	1.8	8.76	9.23	66.71	140	551	0.42

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (℉)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
5/6/2008	0.30	0.14	>2.7	8.48	6.05	70.95	145	588	0.45
6/19/2008	0.16	0.14	1.8	9	12.64	84.9	150	649	0.49
7/10/2008	0.41	0.18	1.8	8.97	8.88	86.3	145	658	0.49
8/21/2008	0.09	0.09	1.2	9.45	9.48	80.63	150	680	0.51
9/11/2008	0.07	0.11	1.2	9.25	8.6	79.53	155	699	0.53
10/23/2008	0.09	0.23	1.2	9.22	12.22	66.78	160	711	0.54
11/26/2008	0.11	>0.65	2.1	8.95	12.1	59.7	155	702	0.54
12/19/2008	0.05	0.18	2.1	8.82	11.76	48.56	150	695	0.52

The Regional Board sampled Lake Calabasas from three shoreline sites on January 15, 2009. Data are presented in Table G-69.

Table G-69. Analytical Data for the January 15, 2009 Lake Calabasas Sampling Event

Station	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total Phosphate (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)
CL1	0.391	1.92	0.154	<0.1	<0.4	<0.5	479	<10
CL1 (duplicate)	0.431	1.64	0.157	<0.1	<0.4	<0.5	623	<10
CL2	0.453	1.79	0.192	<0.1	<0.4	<0.5	610	<10
CL3	0.42	1.7	0.164	<0.1	<0.4	<0.5	622	<10

Total depth and Secchi depth were not measured during this event. Dissolved oxygen concentrations were not measured either. Temperature ranged from 11 ºC to 12 ºC and pH ranged from 8.2 to 8.7 (Table G-70).

Table G-70. Field Data for the January 15, 2009 Lake Calabasas Sampling Event

Site	Time	Depth (m)	Temp (C)	pH
CL1	10:20	surface	12.0	8.7
CL2	11:25	Surface	11.5	8.2
CL3	12:05	surface	11.0	8.3

USEPA sampled Lake Calabasas from two in-lake sites and the potable water input (CL-PW) on August 6, 2009 (Table G-71). In-lake ammonia concentrations were less than or equal to 0.03 mg-N/L; TKN ranged from 1.17 mg-N/L to 1.23 mg-N/L. Nitrate and nitrite samples were less than the detection limit of 0.01 mg-N/L. Orthophosphate ranged from 0.0129 mg-P/L to 0.0453 mg-P/L and total phosphorus ranged from 0.152 mg-P/L to 0.221 mg-P/L. Chlorophyll *a* ranged from 35 to 81 µg/L.

Table G-71. Analytical Data for the August 6, 2009 Lake Calabasas Sampling Event

Station	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	TP (mg/L)	Chlorophyll *a* (ug/L)	Suspended Solids (mg/L)
CL-4	<0.03	1.227	<0.01	<0.01	0.0452	0.221	35.35	8.65
CL-5	0.03	1.166	<0.01	<0.01	0.0129	0.152	81.40	8.00
CL-PW	0.35	0.464	1.13	<0.01	0.0208	<0.016	NA	<0.5

Temperature and dissolved oxygen were not measured in the lake during this event. The total depth at CL-4 was 2.51 m and the depth at CL-5 was 2.06 m. The Secchi readings at CL-4 and CL-5 were 0.737 m and 0.660 m, respectively. Field data for the in-lake sites are presented in Table G-72. Field data were collected for the potable water source on August 6, 2009. After purging the line for approximately 10 minutes, the pH was 7.93 and the temperature was 18.2 ºC.

Table G-72. Field Data for the August 6, 2009 Lake Calabasas Sampling Event

Site	Time	Depth (m)	Secchi Depth (m)
CL-4	9:15	2.51	0.737
CL-5	9:40	2.06	0.660

Profile data were collected at Stations CL-4 and CL-5 on the morning of August 6, 2009 between 9:00 and 9:50. The depth at CL-4 was 2.51 meters, and the Secchi depth was 0.74 meters. The temperature in the lake ranged from 25.6 and 26.1°C. The specific conductivity was constant with depth, around 1.22 mS/cm. The DO ranged from 6.37 to 7.56 mg/L and pH ranged from 7.57 to 8.77. These profile data are shown in Figure G-15.

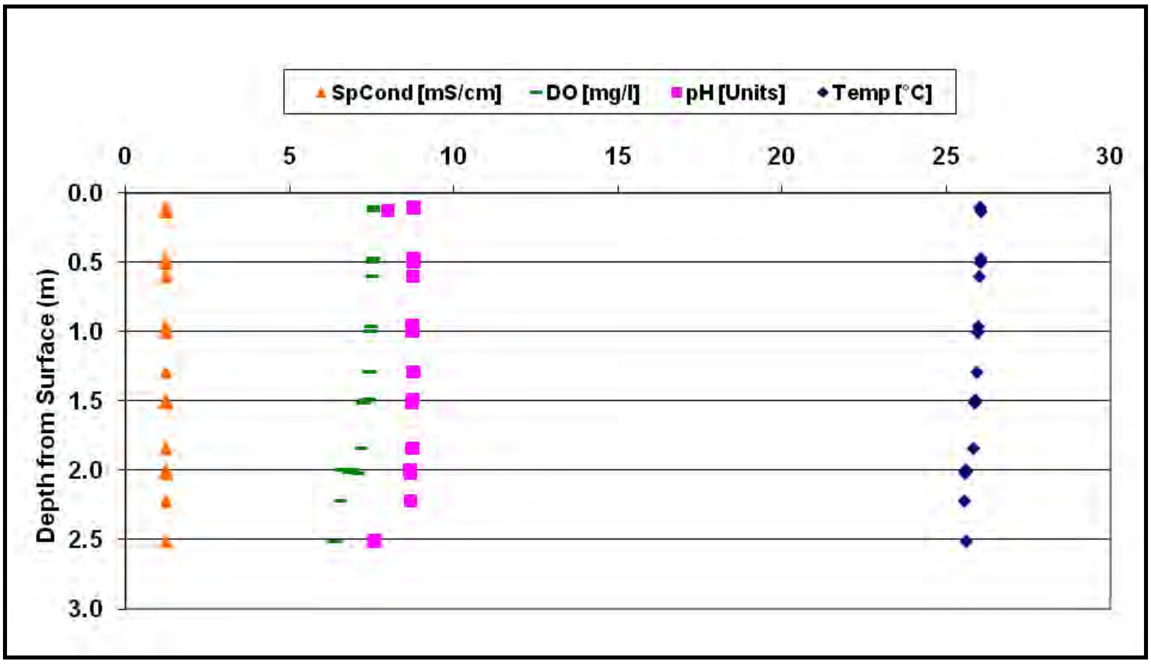

Figure G-15. Profile Data Collected in Lake Calabasas at CL-4 on August 6, 2009

The profile data collected at CL-5 is shown in Figure G-16. The temperature at this station was between 25.82 and 26.45°C. The pH ranged from 9.04 to 9.20. Dissolved oxygen ranged from 8.71 to 9.74 mg/L. The conductivity was between 1.04 to 1.05 mS/cm. The field team observed that this location was close to the tap water inlet and likely affected the conductivity levels, which were lower than those at CL-4. The depth at this station was 1.75 meters and the Secchi depth was 0.66 meters.

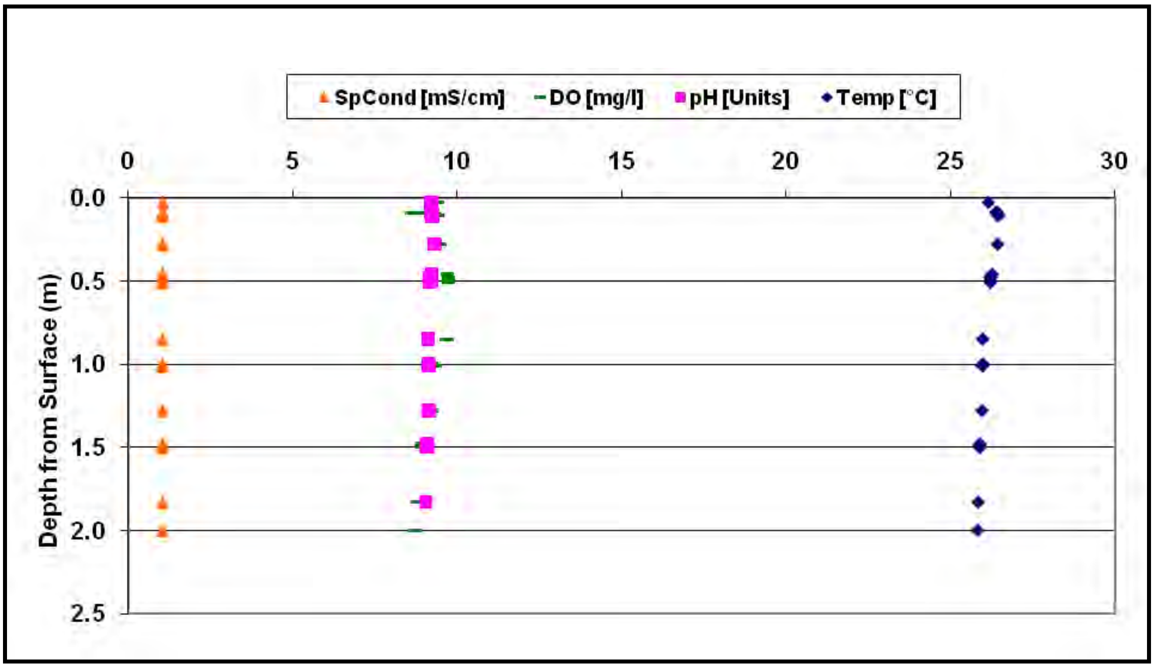

Figure G-16. Profile Data Collected in Lake Calabasas at CL-5 on August 6, 2009

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G.8 Monitoring Data for El Dorado Park Lakes

Monitoring data relevant to the impairments of El Dorado Park lakes are available from 1992, 1993, and 2008 through 2010. In addition, fish tissue data are available for 1991, 1992, 1998, and 2007. Figure G-17 shows the historical and recent monitoring locations for El Dorado Park lakes.

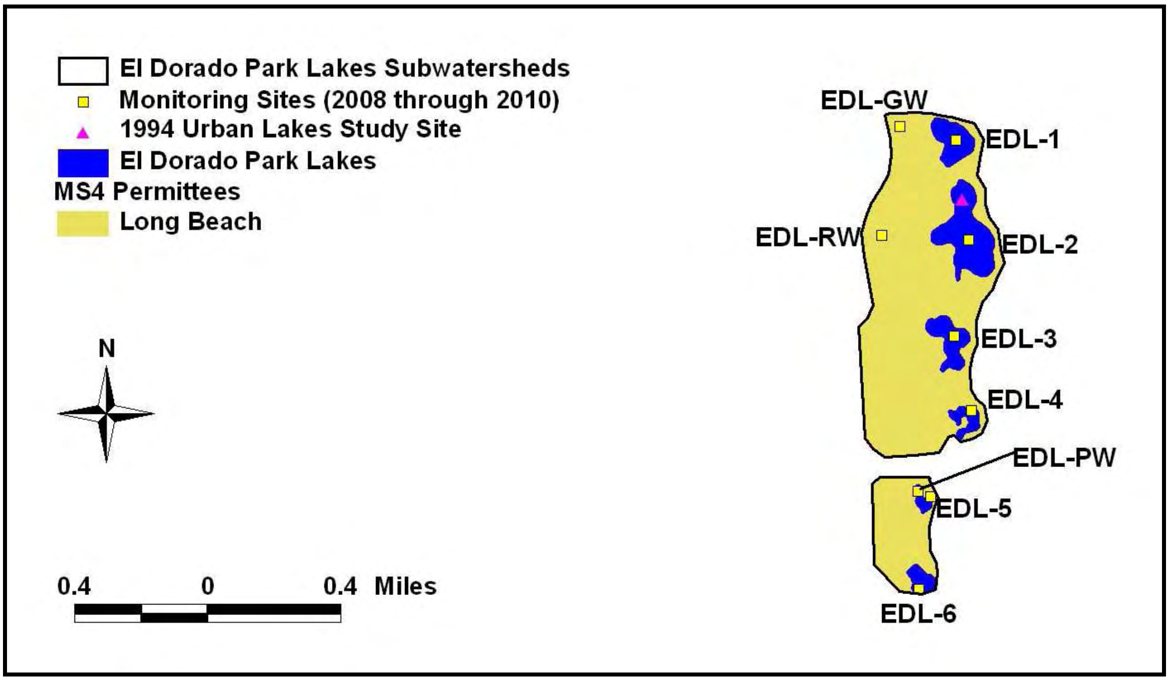

Figure G-17. Monitoring Sites in the El Dorado Park Lakes Watershed

G.8.1 Monitoring Related to Nutrient Impairments

The El Dorado Parks lakes were included in the 1992/1993 sampling effort to support the Urban Lakes Study (Table G-73). Data were collected from the north end of Lake 2 shown in Figure G-17 (pink triangle). TKN concentrations ranged from 1.2 mg/L to 4.2 mg/L. Nineteen of 45 samples for ammonium were less than the reporting limit, and the maximum concentration observed was 1.9 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming the analysis methodology converted all ammonia to ammonium). Nitrite samples were consistently less than the reporting limit, as were the majority of nitrate concentrations. Measurable amounts of nitrate were only observed in January and February of 1993 when concentrations ranged from 0.1 mg/L to 0.3 mg/L. Orthophosphate concentrations ranged from 0.2 mg/L to 0.9 mg/L, and total phosphorus ranged from 0.3 mg/L to 1.1 mg/L. pH ranged from 8.2 to 9.4, and TOC ranged from 7.1 mg/L to 10.7 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 µg/L to 133 µg/L with an average of 48 µg/L.

Although the 1996 Water Quality Assessment Database does not contain monitoring data for the El Dorado Park lakes, the summary table in the Report does include a synopsis of monitoring data and related impairments. pH was listed as partially supporting the aquatic life use and not supporting the contact recreation use: 116 measurements of pH were collected with values ranging from 6.9 to 9.4. Ammonium was not supporting the aquatic life or contact recreation uses; 45 ammonium samples were collected with concentrations ranging from non-detect to 1.92 mg/L , the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Algae were listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

Table G-73. El Dorado Park Lakes 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/16/1992	0	1.5	0.2	<0.01	<0.01	0.2	0.3	9.2	9.6	461
	3	1.4	<0.01	<0.01	<0.01	0.2	0.4	9.2	10.7	459
	4.5	4.2	1.9	<0.01	<0.01	0.9	1.1	8.4	10.3	470
7/16/1992	0	1.6	<0.01	<0.01	<0.01	0.2	0.3	9.3	10.3	459
	3	1.5	<0.01	<0.01	<0.01	0.2	0.4	9.2	10	471
	4.5	1.9	<0.01	<0.01	<0.01	0.3	0.5	9.2	9.3	476
7/16/1992	0	1.5	<0.01	<0.01	<0.01	0.2	0.3	9.1	9.7	488
	3	1.5	0.1	<0.01	<0.01	0.2	0.3	9.2	9.9	449
	4.5	1.7	<0.01	<0.01	<0.01	0.3	0.5	9.2	9.9	475
8/20/1992	0	1.55	<0.01	<0.01	<0.01	0.2	0.3	9.3	10.3	461
	2	1.6	<0.01	<0.01	<0.01	0.2	0.3	9.4	10.4	475
	4	2.3	0.5	<0.01	<0.01	0.3	0.4	9.2	10.7	466
9/24/1992	0	1.5	<0.01	<0.01	<0.01	0.2	0.3	9.3	9.9	442
	2	1.4	<0.01	<0.01	<0.01	0.3	0.3	9.3	9.6	443
	4	1.8	0.3	<0.01	<0.01	0.3	0.4	8.9	9.4	445
10/20/1992	0	2.4	0.3	<0.01	<0.01	0.3	0.5	9.2	10.2	435
	2	2	0.3	<0.01	<0.01	0.4	0.5	9.1	10.1	474
	3.5	2.3	0.3	<0.01	<0.01	0.4	0.5	9.1	10.5	493
11/12/1992	0	2.3	0.5	<0.01	<0.01	0.4	0.5	9	9.9	450
	2.5	2.4	0.4	<0.01	<0.01	0.4	0.5	9	10.3	450
	3.5	2.1	0.5	<0.01	<0.01	0.4	0.4	9.1	9.4	450
12/15/1992	0	3.4	1.5	<0.01	<0.01	0.6	0.5	8.5	8.7	449
	2.5	3.3	1.5	<0.01	<0.01	0.6	0.5	8.5	10.1	452
	3.5	NA	1.5	<0.01	<0.01	0.6	0.6	8.5	8.5	451
1/21/1993	0	2.4	1.1	<0.01	0.3	0.4	0.5	8.3	7.4	380
	2.5	2.4	1.1	<0.01	0.3	0.4	0.4	8.3	7.2	417
	3.5	2.3	1.2	<0.01	0.3	0.4	0.4	8.3	7.1	417
2/10/1993	0	2	<0.01	<0.01	0.2	0.3	0.5	8.9	8.3	407

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	2.5	2.5	<0.01	<0.01	0.1	0.3	0.4	8.9	8.4	496
	3.5	2.1	<0.01	<0.01	0.2	0.3	0.4	8.9	7.7	439
3/8/1993	0	3.7	0.1	<0.01	<0.01	0.3	0.4	9.4	9.6	419
	1.5	4	<0.01	<0.01	<0.01	0.2	0.4	8.8	9.6	423
	2.5	2.2	0.1	<0.01	<0.01	0.2	0.4	8.6	9.4	413
	3.5	1.5	0.1	<0.01	<0.01	0.3	0.4	8.2	7.7	407
4/8/1993	0	1.4	<0.01	<0.01	<0.01	0.3	0.3	9	7.8	413
	1.5	1.3	<0.01	<0.01	<0.01	0.3	0.3	9	7.8	431
	2.5	1.2	<0.01	<0.01	<0.01	0.3	0.3	9	7.6	429
	3.5	1.5	0.3	<0.01	<0.01	0.4	0.3	8.6	7.4	412
5/12/1993	0	1.3	<0.01	<0.01	<0.01	0.3	0.3	8.8	8	459
	2.5	1.2	<0.01	<0.01	<0.01	0.3	0.3	8.8	7.6	460
	3.5	1.5	0.1	<0.01	<0.01	0.3	0.3	8.8	8.2	450
6/15/1993	0	1.9	0.3	<0.01	<0.01	0.4	0.4	9.2	10	468
	1.5	2	0.3	<0.01	<0.01	0.4	0.4	9.2	9.5	487
	2.5	2.1	0.3	<0.01	<0.01	0.4	0.3	9.2	9.1	478
	3.5	1.6	0.3	<0.01	<0.01	0.4	0.4	9	8.6	465

In May 2008, Marine Biochemists sampled water quality in the El Dorado Park lakes system. The data report does not specify who sponsored the sampling. On May 8, 2008 water quality data were collected in the upper four lakes (Table G-74). DO concentrations ranged from 7.36 mg/L to 8.63 mg/L, and pH ranged from 7.37 to 8.76. Temperature was fairly consistent in all four lakes and was approximately 69 ºF. The concentrations of nitrates were highly variable and ranged from 0.3 mg/L to 3.0 mg/L; phosphates ranged from 0.09 mg/L to 0.58 mg/L. It is not clear from the report if the units on the nitrate samples were “as N” or “as NO_3” or if the units on the phosphate samples were “as P” or “as PO_4.” Sampling depth and time and analysis methodologies were not included with the hard copy data report Tetra Tech received, nor were sampling locations.

Table G-74. May 2008 Water Quality Monitoring Data for El Dorado Park Lakes

Lake Number	Dissolved Oxygen (mg/L)	pH	Temperature (ºF)	Nitrates (mg/L)	Phosphates (mg/L)
1	8.63	7.37	69.09	3.0	0.09
2	7.76	8.76	68.84	0.9	0.13
3	7.36	7.94	69.11	0.3	0.19
4	7.90	8.32	69.42	1.5	0.58

The El Dorado Park lakes were sampled February 26, 2009 and July 15, 2009 by USEPA and the Regional Board. The field notes from the event indicate that the top four lakes are supplied primarily by groundwater. Water flows from Lake 1 to 2 to 3 to 4; excess water is pumped out of Lake 4 and discharged to a storm drain. Lakes 5 and 6 are not naturally or artificially connected to Lakes 1 through 4; they are connected to each other. Water is supplied to these two lakes by a pipe that continuously discharges potable water to Lake 5 (Valentina Cabrera-Stagno, USEPA Region IX, personal communication, February 3, 2009). Lakes 1, 2, 5, and 6 were sampled in February and Lakes 1, 2, 3, 4, and 6 were sampled during the July monitoring event (nutrients were not analyzed at Lake 4). Lakes 4, 5, and 6 were treated with algaecides in mid-June (personal communication, Ed Gahafer, July 15, 2009), which may have reduced chlorophyll *a* concentrations during the July sampling event.

Table G-75 presents the in-lake water quality measurements for the February and July 2009 sampling events. During the February event, Lakes 1 and 2 were sampled from a depth of 0.76 meters and the total depth at each station was approximately 4.4 meters. The Secchi depth in Lake 1 was 1.31 meters and in Lake 2 was 1.37 meters. Lake 5 was sampled from a depth of 0.46 meters and the total depth at the sampling location was approximately 4.5 meters; Secchi depth was not measured at this site. Lake 6 was sampled from a depth of 0.92 meters and the total depth at the sampling location was 2.5 meters; the Secchi depth was 1.83 meters. The main source of water to Lakes 1 through 4, water pumped from groundwater, was also sampled. These data are also included for comparison as the nutrient loading from this source may be significant relevant to the upland sources represented by the LSPC/EMC approach (Appendix D).

During the July event, Lake 1 was sampled from a depth of 0.58 meters. The total depth at the Lake 1 station was 3.5 meters, and the Secchi depth was 1.17 meters. Lake 2 was sampled at 0.33 meters below the surface. The Secchi depth at Lake 2 was 0.69 meters and the total lake depth was 3.9 meters. Lake 6 was sampled at a depth of 0.97 meters. The total depth of Lake 6, as measured by the sampling probe, was 2.2 meters. Samples take at Lake 5 were approximately 0.46 meters below the surface. The total depth and Secchi depth were not measured at Lake 5 during this monitoring event. The Secchi depth reading at Lake 6 was 1.96 meters. The groundwater and potable water were also sampled during this event. The results of these efforts are shown in Table G-75. Temperature, DO, pH, and conductivity were not measured during either of the monitoring events.

Table G-75. 2009 In-lake Water Column Measurements for the El Dorado Park Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2/26/2009	EDL-1	8:30	1.8	<0.03	<0.01	<0.01	0.028	0.160	48.7	1.31
	EDL-2	11:15	2.1	0.03	<0.01	<0.01	0.016	0.094	19.2	1.37
	EDL-2 (dup.)	11:30	2.2	<0.03	<0.01	<0.01	0.016	0.102	18.7	1.37
	EDL-5	12:15	1.1	<0.03	<0.01	<0.01	0.015	0.030	5.3	NA
	EDL-6	13:20	1.1	0.03	<0.01	<0.01	0.016	0.031	5.9	1.83
	EDL-GW[1]	9:40	0.84	0.33	<0.01	<0.01	0.074	0.190	NA	NA
7/15/2009	EDL-1	11:15	0.91	<0.03	<0.01	<0.01	<0.0075	0.047	22.9	1.17
	EDL-2	9:40	1.0	<0.03	<0.01	<0.01	<0.0075	0.1605	39.38	0.69
	EDL-2D	9:40	0.84	<0.03	NS	NS	NS	0.151	NS	0.69

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
	EDL-5	15:10	NS	0.1	<0.01	0.12	<0.0075	0.139	1.3	NA
	EDL-6	15:50	0.98	0.04	<0.01	0.09	<0.0075	0.138	6.2	1.96
	EDL-GW[1]	11:30	1.1	0.28	<0.01	<0.01	0.07095	0.291	NA	NA
	EDL-PW[1]	14:40	0.84	0.365	<0.01	0.37	<0.0075	0.1085	NA	NA

[1] EDL-GW represents the groundwater input to lakes 1 through 4 and EDL-PW represents the potable water input to Lake 5. These are not in-lake samples.

Additional water quality samples were collected from the El Dorado Park lakes. Table G-76 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Lakes 1, 2, 5, and 6 during both monitoring events. Duplicate samples were not collected at Lake 2 in the July 2009 monitoring. Lakes 3 and 4 were only measured for hardness in July. The July solids monitoring data were reported percent solids for EDL-1, EDL-2, and EDL-6, while EDL-5 had TSS data reported in mg/L. Again, measurements collected from the groundwater source and potable water are included for comparison to in-lake samples.

Table G-76. Supplemental Water Quality Monitoring for In-lake Samples in the El Dorado Park Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/26/2009	EDL-1	8:30	33.70	25.04	203	100.2	380	4.6	5.2	4.1
	EDL-2	11:15	80.92	30.02	274	113.4	532	4.1	9.3	7.1
	EDL-5	12:15	55.65	88.82	126	NA	406	2.6	5.0	3.9
	EDL-6	13:20	56.07	88.83	126	NA	370	3.5	5.0	6.2
	EDL-GW[1]	9:40	18.20	22.10	194	131.4	304	0.7	0.7	0.4
7/15/2009	EDL 1	11:15	25.56	20.47	210	117.3	350	24 (%)	4.3	4.8
	EDL 2	09:40	80.505	33.28	274	122.5	532	24 (%)	9.2	11.3
	EDL 2D	09:40	81.64	33.71	280	NS	NS	NS	9.1	NS
	EDL 3	13:35	NS	NS	NS	88.1	NS	NS	NS	NS
	EDL 4	14:30	NS	NS	NS	87.4	NS	NS	NS	NS
	EDL 5	15:10	57.77	82.7	120	85.6	388	2.15	3.7	4.1
	EDL 6	15:50	59.25	87.6	118	84.6	400	12.5 (%)	9.9	4.8
	EDL GW[1]	11:30	15.6	20.2	210	155.9	356	NS	0.2	0.4
	EDL PW[1]	14:40	51.53	52.245	116	81.65	345	NS	1.4	1.3

[1] EDL-GW represents the groundwater input to lakes 1 through 4 and EDL-PW represents the potable water input to Lake 5. These are not in-lake samples.

Profile data were also collected for specific conductivity, dissolved oxygen, pH, and temperature in each of the monitored lakes. Figure G-18 through Figure G-21 show the profile data collected on February 26, 2009 at Stations EDL-1, EDL-2, EDL-5, and EDL-6, respectively. Specific conductivity is constant with depth at each station. DO decreases from 5.1 mg/L to 8.7 mg/L near the surface to approximately 3.5 mg/L at the bottom of each lake. pH ranges from 7.3 to 8.4, and temperature ranges from 13.8 ºC to 17.6 ºC at each station.

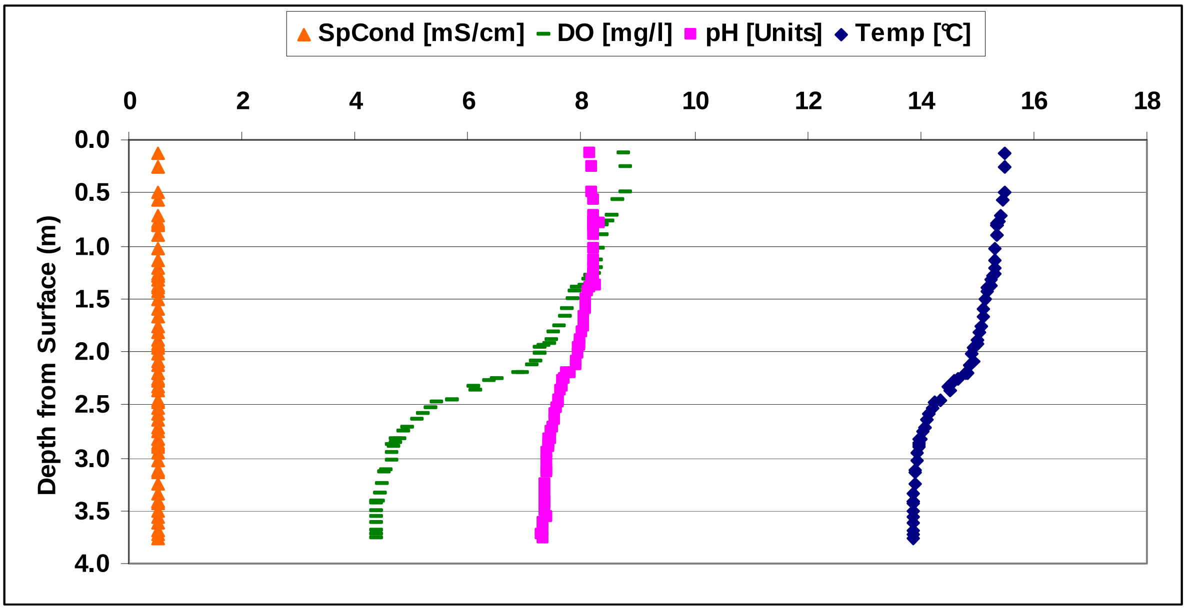

Figure G-18. Profile Data Collected in El Dorado Park Lake 1 on February 26, 2009

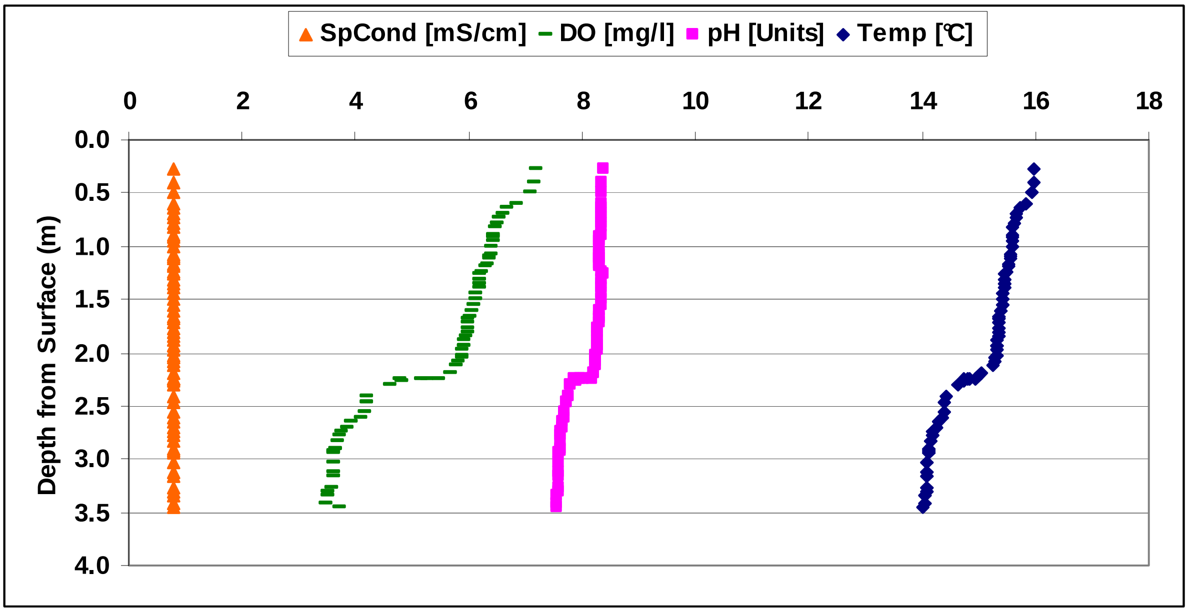

Figure G-19. Profile Data Collected in El Dorado Park Lake 2 on February 26, 2009

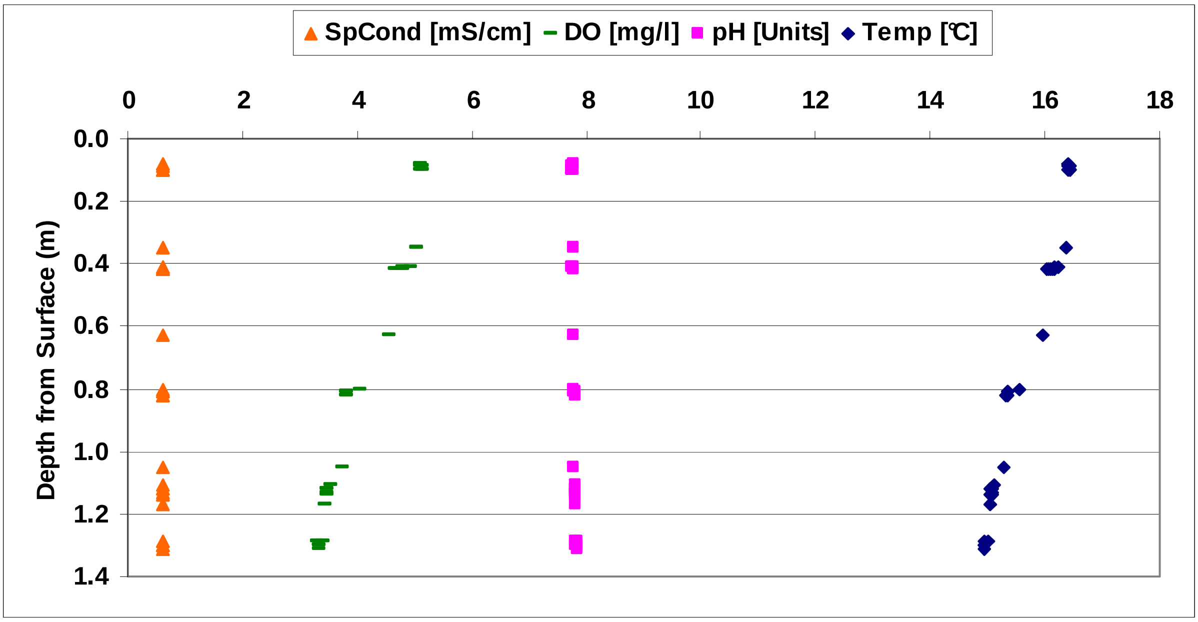

Figure G-20. Profile Data Collected in El Dorado Park Lake 5 on February 26, 2009

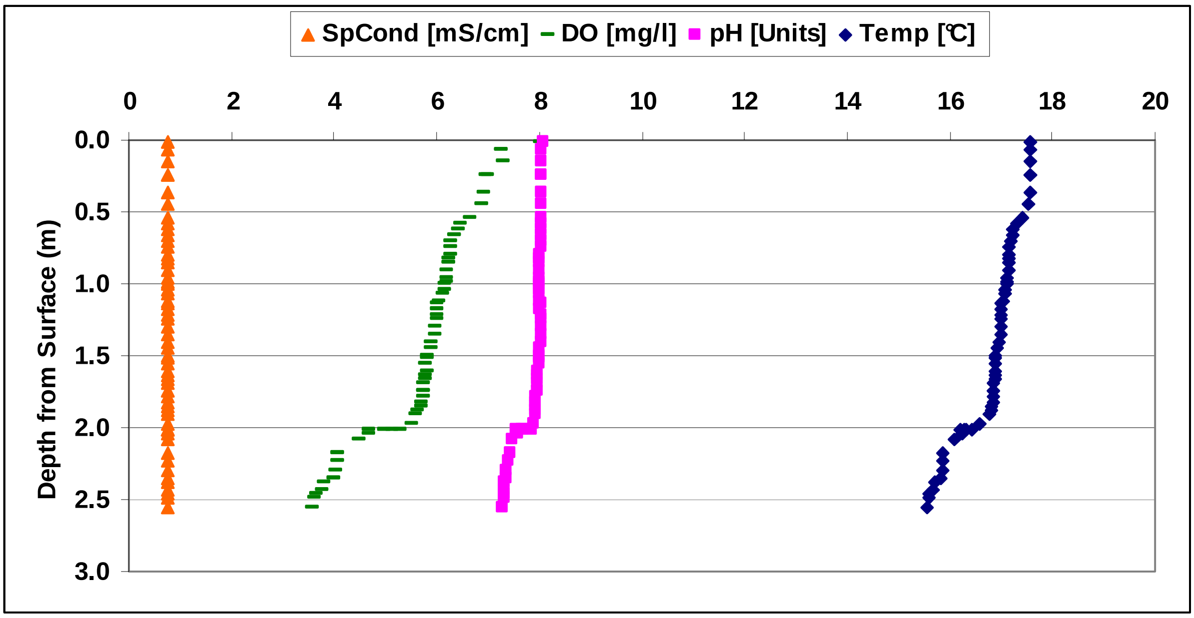

Figure G-21. Profile Data Collected in El Dorado Park Lake 6 on February 26, 2009

Profile data were also collected on July 15, 2009 for Stations EDL-1, EDL-2, and EDL-6. Summer temperatures range from 23.6 to 30.2 ºC at each station. The summer pH range is similar to the winter pH range, from 7.2 to 8.4. The DO ranges from 1.65 mg/L near the bottom of the lakes and up to 9.57 mg/L near the surface of the lakes. Specific conductivity is constant with depth at each station. The July profile data are displayed in Figure G-22 through Figure G-24.

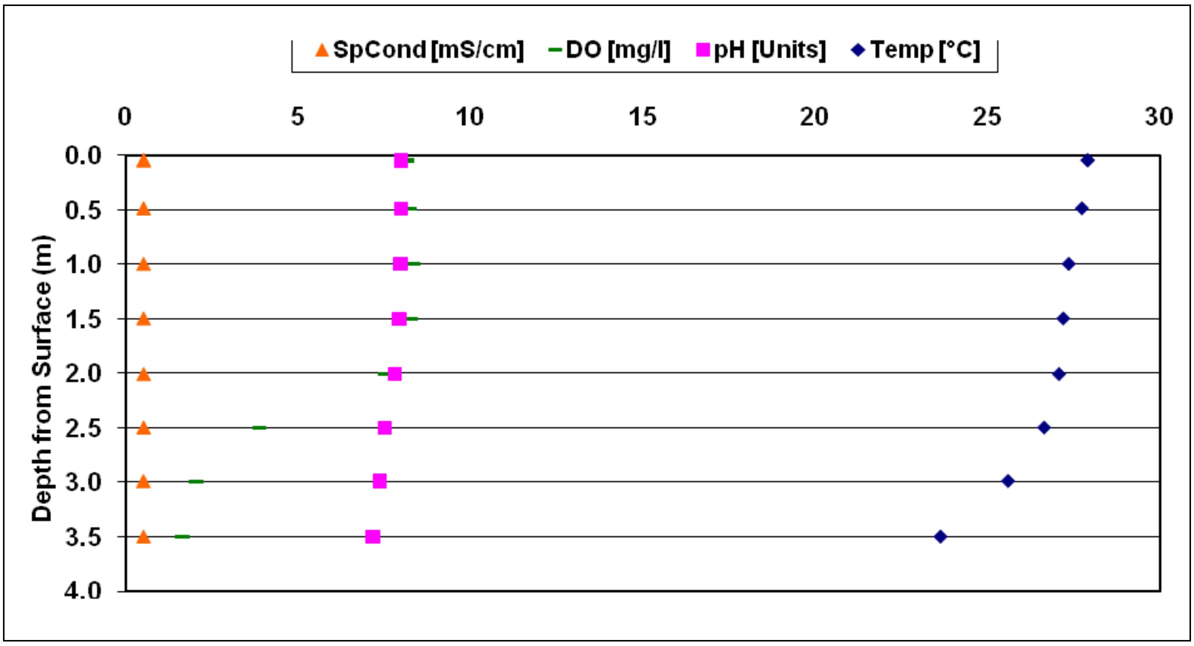

Figure G-22. Profile Data Collected in El Dorado Park Lake 1 on July 15, 2009

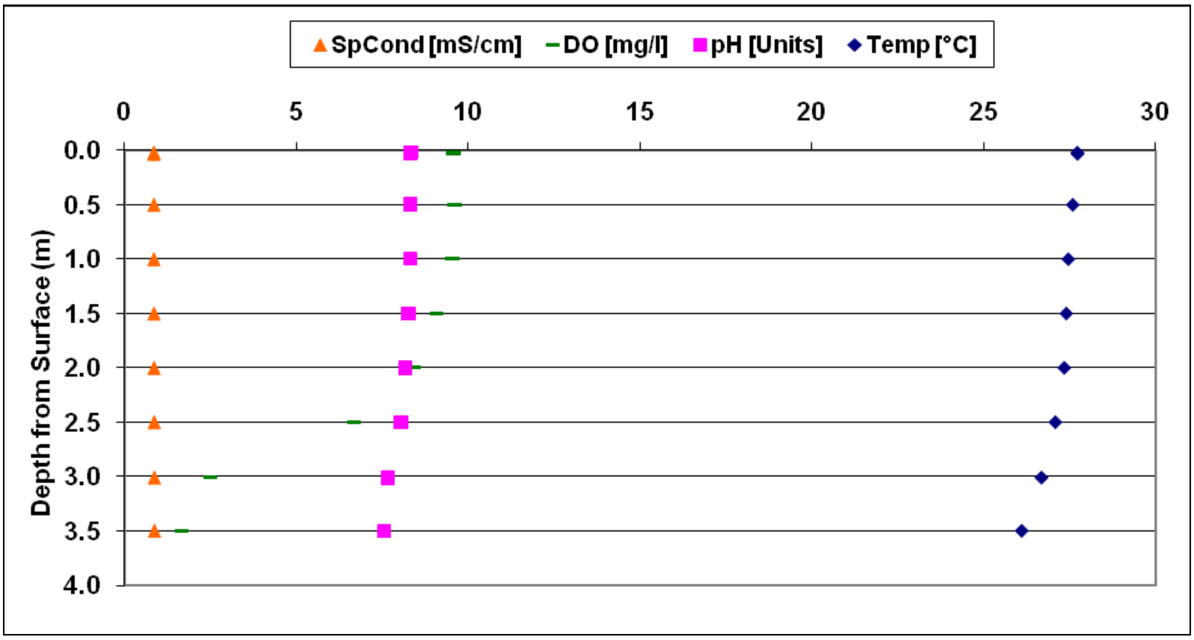

Figure G-23. Profile Data Collected in El Dorado Park Lake 2 on July 15, 2009

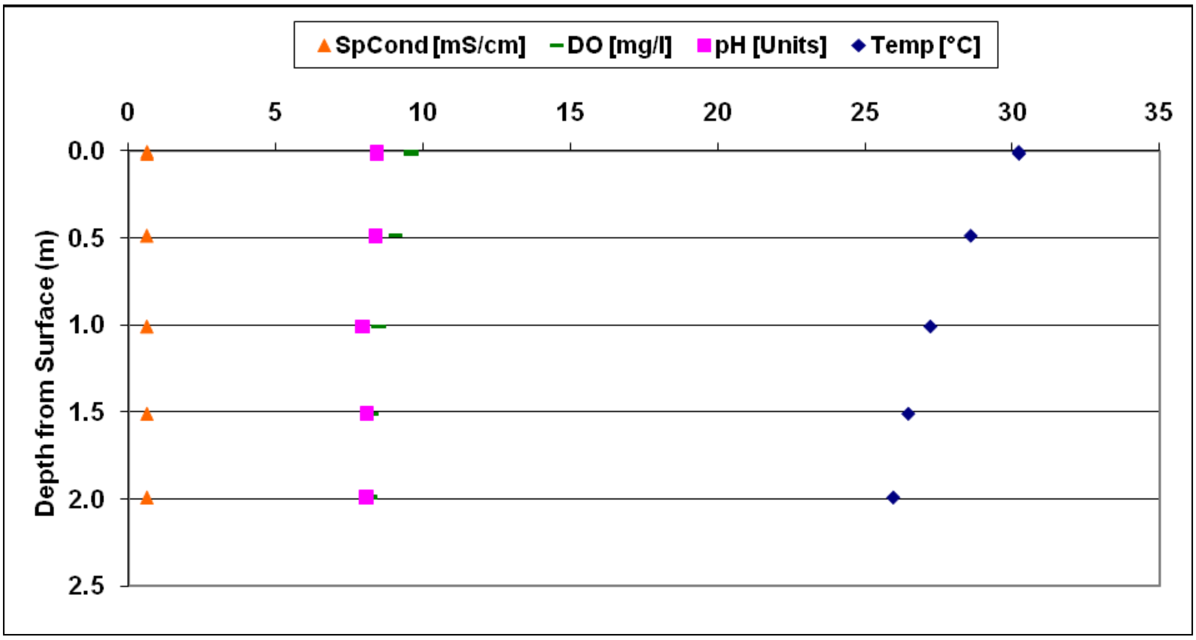

Figure G-24. Profile Data Collected in El Dorado Park Lake 6 on July 15, 2009

Field data were also collected at the potable water source at El Dorado Park during the July sampling event. At 14:40, the temperature was 27 ºC (pH measurements were not taken with a faulty meter and were not considered reliable).

Reclaimed water, used as irrigation on land surrounding the lake, was also sampled on December 1, 2009. Table G-77 presents the December 1, 2009 sampling results collected at EDLRW and ELDRWD (duplicate for ELDRW). In general, total phosphorus averaged 0.166 mg-P/L, and total nitrogen averaged 5.74 mg-N/L. EDLRW was also monitored for chloride, sulfate, alkalinity, hardness, total dissolved solids, dissolved organic carbon, and total organic carbon; results are presented in Table G-78.

Table G-77. Reclaimed Water Measurements for the El Dorado Park Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)
12/1/2009	EDLRW	14:30	1.30	0.61	0.05	4.04	0.07	0.164
	EDLRWD (dup.)	14:30	1.15	0.63	0.05	4.9	0.10	0.168

Table G-78. Supplemental Water Quality Monitoring for Reclaimed Water at the El Dorado Park Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
12/1/2009	EDLRW	14:30	110.53	79.62	198	133.1	583	5.8	5.7

Field data were also collected at shoreline stations in El Dorado Park during the December 1, 2009 sampling event. At EDL-2S (total depth 1 foot) at 1:45 p.m., the temperature was 17.01 ºC and the pH was 8.31. EDL-1S was sampled at 2:15 pm (total depth 2 feet) with a temperature of 16.92 ºC and a pH of 8.23. Temperature at 3:40 p.m. at EDL-6S (total depth 1.5 feet) was 15.34 ºC and the pH was 8.12, while temperature was 14.94 ºC and pH was 8.17 at EDL-5S (total depth 2 feet) about 15 minutes later. The last two sites (EDL-4S [total depth 1 foot] and EDL-3S [total depth 2 feet]) both had a pH reading of 9.20 at 4:10 p.m. and 4:20 p.m., respectively. Temperatures at these sites were 15.92 ºC and 14.71 ºC, respectively.

The southern two lakes at El Dorado Park were resampled for nutrients on August 10, 2010. Table G-79 summarizes the nutrient data collected in each lake as well as the potable water source. TKN concentrations ranged from 0.67 to 1.03 mg-N/L. Ammonia concentrations ranged from 0.03 mg-N/L to 0.05 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.23 mg-N/L to 0.24 mg-N/L. Orthophosphate ranged from 0.022 mg-P/L to 0.027 mg-P/L, and total phosphorus ranged from 0.027 mg-P/L to 0.038 mg-P/L. Chlorophyll *a* ranged from 4.81 µg/L to 6.23 µg/L.

Table G-79. August 10, 2010 In-lake Water Column Measurements for the Nature Center Lakes at El Dorado Park

Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
EDL-5	13:00	1.03	0.0493	0.051	0.244	0.027	0.038	6.23	>1.25
EDL-6	15:00	0.67	0.0328	0.052	0.233	0.022	0.0271	4.81	1.5
EDL-PW	13:40	0.48	0.0359	0.054	0.173	0.026	<0.0165	<1.2	NA

During the August 2010 event, two continuous monitoring probes were deployed in each southern lake over a 24-hour period at depths of about 0.7 to 1.3 meters below the surface. DO concentrations ranged from 8.3 mg/L to 9.5 mg/L in Nature Center North Lake (Figure G-25) and from 9.5 mg/L to 12.6 mg/L in Nature Center South Lake (Figure G-26). pH ranged from 8.5 to 9.0 in both lakes.

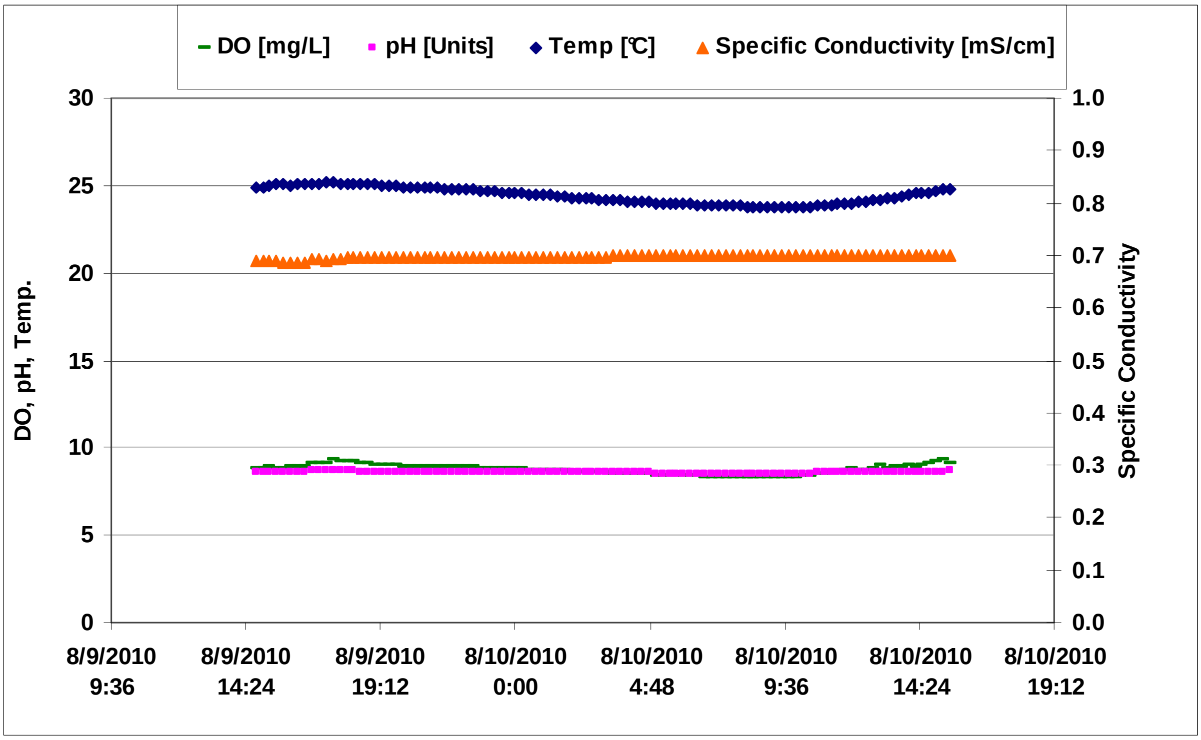

Figure G-25. 24-Hour Probe Data Collected in El Dorado Park Lake 5 on August 9, 2010

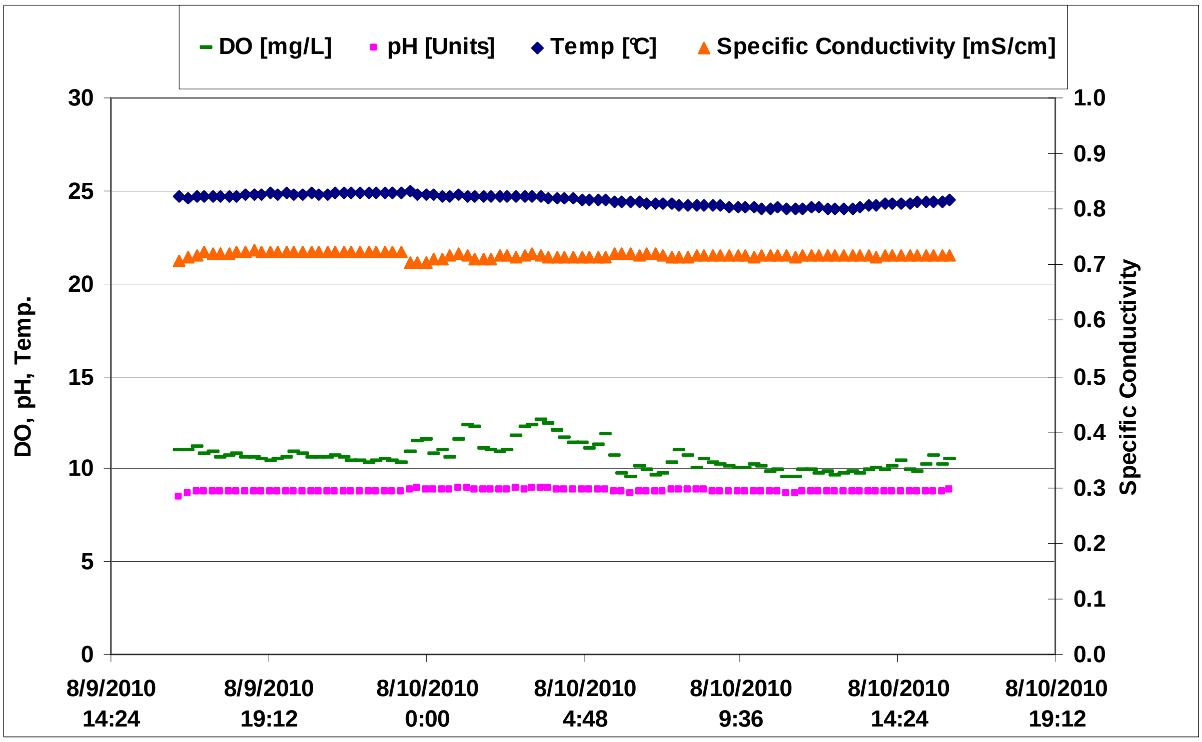

Figure G-26. 24-Hour Probe Data Collected in El Dorado Park Lake 6 on August 9, 2010

On August 10, 2010, depth-profile data were also collected during this water sampling event. Table G-80 summarizes the depth-profile data collected at ELD-5 and ELD-6. DO measurements collected from the surface to 0.3 meters above the bottom of Nature Center North Lake ranged from 8.4 mg/L to 8.5 mg/L. In Nature Center South Lake, DO ranged from 11.8 mg/L at the surface to 9.9 mg/L at 0.3 meters above the bottom of the lake. Figure G-27 and Figure G-28 show the profile data collected on August 10, 2010, 2010 at stations ELD-5 and ELD-6 respectively.

Table G-80. Profile Data Collected in the Nature Center Lakes at El Dorado Park on August 10, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
EDL-5	12:52	0.1	24.48	8.59	8.40	0.707	162
		0.55	24.16	8.53	8.44	0.706	164
		1.02	24.04	8.57	8.53	0.705	161
EDL-6	14:39	0.11	27.22	8.95	11.78	0.707	279
		0.54	25.26	8.75	11.08	0.713	268
		1.04	24.63	8.60	10.28	0.715	265
		1.49	24.24	8.55	9.89	0.713	262
		2.03	23.57	8.60	9.96	0.712	259

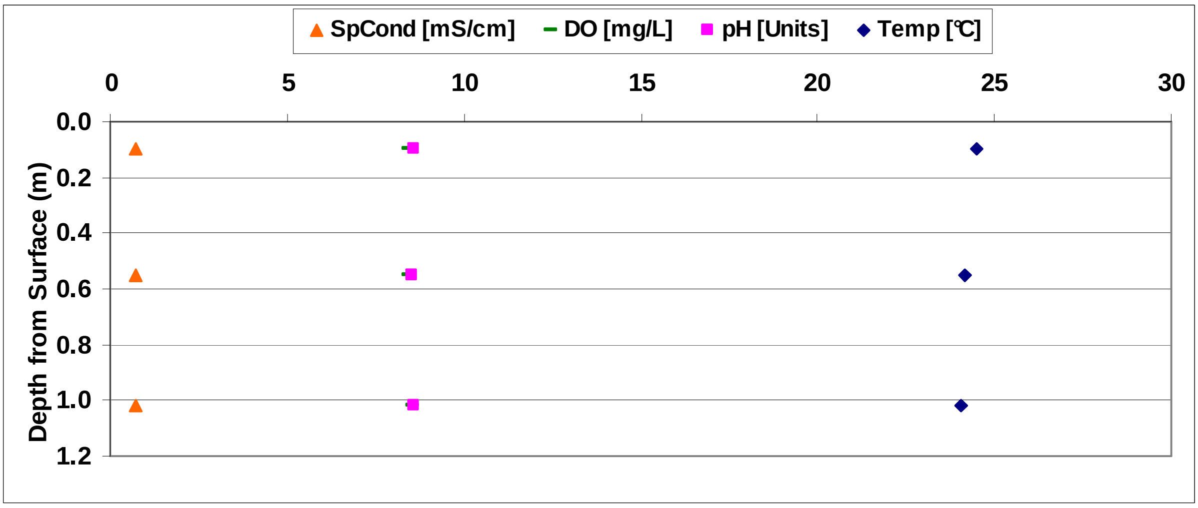

Figure G-27. Profile Data Collected in El Dorado Park Lake 5 on August 10, 2010

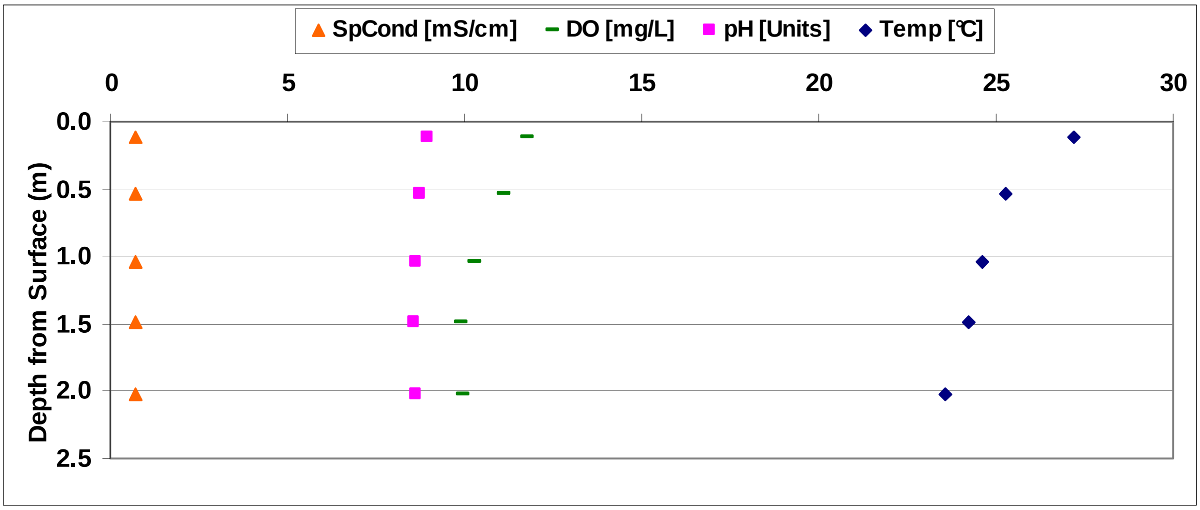

Figure G-28. Profile Data Collected in El Dorado Park Lake 6 on August 10, 2010

Sediment samples were also collected during the August 2010 monitoring event. Table G-81 summarizes these data.

Table G-81. August 10, 2010 Sediment Monitoring Data for the Nature Center Lakes (5 and 6) at El Dorado Park

Location	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hardness (mg/kg)
EDL-5	13:20	2,570	15.5	1.54	2.81	1.56	1,210	7.34	5.33	19.8	12,900
EDL-6	15:15	4,950	41.9	2.71	4.86	1.73	1,050	11.0	3.63	12.5	9,370

In addition to the August 2010 sample, the southern two lakes at El Dorado Park were resampled for nutrients on September 28, 2010. Table G-82 summarizes the nutrient data collected in each lake as well as the potable water source. TKN concentrations ranged from 0.79 to 0.86 mg-N/L. Ammonia concentrations ranged from <0.03 mg-N/L to 0.05 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.36 mg-N/L to 0.41 mg-N/L. Orthophosphate ranged from 0.008 mg-P/L to 0.017 mg-P/L. Total phosphorus was measured as below the detection limit of 0.0165 mg-P/L in both lakes. Chlorophyll *a* ranged from 6.01 µg/L to 6.68 µg/L.

Table G-82. August 9, 2010 In-lake Water Column Measurements for the Nature Center Lakes (5 and 6) at El Dorado Park

Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
EDL-5A	15:10	0.864	0.0475	0.0470	0.378	0.0100	<0.0165	6.68	1.6
EDL-5A (duplicate)	15:10	0.808	0.0490	0.0480	0.364	0.0170	<0.0165	6.01	1.6
EDL-6	15:10	0.792	<0.0300	0.0540	0.409	0.00800	<0.0165	6.01	1.4
EDL-PW	12:45	0.672	0.292	0.0600	0.173	0.00900	<0.0165	<1.00	NA

Similar to the August 2010 event, two continuous monitoring probes were deployed September 27, 2010 in each southern lake over a 24-hour period at depths of about 1 to 1.3 meters below the surface. DO concentrations ranged from 7.4 mg/L to 8.2 mg/L in Nature Center North Lake (Figure G-29) and from 6.6 mg/L to 9.7 mg/L in Nature Center South Lake (Figure G-30). pH ranged from about 7.6 to 8.1 in both lakes.

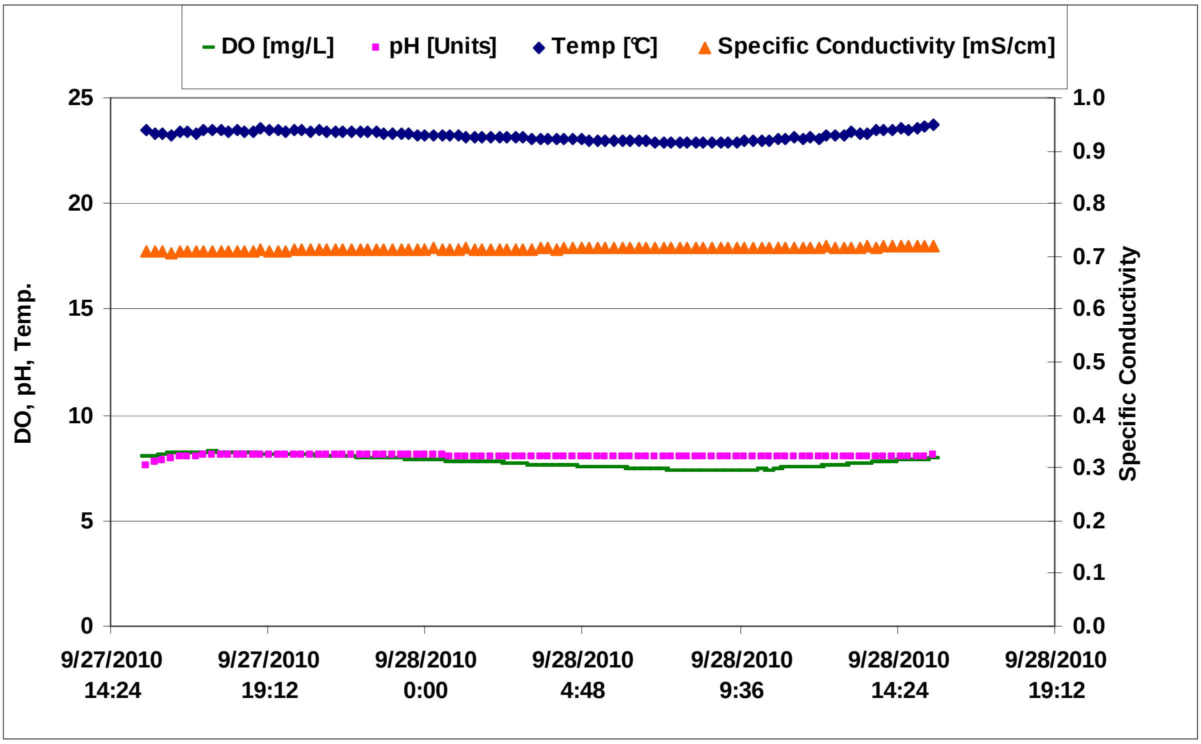

Figure G-29. 24-Hour Probe Data Collected at in El Dorado Park Lake 5 on September 27, 2010

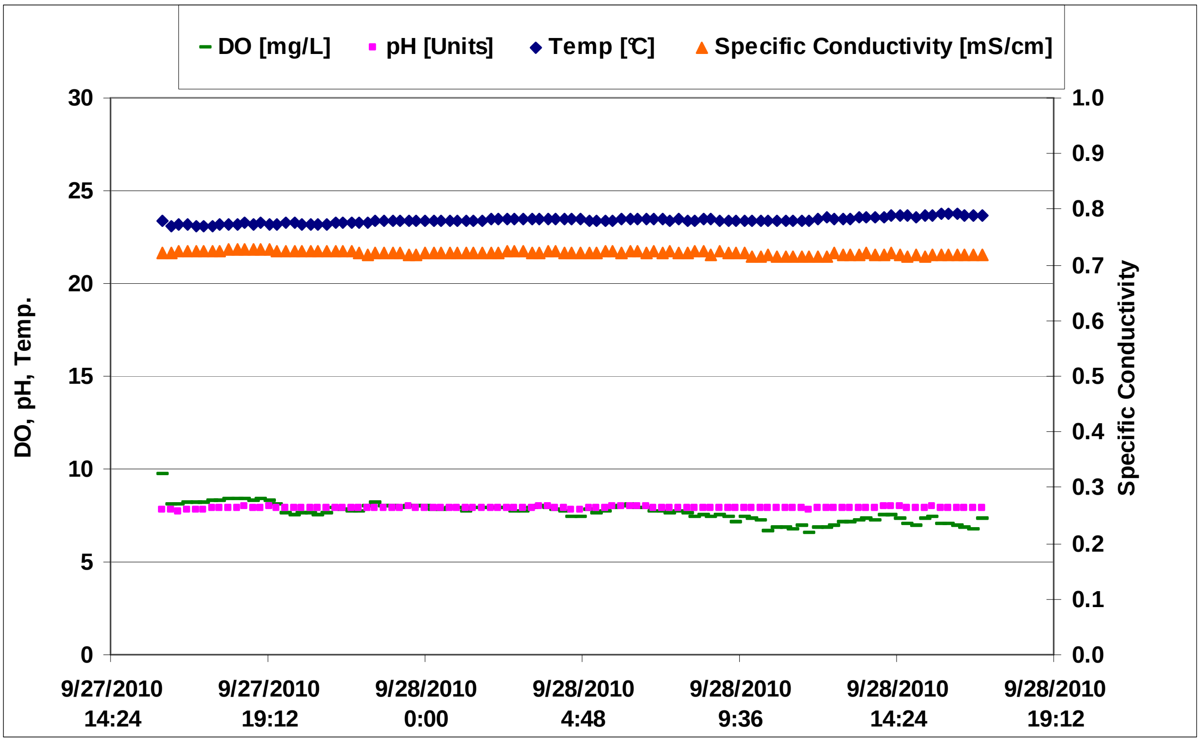

Figure G-30. 24-Hour Probe Data Collected in El Dorado Park Lake 6 on September 27, 2010

On September 28, 2010, depth-profile data were collected for Nature Center North Lake (EDL-5A) during this sampling event, which are summarized in Table G-83. These data were not collected at Nature Center South Lake due to time constraints. DO measurements collected from the surface of Nature Center North Lake ranged from 9.2 mg/L to 10.9 mg/L. At 0.4 meters above the bottom, DO was measured as 9.2 mg/L. Figure G-31 shows the profile data collected on September 28, 2010 at station EDL-5A.

Table G-83. Profile Data Collected in the Nature Center Lakes at El Dorado Park on September 28, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
EDL-5A	15:10	0.5	23.86	7.8	9.21-10.85	0.668	127-104
		1	23.75	7.78	9.17	0.668	99
		1.5	23.82	7.79	9.14	0.668	92
		2	23.76	7.79	9.15	0.669	87
		2.5	23.71	7.78	9.18	0.667	82

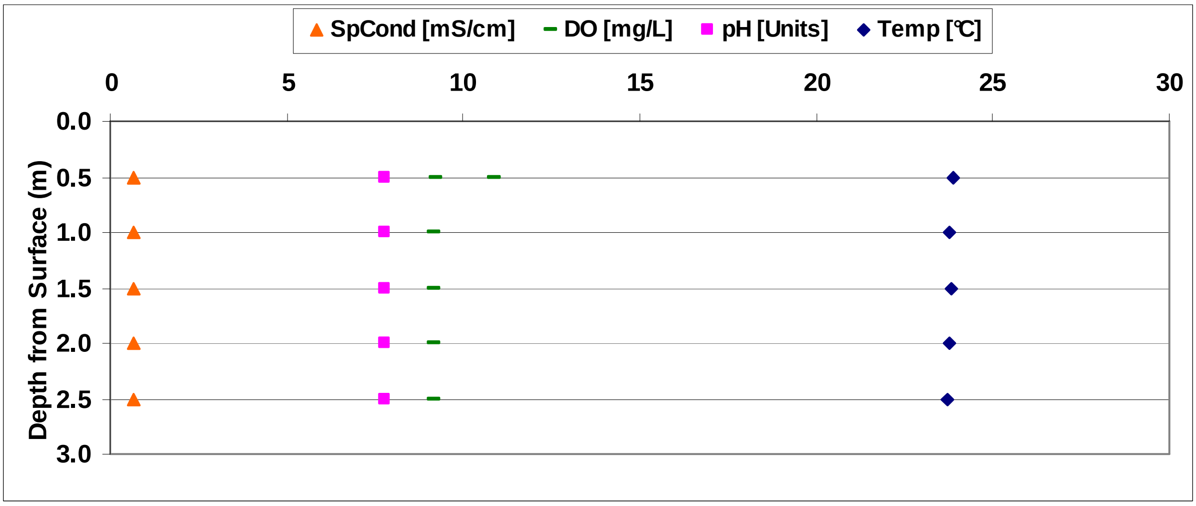

Figure G-31. Profile Data Collected in El Dorado Park Lake 5 on September 28, 2010

Sediment samples were also collected during the September 2010 monitoring event. Table G-84 summarizes these data.

Table G-84. September 28, 2010 Sediment Monitoring Data for the Nature Center Lakes (5 and 6) at El Dorado Park

Loca-tion	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hard-ness (mg/kg)
EDL-5A	15:50	1,140	5.16	0.922	1.34	0.0264	253	2.00	15.4	43.4	11,600
EDL-5B	14:30	4,840	11.5	2.08	2.89	<0.00750	189	5.06	70.6	19.8	10,800
EDL-5C	14:40	4,530	24.0	1.85	2.64	0.0191	435	4.80	25.2	21.6	11,700
EDL-6	17:10	23,200	37.1	2.98	4.17	0.00891	281	8.60	118	13.4	9,610

G.8.2 Monitoring Related to Mercury Impairment

Mercury data have been collected in the El Dorado Park lakes watershed since 1991. Fish tissue concentrations were measured three times under the Toxic Substances Monitoring Program (TSMP, 2009) from 1991 to 1998 and by the Regional Board in 2007 (Davis et al., 2008) and 2010. In-lake water column concentrations were measured as part of the Urban Lakes Study (UC Riverside, 1994) in 1992. USEPA and the Regional Board sampled in-lake and tributary water column and sediment mercury concentrations during two events in 2009.

G.8.2.1 In-Lake Sampling

G.8.2.1.1 Water Column Measurements

Mercury concentrations were measured in the water column of Lake 2 (pink triangle, Figure G-17) as part of the Urban Lakes Study (UC Riverside, 1994) in July and August of 1992. The detection limit of this dataset was relatively high (500 ng/L) and all 12 samples were less than detection.

In February 2009, the Regional Board and USEPA sampled mercury concentrations at Stations EDL-1 and EDL-2. Both samples were collected from a depth of 0.76 meters and the total depth at each location was approximately 4.4 meters. A duplicate sample was collected at EDL-2 and analyzed for total mercury. In July 2009, the Regional Board and USEPA sampled Lakes 1, 2, and 6 for mercury with a duplicate sample collected in Lake 2. Sampling depths were 0.58 m, 0.33 m, and 0.96 m, respectively. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.021 ng/L.

Table G-85 presents the in-lake mercury and TSS measurements for the two sampling events. Methylmercury concentrations ranged from 0.020 ng/L to 0.072 ng/L. Total mercury concentrations were consistently below the water quality standard (50 ng/L) and ranged from 0.41 to 1.17 ng/L.

Table G-85. In-lake Water Column Measurements for the El Dorado Park Lakes

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
EDL-1	2/26/2009	8:30	0.046	0.89	4.6
EDL-2		11:15	0.041	1.08	4.0
EDL-2 (duplicate)		11:30	NA	1.17	NA
EDL-1	7/15/2009	11:15	0.063	0.50	5.9
EDL-2		9:40	0.072	0.41	9.6
EDL-2 (duplicate)		9:40	NA	0.42	NA
EDL-6		15:50	0.020	1.03	2.9

Additional water quality samples were collected from the El Dorado Park lakes. Table G-86 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Lakes 1 and 2 on February 26, 2009 and July 15, 2009.

Table G-86. Supplemental Water Quality Monitoring for In-lake Samples in the El Dorado Park Lakes

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
EDL-1	2/26/2009	8:30	33.7	283.3	203	380	3.9
EDL-2		11:15	80.9	80.2	274	532	6.0
EDL-5		12:15	55.65	88.2	126	406	3.9
EDL-6		13:20	56.1	88.3	126	370	6.2
EDL-1	7/15/2009	11:15	25.56	20.47	210	350	4.8
EDL-2		9:40	80.51	33.28	274	532	11.3
EDL-2 (duplicate)		9:40	81.64	33.71	274	NA	NA

G.8.2.1.1 Sediment Samples

During the February and July sampling events, USEPA and the Regional Board also collected sediment samples at each station to measure total and methylmercury concentrations in sediment. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.51 µg/kgto 6.96 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.023 µg/kg to 0.049 µg/kgg. Detection limits were adjusted to account for sample aliquot size.

Table G-87 presents the sediment mercury concentrations measured in the El Dorado Park lakes. Concentrations are reported on a dry weight basis. The methylmercury concentrations measured in sediments at these three stations ranged from approximately 0.1 µg/kg to 0.2 µg/kg; the total mercury concentration ranged from 78 µg/kg to 188 µg/kg.

Table G-87. In-lake Sediment Concentrations for the El Dorado Park Lakes

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
EDL-1	2/26/2009	8:30	0.198	123	28.5	541.5
EDL-2		11:15	0.202	86.8	36.74	130.4
EDL-2 (duplicate)		11:30	NA	89.5	35.88	Not sampled
EDL-1	7/15/2009	11:30	0.167	126	28.28	219.1
EDL-2		9:40	0.102	78.0	28.36	192.98
EDL-2 (duplicate)		9:40	0.121	94.8	29.04	NA
EDL-6		15:50	0.113	188	18.26	822.89

G.8.2.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in the El Dorado Park lakes since 1991. Lake 1 was sampled by the TSMP in the 1990s as composite samples: the number in each composite was not provided. The Surface Water Ambient Monitoring Program (SWAMP) sampled individual fish from Lake 2 during the summer of 2007 (Davis et al., 2008) and March 2010. Table G-88 presents the fish tissue mercury concentrations on a wet weight basis. Concentrations range from 0.131 ppm to 0.678 ppm. The applicable fish tissue target for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-88. Fish Tissue Mercury Concentrations Measured in the El Dorado Park Lakes

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	4/21/1991	382	0.470
TSMP	4/26/1992	378	0.550
TSMP	6/23/1998	350	0.602
SWAMP	Summer 2007	537	0.318
SWAMP	Summer 2007	479	0.672

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	386	0.432
SWAMP	Summer 2007	391	0.408
SWAMP	Summer 2007	380	0.480
SWAMP	Summer 2007	386	0.351
SWAMP	Summer 2007	400	0.310
SWAMP	Summer 2007	387	0.559
SWAMP	Summer 2007	391	0.500
SWAMP	Summer 2007	378	0.491
SWAMP	Summer 2007	370	0.446
SWAMP	Summer 2007	304	0.190
SWAMP	Summer 2007	294	0.188
SWAMP	Summer 2007	206	0.150
SWAMP	Summer 2007	219	0.131
SWAMP	3/30/2010	409	0.678
SWAMP	3/30/2010	348	0.259
SWAMP	3/30/2010	345	0.199
SWAMP	3/30/2010	343	0.235
SWAMP	3/30/2010	352	0.151

Piscivorous fish tend to have increased mercury tissue concentrations with age. Figure G-32 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length. As expected, fish tissue mercury concentrations increase with length. All fish specimens with a mean or individual length greater than 350 mm exceed the fish tissue target of 0.22 mg/kg. Eleven individual and three composite samples had fish tissue concentrations greater than the target, while four individual samples had concentrations less than the target.

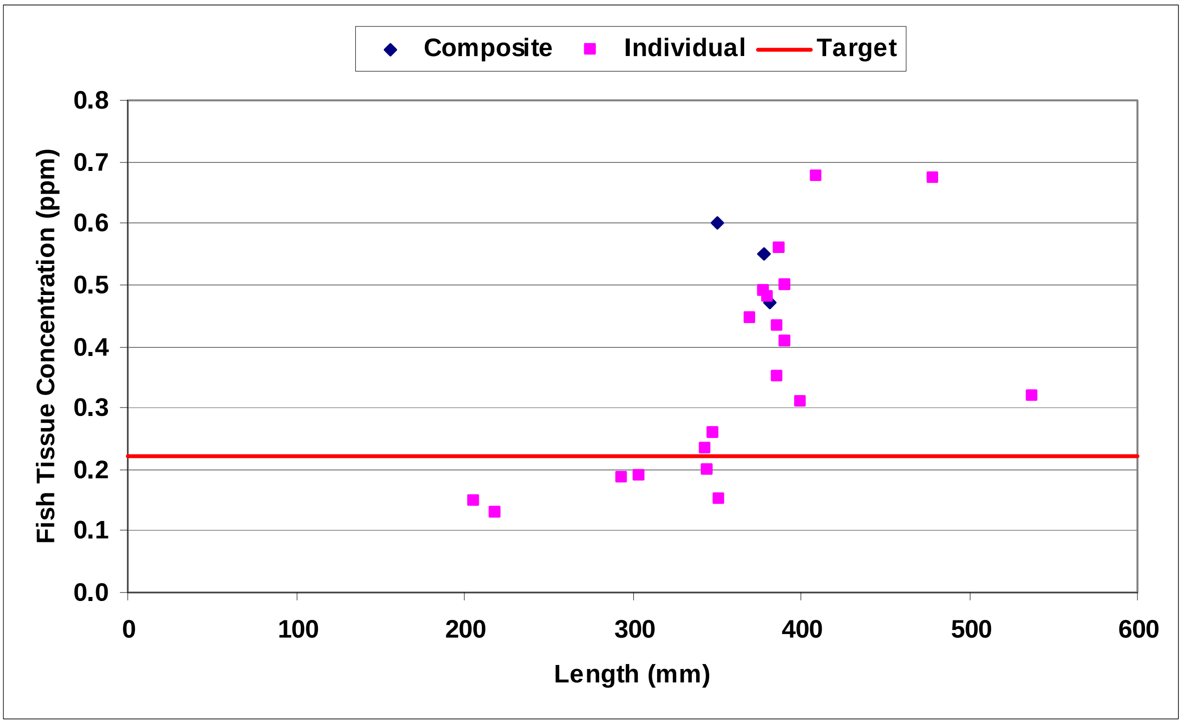

Figure G-32. Mercury Concentrations in Largemouth Bass in the El Dorado Park Lakes

G.8.2.3 Tributary/Inflow Monitoring

G.8.2.1.3 Water Column Measurements

During both the February and July 2009 sampling events, the only visible inputs to the El Dorado Park lakes were the groundwater (GW) input to the most upstream lake in the northern four lakes and the potable water (PW) input to the most upstream lake in the southern two lakes. No culverts in the park area were discharging. Concentrations of methyl and total mercury observed in these inputs are reported in Table G-89. Total mercury was analyzed with EPA Method 1631 with a detection limit of 3.03 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. Detection limits for the groundwater analyses were adjusted to account for sample aliquot size.

The groundwater input was sampled during both events near the pump house after allowing the line to purge for at least ten minutes. Methylmercury concentrations ranged from 0.109 ng/L to 0.215 ng/L; total mercury ranged from 131 ng/L to 142 ng/L. The concentration of total mercury in these samples were almost three times higher than the water quality standard of 50 ng/L and 100 to 200 times higher than the concentrations observed in the water columns of the northern lakes (Section G.8.2.1.1). The portion of mercury in the methyl form ranged from 0.08 to 0.15 percent; the methylmercury concentrations were two to five times higher than the average measured in the northern lakes. The potable water input was only sampled during the July event. Concentrations of methyl and total mercury were 0.020 ng/L and 2.84 ng/L, respectively.

Reclaimed water (RW) is used at the park for irrigation. This source was sampled in December 2009. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. These values are similar to the potable water results.

Table G-89. Tributary/Inflow Water Column Measurements for the El Dorado Park Lakes

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
EDL-GW	2/26/2009	9:40	0.215	142	0.7
EDL-GW	7/15/2009	11:30	0.109	131	1.7
EDL-PW	7/15/2009	14:40	0.020	2.84	0.3
EDL-RW	12/1/2009	14:30	0.021	1.46	0.8

The Long Beach Water Department samples five wells in the vicinity of the El Dorado Park Lakes. However, the analysis employed has relatively high detection limits (200 ng/L), and all samples have been less than detection.

G.8.2.1.3 Sediment Samples

During both the February and July 2009 monitoring events, the only inputs observed to the El Dorado Park lakes were the groundwater and potable water inputs. Neither of these inputs has a sediment-transport capacity.

G.8.3 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 El Dorado Park lakes was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. El Dorado Park lakes is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metal samples were collected from the north end of Alamo Lake at El Dorado Park lakes (shown in Figure G-17 (pink triangle)), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994). Results are shown in Table G-90. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 4.5 meters. The range of the 45 dissolved copper samples was between less than 10 µg/L and 99 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 4.5 meters. The 45 samples collected ranged in concentration from less than 1 µg/L to 108 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for El Dorado Park lakes states that copper and lead were not supporting their assessed uses: 45 measurements had a maximum lead concentration of 108 µg/L, a maximum copper concentration of 99 µg/L, and a maximum zinc concentration of 21 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metal levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-90. El Dorado Park Lakes 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/16/1992	0	23	<1
	3	21	<1
	4.5	17	2
7/16/1992	0	N/A	<1
	3	25	1
	4.5	27	2
7/16/1992	0	40	14
	3	28	8
	4.5	29	<1
8/20/1992	0	31	2
	2	21	<1
	4	18	1
9/24/1992	0	13	2
	2	<10	3
	4	<10	3
10/20/1992	0	16	<1
	2	21	<1
	3.5	24	<1
11/12/1992	0	21	2
	2.5	19	2
	3.5	34	3
12/15/1992	0	<10	1
	2.5	<10	1
	3.5	<10	1
1/21/1993	0	<10	2
	2.5	<10	<1
	3.5	<10	<1
2/10/1993	0	<10	<1
	2.5	18	<1
	3.5	99	<1
3/8/1993	0	<10	17

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	1.5	19	<1
	2.5	N/A	N/A
	3.5	36	<1
4/8/1993	0	16	37
	1.5	14	3
	2.5	12	<1
	3.5	11	<1
5/12/1993	0	15	28
	2.5	13	4
	3.5	12	1
6/15/1993	0	<10	82
	1.5	<10	34
	2.5	<10	86
	3.5	<10	108

Table G-91 presents38 additional metals samples that were collected by the USEPA and Regional Board between February 2009 and September 2010 at the El Dorado Park lakes. Samples were collected at locations EDL-1, EDL-2, EDL-3, EDL-4, EDL-5, EDL-6, and shoreline samples at EDL-1S, EDL-2S, EDL-3S and EDL-6S. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.6 µg/L; copper concentrations were between 0.4 µg/L and 6.7 µg/L; lead concentrations ranged from <0.1 µg/L to 0.4 µg/L; and zinc concentrations were <0.1 µg/L to 22.7 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 56 mg/L to 138.7 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-91 were weighted more heavily in the assessment.

Table G-91. Water Column Metals Data for the 2008-2010 El Dorado Park Lakes Sampling Events

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
2/26/2009	EDL 1	100.2	<0.2	1	<0.1	0.3	
2/26/2009	EDL 2	113.4	<0.2	1.9	0.1	0.4	average of duplicate
7/15/2009	EDL 1	117.3	<0.2	1.2	0.1	1.6	
7/15/2009	EDL 2	122.5	<0.2	2.5	0.1	2.3	average of duplicate

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
7/15/2009	EDL 3	88.1	<0.2	2.7	0.1	<0.1	
7/15/2009	EDL 4	87.4	<0.2	3.8	0.2	0.6	
7/15/2009	EDL 6	84.6	<0.2	2.5	0.1	4.3	
7/15/2009	EDL 5	85.6	<0.2	2.7	0.1	3.9	
12/1/2009	EDL1S	112.7	<0.2	0.5	<0.1	0.6	average of replicates
12/1/2009	EDL2S	132.2	<0.2	0.9	0.1	2.6	
12/1/2009	EDL3S	94.3	<0.2	1.6	0.2	1.6	
12/1/2009	EDL5	125	<0.2	2.9	0.1	11.5	
12/1/2009	EDL6S	120.8	<0.2	2.9	0.3	13	average of duplicate
12/1/2009	EDL4S	93.1	<0.2	1.4	0.2	2.3	
12/15/2009	EDL1	124.3	<0.2	0.4	<0.1	<0.1	average of replicates
12/15/2009	EDL2	138.7	<0.2	1.1	0.1	<0.1	
12/15/2009	EDL3	97.7	<0.2	1.8	0.3	2.5	average of duplicates
12/15/2009	EDL4	97.9	<0.2	2.5	0.4	1.1	
12/15/2009	EDL5	120.3	<0.2	2.8	0.2	14.2	
12/15/2009	EDL6	124.4	<0.2	2.7	0.3	10.6	
1/26/2010	EDL1S	107.8	<0.2	1.2	<0.1	1.4	average of replicates & duplicate
1/26/2010	EDL2S	123.8	<0.2	1.7	0.1	1.3	
1/26/2010	EDL3S	95.2	<0.2	2.5	0.2	<0.1	
1/26/2010	EDL4S	94.9	<0.2	3	0.2	1.6	
1/26/2010	EDL5	81.2	<0.2	3.4	0.2	13.9	
1/26/2010	EDL6S	103.9	<0.2	3.7	0.2	22.7	
8/10/2010	EDL1	NA	0.585	0.509	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL2	NA	0.502	0.915	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL3	NA	0.516	1.76	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL4	NA	0.525	2.16	<0.05	3.60	Hardness not analyzed
8/10/2010	EDL5	60.5	0.493	3.70	<0.05	5.21	
8/10/2010	EDL6	58.1	0.495	3.66	<0.05	10.4	
9/27/2010	EDL 1S	61	<0.2	<0.4	<0.05	<0.1	

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
9/27/2010	EDL 2S	77	<0.2	1.14	<0.05	<0.1	
9/27/2010	EDL 3S	71	<0.2	4.51	<0.05	2.51	
9/27/2010	EDL 4S	72	<0.2	4.29	<0.05	1.25	
9/27/2010	EDL 5	56	<0.2	5.895	<0.05	12.25	
9/27/2010	EDL 6	57	<0.2	6.7	<0.05	12.2	

Note: All data collected by the Regional Board or USEPA.

USEPA collected eight sediment samples between August and September 2010 to further evaluate lake conditions. Table G-92 summarizes the copper and lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target. There were four sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-92. Sediment Metals Data for the August 2010 El Dorado Park Lakes Sampling Event

Organization	Date	Station ID	Copper mg/kg)	Lead (mg/kg)	Notes
EPA	8/10/2010	EDL1	101	18.7	
EPA	8/10/2010	EDL2	109	19.8	
EPA	8/10/2010	EDL3	97.6	16.1	
EPA	8/10/2010	EDL4	121	16.0	
EPA	8/10/2010	EDL5	533	47.2	
EPA	8/10/2010	EDL6	278	34.6	
EPA	09/28/2010	EDL5	237.3	23.7	Average of field replicates
EPA	09/28/2010	EDL6	466	55.7	

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G.9 Monitoring Data for North, Center, and Legg Lakes

Monitoring data relevant to the impairments of North, Center, and Legg lakes are available from 1992, 1993, 2009, and 2010. Figure G-33 shows the historical and recent monitoring locations for these lakes.

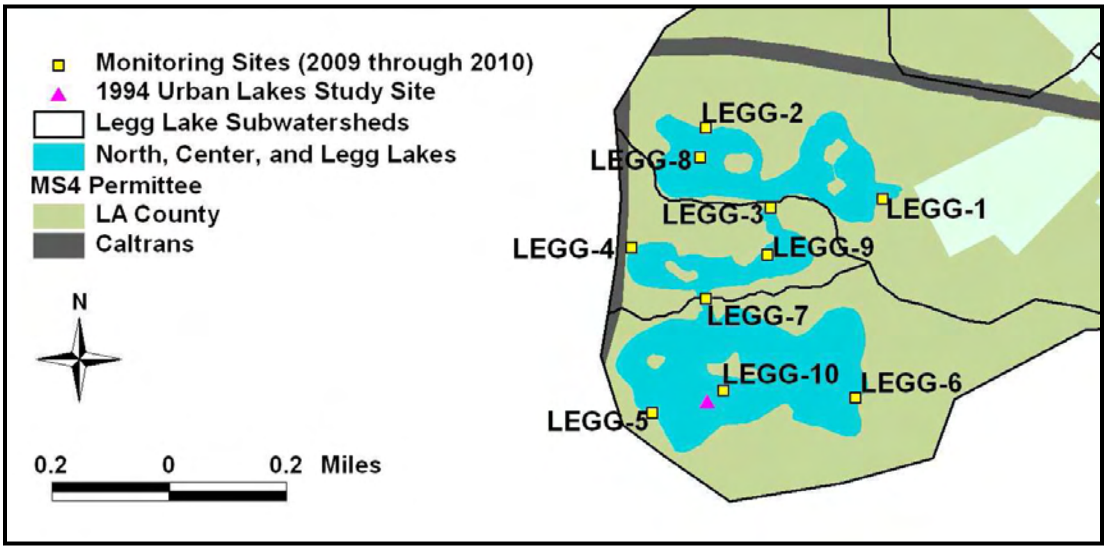

Figure G-33. North, Center, and Legg Lakes Monitoring Sites

G.9.1 Monitoring Related to Nutrient Impairments

Legg Lake was monitored in 1992 and 1993 for water quality as part of the Urban Lakes Study from the lower section of the lake on the western side (pink triangle, Figure G-33) (Table G-93). TKN generally ranged from 0.6 mg/L to 1.0 mg/L although three samples were less than the reporting limit and one outlier had a concentration of 37 mg/L. The majority of the ammonium samples (33 of 43) were less than the reporting limit; ammonium concentrations as high as 0.4 mg/L were observed, which are above both the chronic and acute targets (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All nitrite samples were less than the reporting limit, and nitrate concentrations did not exceed 0.2 mg/L. Both phosphate and total phosphorus were less than the reporting limit in all 43 samples. pH ranged from 8.0 to 8.9, and TOC ranged from 2.3 mg/L to 6.6 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 2 µg/L to 27 µg/L with an average of 15 µg/L.

Table G-93. Legg Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/6/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5.4	200
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	197
	2.1	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.8	209
7/6/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5	199

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	1.3	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	202
	1.6	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.6	200
7/6/1992	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.9	4.6	206
	1.4	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	4.7	193
	1.8	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	5.1	201
8/12/1992	0	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.6	5.5	248
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.5	196
	2.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	217
8/12/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	6.3	204
	2	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.3	207
8/12/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	6.6	191
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.2	218
9/21/1992	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5.1	201
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.8	192
	2.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.7	190
10/8/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.9	206
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.2	212
	2.5	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.1	211
11/3/1992	0	37	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.9	179
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5	200
	3	1	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.7	244
12/15/1992	0	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	3	228
	2	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	3.2	235
1/13/1993	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8	3.3	191
	2	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.1	3.5	190
2/3/1993	0	0.7	<0.01	<0.01	0.2	<0.01	<0.01	8.7	4.4	215
	2	0.9	<0.01	<0.01	0.2	<0.01	<0.01	8.7	3.5	222
3/4/1993	0	0.8	0.4	<0.01	0.2	<0.01	<0.01	8	3	199
	1.5	0.8	0.3	<0.01	0.2	<0.01	<0.01	8.1	2.8	197
	2.5	1	0.3	<0.01	0.1	<0.01	<0.01	8.1	2.9	195
4/13/1993	0	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8.4	2.9	227
	1.5	0.8	0.2	<0.01	<0.01	<0.01	<0.01	8.4	2.5	228

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	2.5	0.9	0.2	<0.01	0.1	<0.01	<0.01	8.5	2.7	223
5/5/1993	0	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	2.8	202
	2	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	2.4	198
	3	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.3	192
6/8/1993	0	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.9	215
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	3.1	215
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.9	214

The Regional Board's 1996 Water Quality Assessment Database does not include data for Legg Lake or its watershed. The Assessment Report does include summary information for the impairments. Ammonia was partially supporting the aquatic life use; 43 ammonium samples were collected with concentrations ranging from non-detect to 0.35 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. pH was partially supporting the aquatic life use and not supporting the secondary drinking water use. Eighty-four measurements of pH ranged from 7.6 to 8.9. Odor was listed as not supporting the contact and non-contact recreation uses.

The Legg Lake system was sampled multiple times during May, June, and July 2007 (Table G-94; data provided by the county of Los Angeles). Nineteen of 21 samples of ammonia had concentrations ranging from less than the detection limit of 0.01 mg-N/L to 0.36 mg-N/L; two samples had ammonia concentrations of 0.51 mg-N/L and 0.53 mg-N/L (both were collected from Center Lake in May). Nitrate concentrations ranged from less than the detection limit of 0.02 mg-N/L to 0.59 mg-N/L. Orthophosphate ranged from less than the detection limits (either 0.01 mg-P/L or 0.02 mg-P/L, depending on the sampling event) to 0.07 mg-P/L.

Table G-94. 2007 County of Los Angeles Water Quality Data for the Legg Lake System

Monitoring Location	Date	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Orthophosphate (mg-P/L)
Center Lake open water	5/18/07	0.23	0.02	0.02
	5/25/07	0.51	0.02	0.02
	5/31/07	0.53	0.02	0.07
	6/18/07	0.06	0.12	0.02
	6/21/07	0.1	0.15	0.02
	6/29/07	0.36	0.18	0.02
	7/5/07	0.25	0.07	0.01
North Lake east storm drain inlet	5/18/07	0.61	0.02	0.02
	5/25/07	0.01	0.04	0.02
	5/31/07	0.04	0.02	0.07

Monitoring Location	Date	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Orthophosphate (mg-P/L)
	6/18/07	0.22	0.02	0.02
	6/21/07	0.33	0.02	0.02
	6/29/07	0.17	0.02	0.02
	7/5/07	0.33	0.03	0.01
North Lake west storm drain inlet	5/18/07	0.35	0.02	0.02
	5/25/07	0.01	0.05	0.02
	5/31/07	0.04	0.02	0.07
	6/18/07	0.12	0.02	0.02
	6/21/07	0.01	0.02	0.02
	6/29/07	0.99	0.02	0.02
	7/5/07	0.25	0.03	0.01
North Lake open water	5/18/07	0.01	0.02	0.02
	5/25/07	0.01	0.17	0.02
	5/31/07	0.02	0.02	0.07
	6/18/07	0.01	0.02	0.02
	6/21/07	0.04	0.02	0.02
	6/29/07	0.2	0.02	0.02
	7/5/07	0.03	0.05	0.01
North PVC irrigation pipe outlet	7/5/07	0.07	0.02	0.01
South Lake open water	5/18/07	0.01	0.1	0.02
	5/25/07	0.01	0.06	0.02
	5/31/07	0.1	0.02	0.07
	6/18/07	0.1	0.31	0.02
	6/21/07	0.12	0.59	0.02
	6/29/07	0.05	0.49	0.02
	7/5/07	0.07	0.21	0.01
South Lake near EPA treatment plant	5/18/07	0.01	0.17	0.02
	5/25/07	0.01	0.25	0.02
	5/31/07	0.12	0.34	0.07
	6/18/07	0.05	0.86	0.02
	6/21/07	0.08	0.65	0.02
	6/29/07	5.76	0.59	0.02

Monitoring Location	Date	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Orthophosphate (mg-P/L)
	7/5/07	0.01	0.22	0.01
South Lake near well water inlet	5/18/07	0.01	0.11	0.02
	5/25/07	0.01	0.19	0.02
	5/31/07	0.1	0.12	0.07
	6/18/07	0.11	0.46	0.02
	6/21/07	0.19	0.5	0.02
	6/29/07	0.06	0.37	0.02
	7/5/07	0.15	0.22	0.01

On February 3, 2009, the Regional Board sampled water quality around the shoreline of Legg Lake (stations LEGG-5 and LEGG-6) as well as the two smaller lakes to the north (stations LEGG-1 through LEGG-4) and the connecting channel to Legg Lake (LEGG-7). Site LEGG-44 is a field duplicate site for LEGG-4. Note that the 2006 303(d) lakes coverage shows only Legg Lake proper. Table G-95 presents these monitoring results. As expected with shoreline monitoring, nutrients and chlorophyll *a* concentrations were relatively high (see Section 6 in main document). TKN ranged from 0.63 mg/L to 2.6 mg/L. Ammonia ranged from non-detect to 0.07 mg/L. Nitrite ranged from 0.04 mg/L to 0.05 mg/L, and nitrate ranged from 0.04 mg/L to 0.74 mg/L. Dissolved orthophosphate was only greater than the detection limit at LEGG1 with a concentration of 0.0106 mg/L. Total phosphorus concentrations ranged from 0.017 mg/L to 0.089 mg/L. TOC ranged from 3.0 mg/L to 5.9 mg/L. TDS ranged from 46 mg/L to 476 mg/L; TSS ranged from 5.7 mg/L to 16.6 mg/L. Chlorophyll *a* concentrations ranged from 26.7 μg/L to 115 μg/L. In general, concentrations were lower at the two Legg Lake shoreline sites relative to the other sites. This was particularly true of chlorophyll *a* concentrations, which ranged from 26.7 μg/L to 38.3 μg/L at the two locations. Secchi depths were not measured at these shoreline sites.

Table G-95. February 2009 Water Quality Monitoring Around the Shoreline of Legg Lake

Sample Location	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	TOC (mg/L)	TDS (mg/L)	TSS (mg/L)	Chl *a* (μg/L)
LEGG1	2.2	<0.03	0.04	0.185	0.01060	0.08	5.55	121	16.6	115
LEGG2	2.6	0.03	0.04	0.04	<0.0075	0.089	5.9	46	15.8	103.6
LEGG3	1.4	0.06	0.05	0.39	<0.0075	0.087	5.1	256	8.8	37.4
LEGG4	0.63	0.07	0.04	0.45	<0.0075	0.047	3.8	374	6.7	27.6
LEGG44	1.5	0.06	0.04	0.45	<0.0075	0.03	5.4	436	5.7	29.4
LEGG5	1.4	<0.03	0.04	0.64	<0.0075	0.033	4.4	444	10.8	38.3
LEGG6	0.70	<0.03	0.04	0.74	<0.0075	0.03	3.5	476	6.7	26.7
LEGG7	1.4	<0.03	0.04	0.63	<0.0075	0.017	3	434	10	32

Field data for the February 2009 monitoring event are summarized in Table G-96. At the two Legg Lake shoreline sites, temperature ranged from 16 ºC to 16.5 ºC, and pH ranged from 8.1 to 8.3. Across all sites, temperature ranged from 12.5 ºC to 16.5 ºC, and pH ranged from 8.0 to 9.0.

Table G-96. February 2009 Field Data for the Legg Lake Monitoring Event

Sample Location	Temperature ºC	pH
LEGG1	12.5	9.0
LEGG2	14.5	8.8
LEGG3	12.5	8.0
LEGG4	14.5	8.2
LEGG44	15.0	8.3
LEGG5	16.5	8.1
LEGG6	16.0	8.3
LEGG7	12.5	9.0

The North, Center and Legg lakes were sampled for summer conditions on July 14, 2009. In-lake samples were taken at Legg Lake sites 8, 9, and 10. A duplicate was performed at Legg Lake 10 as a quality control measure. Site 7 is in the channel that connects Legg Lake and Center Lake. The nutrients measured during this monitoring event are shown in Table G-97. Groundwater was also sampled from a pump. The groundwater pump provides flow to the North Lake and the South/Legg Lake via two cascading waterfall areas. Water flowing in North Lake at station Legg-3 was sampled from the pipe on the center lakeside flowing towards the north lake. The total depth at the sampling location and entrance of the pipe was 0.25 m. The samples at Legg-7 were taken at a depth of 0.20 m. The total depth of Legg-7 is 0.61 meters and has a Secchi depth of 0.41 meters. Legg-9 was sampled at 0.30 meters and has a total depth of 0.88 meters. The Secchi depth at Legg-9 was 0.61 meters. Samples at Legg-8 and Legg-10 (including the duplicate) were taken at approximately 0.20 meters. The depth of Legg-8 and Legg-10 are 2.2 and 2.5 meters, respectively. The Secchi depth at Legg-8 was 0.38 meters and the Secchi depth at Legg-10 was 0.48 meters.

Table G-97. July 2009 Water Column Measurements for the Legg Lakes

Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
LEGG-7[1]	11:05	1.4	<0.03	<0.01	<0.01	<0.0075	0.043	64.1	0.41
LEGG-8	12:15	1.7	<0.03	<0.01	<0.01	<0.0075	0.066	63.1	0.38
LEGG-9	9:30	1.4	<0.03	<0.01	<0.01	<0.0075	0.043	37.4	0.61
LEGG-10	10:45	1.47	<0.03	<0.01	<0.01	<0.0075	0.046	93.45	0.48
LEGG-10D	10:45	1.4	<0.03	<0.01	<0.01	<0.0075	0.089	NS	0.48
LEGG-GW[2]	8:18	<0.46	0.03	<0.01	1.26	<0.0075	0.036	NS	NA
LEGG-3[3]	13:30	1.5	0.03	<0.01	<0.01	<0.0075	0.185	26.7	NA

[1] LEGG-7 represents a channel sample location.

[2] LEGG-GW represents the groundwater input to the North and South Lakes, not an in-lake sample.

[3] LEGG-3 represents input from Center Lake to the North Lake, sampled from a pipe, not an in-lake sample.

The July 2009 sampling event also monitored for chloride, sulfate, total alkalinity, hardness, TDS, TSS, DOC and TOC. These samples were taken at Legg-7, 8, 9, and 10. No duplicate was performed for these parameters. These data are shown in Table G-98.

Table G-98. July 2009 Supplemental Water Quality Monitoring for In-lake Samples in the Legg Lakes

Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
LEGG-7[1]	11:05	68.36	117.13	56	138.6	500	25	16.4	17.6
LEGG-8	12:15	42.12	62.38	50	97.6	302	26	12.1	13.3
LEGG-9	9:30	69.82	119.86	76	158.7	434	10.5	16.4	16.9
LEGG-10	10:45	66.18	114.21	54	136.2	440	21	14.85	18.4
LEGG-GW[2]	8:18	36.3	64.13	160	187.8	394	1	0.3	0.2
LEGG-3[3]	13:30	68.4	118.34	78	165.7	512	14.15	17.6	17

[1] LEGG-7 represents a channel sample location.

[2] LEGG-GW represents the groundwater input to the North and South Lakes, not an in-lake sample.

[3] LEGG-3 represents input from Center Lake to the North Lake, sampled from a pipe, not an in-lake sample.

Profile data were collected at LEGG-7, LEGG-8, LEGG-9, and LEGG-10 during the July 14, 2009 sampling event by USEPA and the Regional Board. These data are presented in Table G-99. The North Lake depth was 2.20 meters and the Secchi depth was 0.38 meters. The temperature in this lake was between 26.3 and 27.1 °C. The average DO is 12.7 mg/L, excluding the much lower bottom DO measurement, which was 7.9 mg/L. The DO maximum in Center Lake, LEGG-9, occurred at 1 meter of depth (11.33 mg/L) and the DO below 2.5 meters of depth was less than 2.0 mg/L. The temperature was between 25.3 and 28.6 °C. Center Lake had a depth of 2.9 meters and a Secchi depth reading of 0.61 meters. A reading was taken in the channel between the Center and South lakes at LEGG-7. The depth was 0.61 meters and the Secchi depth was 0.41 meters. The DO at the Secchi depth was 12.4 mg/L and the temperature was 28.2 °C. The DO in the South Lake, LEGG-10, was as high as 12.9 mg/L in the upper water column to declines to 2.8 mg/L at the bottom off the lake. Based on this data, the lakes appear to be stratified and have a euphotic zone of greater production, occurring just before or around the first meter of depth in each lake.

Table G-99. Data Collected in Legg Lakes on July 14, 2009

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	S Cond (mS/cm)	Secchi Depth (m)	Total Depth (m)
LEGG-7	11:20	0.4	28.2	9.1	12.4	0.633	0.41	0.61
LEGG-8	9:00	Surface	27.1	8.1	13.0	0.381	0.38	2.20
		0.5	27.1	9.1	13.6	0.381		
		0.99	26.8	8.9	13.1	0.383		
		1.01	26.8	8.9	13.1	0.383		
		1.5	26.7	8.5	10.6	0.402		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	S Cond (mS/cm)	Secchi Depth (m)	Total Depth (m)
		2.0	26.3	7.9	7.2	0.435		
LEGG-9	12:00	Surface	28.6	8.7	10.8	0.677	0.61	2.90
		0.03	28.6	8.7	10.8	0.677		
		0.5	28.4	8.8	11.0	0.677		
		1.0	27.6	8.8	11.3	0.677		
		1.5	27.3	8.9	11.2	0.678		
		2.0	26.7	8.1	6.7	0.697		
		2.5	26.3	7.7	1.9	0.707		
		2.8	25.3	7.7	1.7	0.748		
LEGG-10	10:30	Surface	27.7	9.1	12.7	0.631	0.48	2.50
		0.1	27.6	9.1	12.7	0.631		
		0.5	27.4	9.2	12.9	0.630		
		1.0	26.8	9.1	12.8	0.630		
		1.5	26.4	8.8	12.5	0.643		
		2.0	25.9	8.1	7.2	0.671		
		2.4	25.2	8.0	2.8	0.716		

The South Lake was measured again at 15:00 during the July sampling event. The DO in the afternoon was much higher in the first meter of depth. The maximum DO in the afternoon is 16.3 mg/L and the maximum DO in the morning is 12.9 mg/L. The afternoon temperature was slighter higher than the morning temperature in the first meter of the lake. At the surface of the lake, the temperature in the morning was 27.7 °C and rose to 29.4 °C in the afternoon. These data are displayed in Figure G-34.

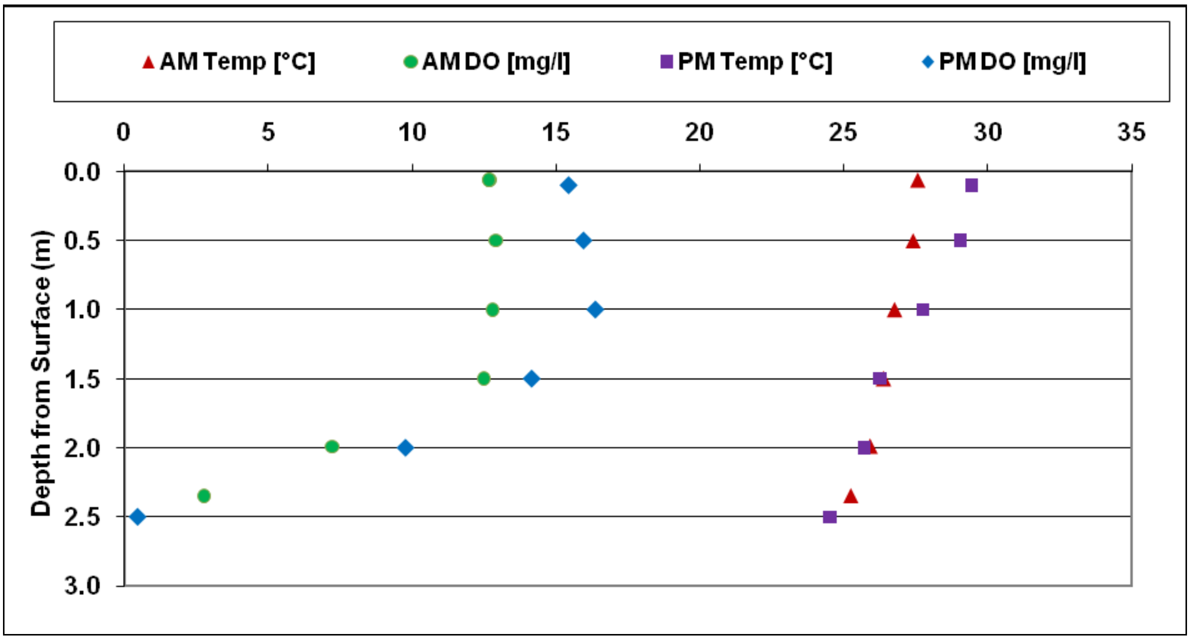

Figure G-34. Profile Data Collected in Lake Legg 10 at approximately 10:30 a.m. and 3:00 p.m. on July 14, 2009

The increase of dissolved oxygen in the afternoon indicate algal productivity in the lake. The saturation level of DO is used to give insight into the impacts of this productivity. The percent saturation in the morning and afternoon are shown in Figure G-35 and listed in Table G-100. The dissolved oxygen saturation is highest in the photic zone in the afternoon at 210 percent. In the morning, the DO saturation in this zone is 165 percent. Saturation above 100 percent indicates DO input from the algal production. The saturation at the bottom of the lake is 34 percent in the morning and 5 percent in the afternoon.

The algae have produced DO and caused the saturation level to exceed 100 percent. The algal productivity is higher in the afternoon when the light intensity is greater. The DO in the bottom of the lake has also been deleted by the increased producvitiy. The DO saturation at the bottom of the lake was 34 percent in the morning and 5 percent in the afternoon.

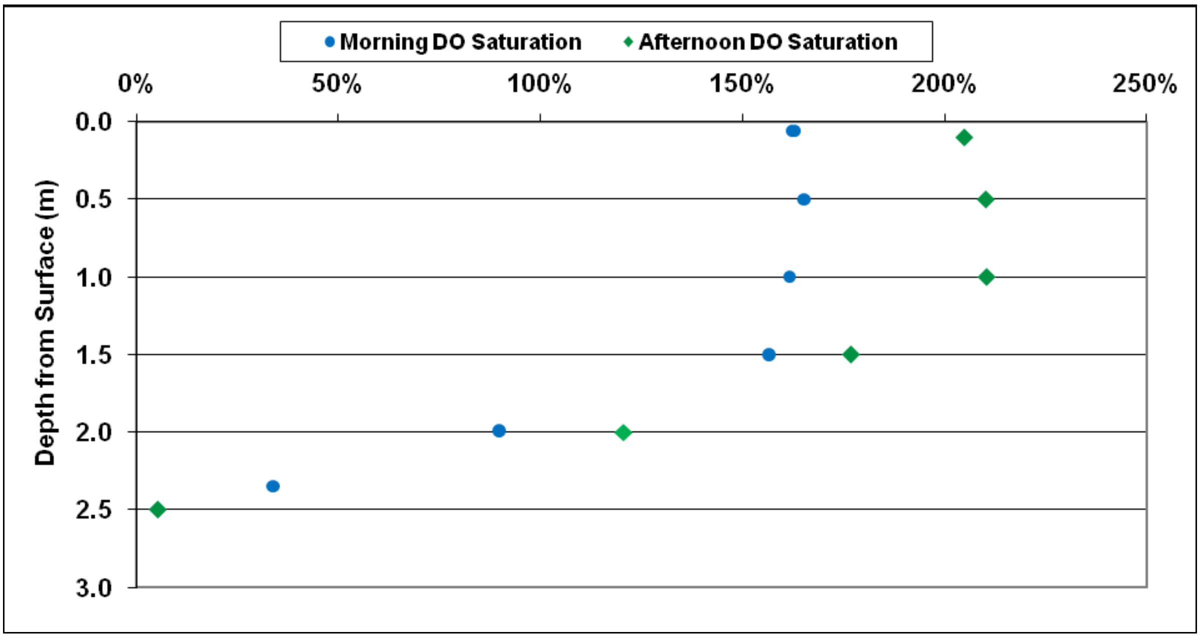

Figure G-35. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 10:30 a.m. and 3:00 p.m. on July 14, 2009

Table G-100. Calculated DO Saturation from Data Collected in Legg Lake 10 on July 14, 2009

Depth (m)	DO Saturation at 10:30 a.m.	DO Saturation at 3:00 p.m.
~0.1	162%	205%
0.5	165%	210%
1.0	162%	210%
1.5	157%	177%
2.0	90%	121%
~2.5	34%	5%

USEPA sampled North, Center, and Legg lakes on June 8, August 11, and September 29, 2010 (Table G-101). Secchi depth ranged from 0.4 m to 1.3 m. In-lake samples of TKN ranged from 0.57 to 1.4 mg-N/L. Ammonia samples ranged from 0.03 to 0.082 mg-N/L. Nitrate-nitrite concentrations were below the detection limit of 0.015 mg-N/L during the June event for all stations and the September events at all Legg 9 and 10 stations; nitrate-nitrite of 0.059 to 0.081 mg-N/L was observed at Legg 8 in September. During the August event, nitrate ranged from below the detection limit of 0.05 mg-N/L to 0.29 mg-N/L, and nitrite samples were below detection limits of 0.25 mg-N/L. All 2010 orthophosphate measurements were below the detection limit of 0.5 mg-P/L; total phosphorus concentrations ranged from 0.02 mg-P/L to 0.06 mg-P/L. Chlorophyll *a* concentrations ranged from 11 µg/L to 44 µg/L. The August and September chlorophyll *a* data represent estimated values as the samples were held past the holding times.

Table G-101. 2010 In-lake Water Column Measurements for North, Center, and Legg Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	NO_2-NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll a (µg/L)	Secchi Depth (m)
6/8/2010	LEGG-8	10:30	0.9	0.046	NM	NM	<0.015	<0.5	0.05	15	1.1
6/8/2010	LEGG-9	9:45	0.64	0.03	NM	NM	<0.015	<0.5	0.03	11	1.3
6/8/2010	LEGG-10	8:45	0.57	0.05	NM	NM	<0.015	<0.5	0.04	13	0.8
8/11/2010	LEGG-8	13:00	1.4	0.062	<0.25	0.29	NM	<0.5	0.03	36	NM
8/11/2010	LEGG-8 (Duplicate)	13:00	NM	NM	<0.25	0.28	NM	<0.5	NM	36[2]	NM
8/11/2010	LEGG-9	11:45	0.8	0.046	<0.25	<0.05	NM	<0.5	0.03	18[2]	NM
8/11/2010	LEGG-10	9:15	1.1	0.056	<0.25	0.09	NM	<0.5	0.02	44[2]	0.5
9/29/2010	LEGG-8	10:20	1.1	0.082	<0.25	<0.05	0.081	<0.5	0.06	40	0.5
9/29/2010	LEGG-8 (Duplicate)	10:20	1.2	0.080	<0.25	0.05	0.059	<0.5	0.06	35	0.5
9/29/2010	LEGG-9	11:00	0.99	0.068	<0.25	<0.05	<0.015	<0.5	0.06	24	0.5
9/29/2010	LEGG-10	8:45	1.3	0.082	<0.25	<0.05	<0.015	<0.5	0.05	42	0.4

[1]NM indicates that this value was not measured.
[2]The August chlorophyll *a* data represent estimated values as the samples were held past the holding times.

Ground water quality data for June 8, August 11, and September 29, 2010 are shown in Table G-102. These data represent groundwater quality after being treated and before entering the lake. Ammonia and nitrite concentrations in the groundwater input were similar to those in the lake. TKN in the groundwater samples ranged from below the detection limit of 0.05 to 0.14 mg-N/L, and nitrate ranged from 2.1 to 2.6. Nitrate-nitrite ranged from below the detection of 0.015 to 2.2 mg-N/L. Orthophosphate concentration of the groundwater was below the detection limit of 0.5 mg-P/L; total phosphorus ranged from 0.02 to 0.05 mg/L.

Supplemental water quality data were also collected during the three 2010 sampling events for the in-lake and groundwater sites. These data are shown in Table G-103.

Table G-102. 2010 Ground Water Quality Measurements for North, Center, and Legg Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	NO_2-NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)
6/8/2010	EPA-GW	11:15	<0.05	0.059	NM	NM	<0.015	<0.5	0.05
8/11/2010	EPA-GW	10:12	0.05	0.040	<0.25	2.6	NM	<0.5	0.02
	EPA-GW (Duplicate)		0.06	0.042	<0.25	2.6	NM	<0.5	0.02
9/29/2010	EPA-GW	14:40	<0.05	0.067	<0.25	2.1	2.2	<0.5	0.03
	EPA-GW (Duplicate)		0.14	0.060	<0.25	2.1	2.2	<0.5	0.03

Table G-103. 2010 Supplemental Water Quality Monitoring for In-lake Samples in the Legg Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
6/8/2010	LEGG-8	10:30	62	74	130	NM	360	5	4.5	5.8
6/8/2010	LEGG-9	9:45	83	92	150	NM	450	<5	4.3	4.5
6/8/2010	LEGG-10	8:45	83	94	120	NM	420	<5	4.4	3.8
6/8/2010	EPA-GW[2]	11:15	82	98	180	NM	480	<5	1.6	1.1
8/11/2010	LEGG-8	13:00	95	140	150	270	510	7	4.1	3.9
8/11/2010	LEGG-8 (Duplicate)	13:00	95	140	150	NM	NM	NM	NM	NM
8/11/2010	LEGG-9	11:45	100	130	150	250	490	ND	5.1	5.8
8/11/2010	LEGG-10	9:15	99	140	120	240	480	10	3.9	9.1
8/11/2010	EPA-GW	10:12	93	140	200	NM	650	<5	0.78	0.70
	EPA-GW (Duplicate)	10:12	93	140	NM	NM	NM	NM	NM	NM
9/29/2010	LEGG-8	10:20	95	140	150	270	540	6	4.1	3.9
9/29/2010	LEGG-8 (Duplicate)	10:20	95	140	150	250	NM	NM	NM	NM
9/29/2010	LEGG-9	11:00	100	130	150	240	530	5	5.1	5.8
9/29/2010	LEGG-10	8:45	99	140	120	250	500	11	3.9	9.1
9/29/2010	EPA-GW	14:40	93	140	200	280	580	<5	0.78	0.70
	EPA-GW (Duplicate)	14:40	93	140	NM	240	NM	NM	NM	NM

[1]NM indicates that this value was not measured.
[2]EPA-GW represents the groundwater input after treatment by the EPA facility, not an in-lake sample.

Depth-profile data were also collected during the three 2010 sampling events. As shown in Table G-104 and Table G-105, depth-profile data were collected during the morning and afternoon hours on June 8 and August 11, 2010. On September 29, depth-profile data were collected in the morning hours only due to equipment malfunction (Table G-106); specific conductivity was not measured during this sampling event.

Table G-104. Profile Data Collected in North, Center, and Legg Lakes (6/8/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
Legg-8	10:30	0.32	25.5	8.5	9.9	0.604	134
		0.19	25.6	8.5	10.1	0.607	134
		0.50	25.6	8.5	10.4	0.609	134

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		1.01	25.5	8.5	10.0	0.608	135
		1.50	25.3	8.3	9.3	0.617	138
		1.89	24.8	7.7	3.4	0.633	85
Legg-8	16:00	0.16	27.7	8.7	11.0	0.607	-4
		0.50	27.7	8.7	11.0	0.607	2
		1.01	26.3	8.6	11.3	0.637	10
		1.49	25.7	8.3	9.6	0.625	19
Legg-9	AM	0.15	25.9	8.2	7.6	0.768	172
		0.50	26.0	8.2	7.7	0.767	173
		1.00	26.0	8.3	8.0	0.766	174
		1.51	26.0	8.3	8.2	0.766	174
		2.00	25.6	8.2	7.8	0.767	175
		2.49	24.7	8.0	6.0	0.771	179
Legg-9	PM	0.19	27.7	8.5	8.9	0.765	-2
		0.50	27.7	8.5	9.1	0.765	0
		0.98	27.2	8.4	8.6	0.768	4
		1.50	26.2	8.3	8.3	0.767	8
		1.95	25.7	8.2	7.6	0.768	13
		2.39	25.1	7.9	5.8	0.771	0
Legg-10	8:45	0.16	25.6	8.4	9.3	0.726	216
		-0.01	24.4	13.2	8.5	0.002	-60
		0.51	25.6	8.5	9.2	0.726	215
		1.00	25.6	8.5	9.3	0.727	214
		1.50	25.6	8.5	9.3	0.727	213
		1.97	25.6	8.5	9.3	0.728	212
		2.08	25.3	7.9	5.1	0.738	216
Legg-10	14:45	0.08	27.3	8.6	9.7	0.73	120
		0.51	27.1	8.7	10.0	0.729	115
		1.02	26.2	8.7	10.4	0.732	116
		1.53	25.8	8.7	10.4	0.726	116
		2.02	25.2	8.2	7.5	0.748	122
		2.47	24.2	7.7	1.2	0.755	-144

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		2.61	23.8	7.3	1.2	0.732	Over Range
		2.31	24.5	8.0	3.5	0.753	-137
EPA-GW[1]	11:30	0.01	21.3	7.8	5.0	0.844	105
		0.01	21.4	7.7	4.7	0.846	105
		0.00	21.2	7.6	5.8	0.842	106

[1]EPA-GW represents the groundwater input after treatment by the EPA facility, not an in-lake sample.

Table G-105. Profile Data Collected in North, Center, and Legg Lakes (8/11/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
LEGG8	11:15	0.06	25.8	7.75	7.48	0.815	180
		0.46	25.77	7.73	7.43	0.816	180
		1.05	25.63	7.71	7.34	0.816	178
		1.52	25.32	7.65	6.59	0.818	176
		2.08	25.32	7.49	4.5	0.818	175
		2.53	25.26	7.66	5.57	0.817	71
LEGG8	16:45	0.13	28.43	8.16	8.79	0.818	161
		0.48	28.23	8.15	8.86	0.819	160
		1.01	27.25	8.13	8.95	0.816	159
		1.5	25.66	7.98	8.67	0.816	158
		2.04	25.38	7.81	7.37	0.816	158
		2.47	25.26	7.73	6.54	0.817	157
		2.74	25.16	7.59	4.54	0.818	-10
LEGG9	11:15	0.06	25.8	7.75	7.48	0.815	180
		0.46	25.77	7.73	7.43	0.816	180
		1.05	25.63	7.71	7.34	0.816	178
		1.52	25.32	7.65	6.59	0.818	176
		2.08	25.32	7.49	4.5	0.818	175
		2.53	25.26	7.66	5.57	0.817	71
LEGG9	16:45	0.13	28.43	8.16	8.79	0.818	161
		0.48	28.23	8.15	8.86	0.819	160
		1.01	27.25	8.13	8.95	0.816	159

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		1.5	25.66	7.98	8.67	0.816	158
		2.04	25.38	7.81	7.37	0.816	158
		2.47	25.26	7.73	6.54	0.817	157
		2.74	25.16	7.59	4.54	0.818	-10
LEGG-10	10:10	0.06	24.74	8.22	8.98	0.809	181
		0.44	24.75	8.17	9.01	0.809	183
		0.99	24.7	8.17	9.02	0.809	183
		1.49	24.65	8.15	9	0.808	184
		1.98	24.62	8.15	8.99	0.808	183
		2.5	24.54	7.47	4.07	0.834	188
		2.21	24.56	7.75	6.36	0.825	66
LEGG-10	16:00	0.04	26.7	8.39	10.52	0.808	156
		0.49	26.62	8.44	10.85	0.808	152
		0.99	24.94	8.23	10.48	0.813	152
		1.52	24.7	8.24	9.85	0.807	149
		1.98	24.62	8.14	9.31	0.808	149
		2.5	24.57	7.71	6.6	0.829	153

Table G-106. Profile Data Collected in North, Center, and Legg Lakes (9/292010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
Legg-8	10:20	0.5	24.2	8.6	9.9	74.0
		1.0	23.9	8.3	9.9	76.0
		1.5	23.4	8.2	10.0	77.3
		2.0	22.7	7.9	9.7	81.6
		2.5	22.4	7.7	5.1	18.0
Legg-9	11:00	0.5	24.6	8.9	10.0	28.4
		1.0	23.4	8.8	9.7	31.0
		1.5	23.3	8.1	8.5	45.0
		2.0	22.6	7.9	6.5	42.7
Legg-10	8:45	0.5	23.8	8.8	10.6	119.5

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
		1.0	23.8	8.8	10.5	117.4
		1.5	22.7	8.0	9.0	126.7
		2.0	22.2	7.8	6.2	119.0
		2.5	22.0	7.9	3.0	111.2

Figure G-36 through Figure G-38 display the depth-profile data for Legg 10 during these events. Dissolved oxygen saturation for Legg 10 is displayed in Figure G-39 through Figure G-41. Similar trends were observed compared to earlier profile sampling.

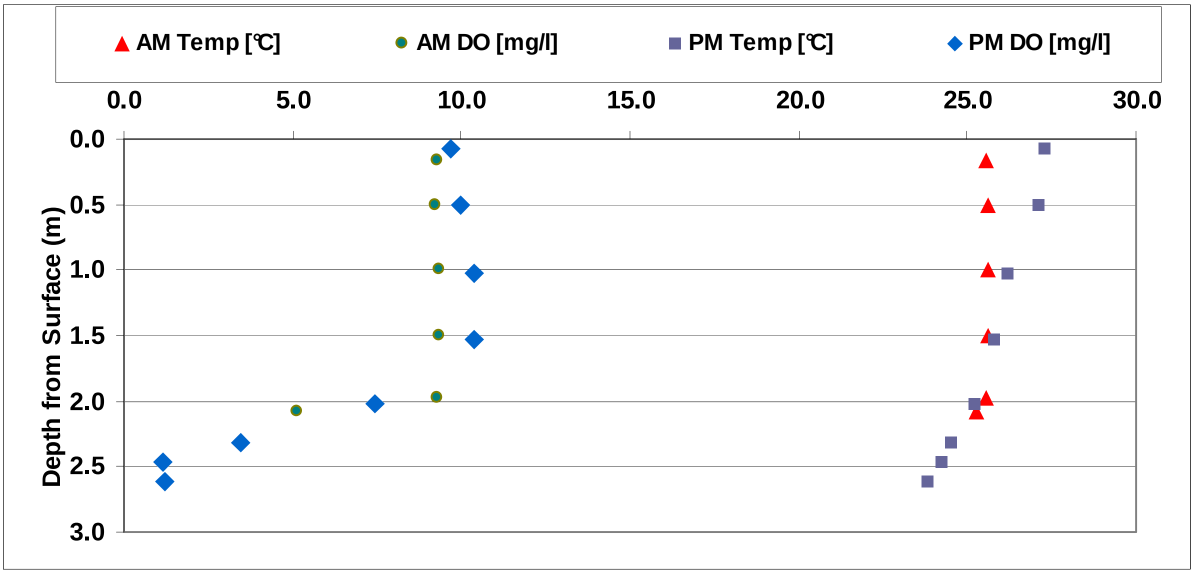

Figure G-36. Profile Data Collected in Lake Legg 10 at approximately 8:45 a.m. and 2:45 p.m. on June 8, 2010

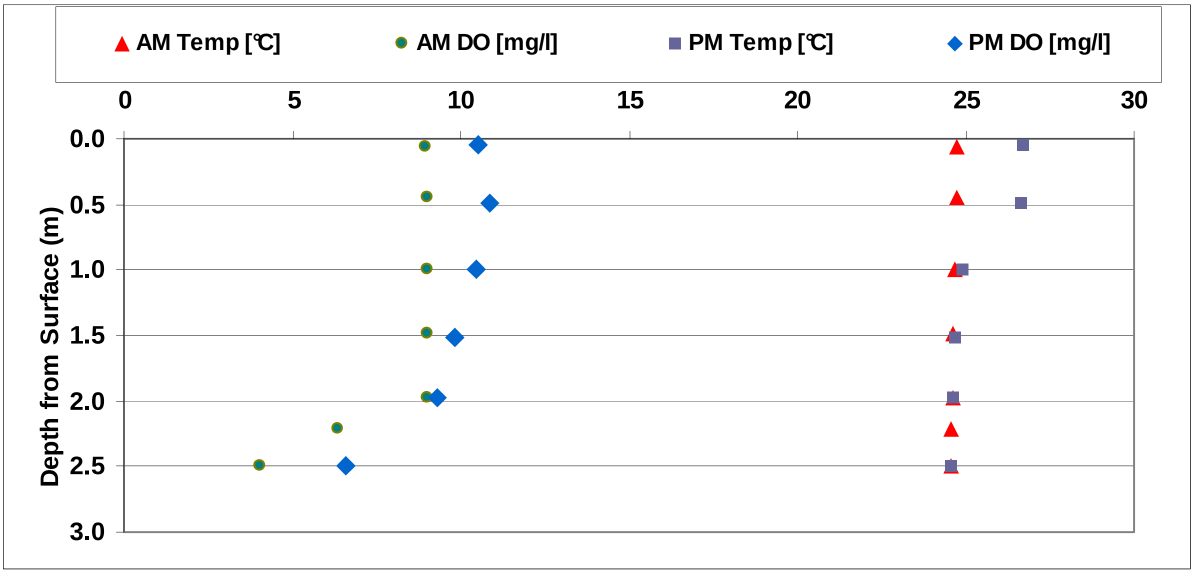

Figure G-37. Profile Data Collected in Lake Legg 10 at approximately 10:10 a.m. and 3:00 p.m. on August 11, 2010

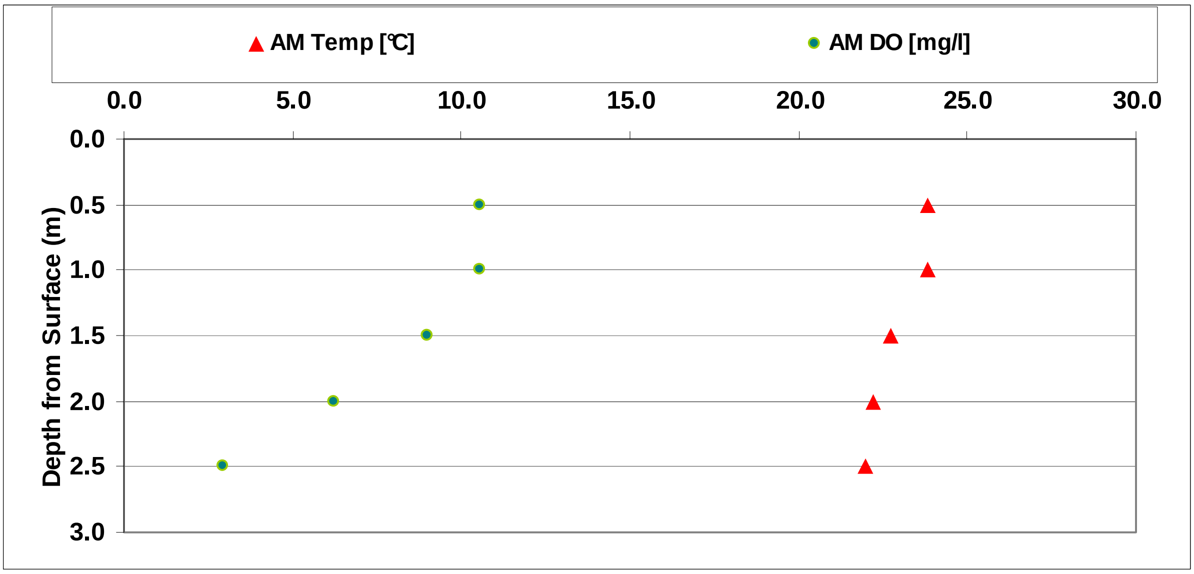

Figure G-38. Profile Data Collected in Lake Legg 10 at approximately 8:45 a.m. on September 29, 2010

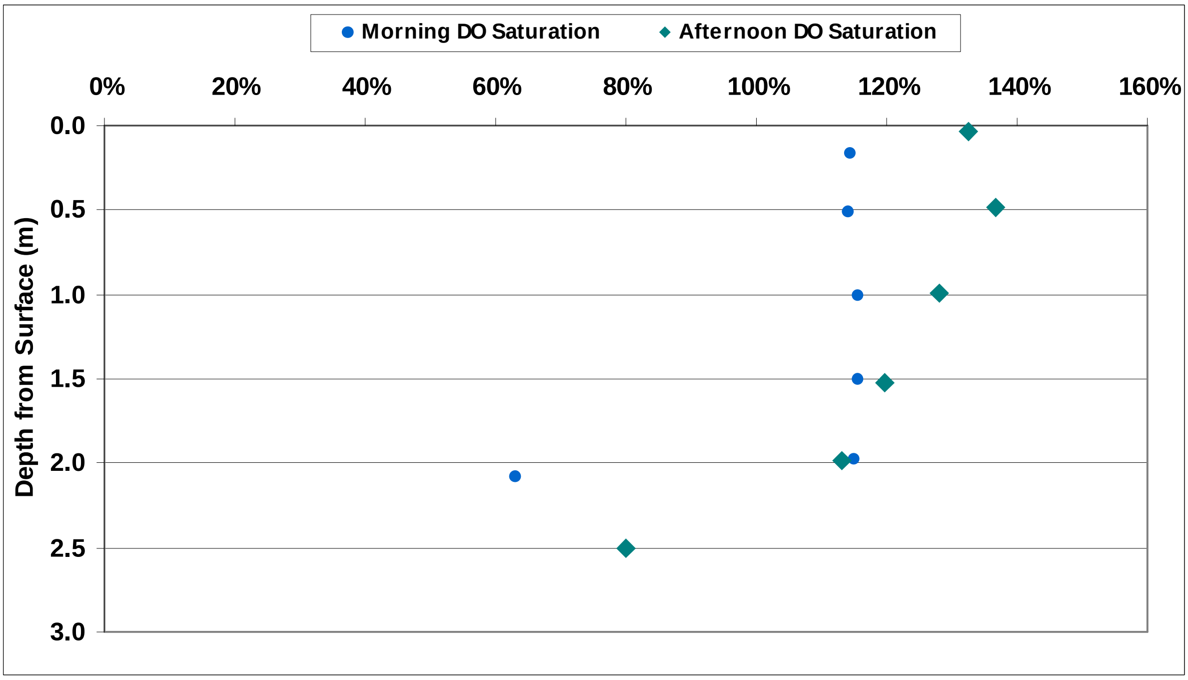

Figure G-39. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 8:45 a.m. and 2:45 p.m. on June 8, 2010

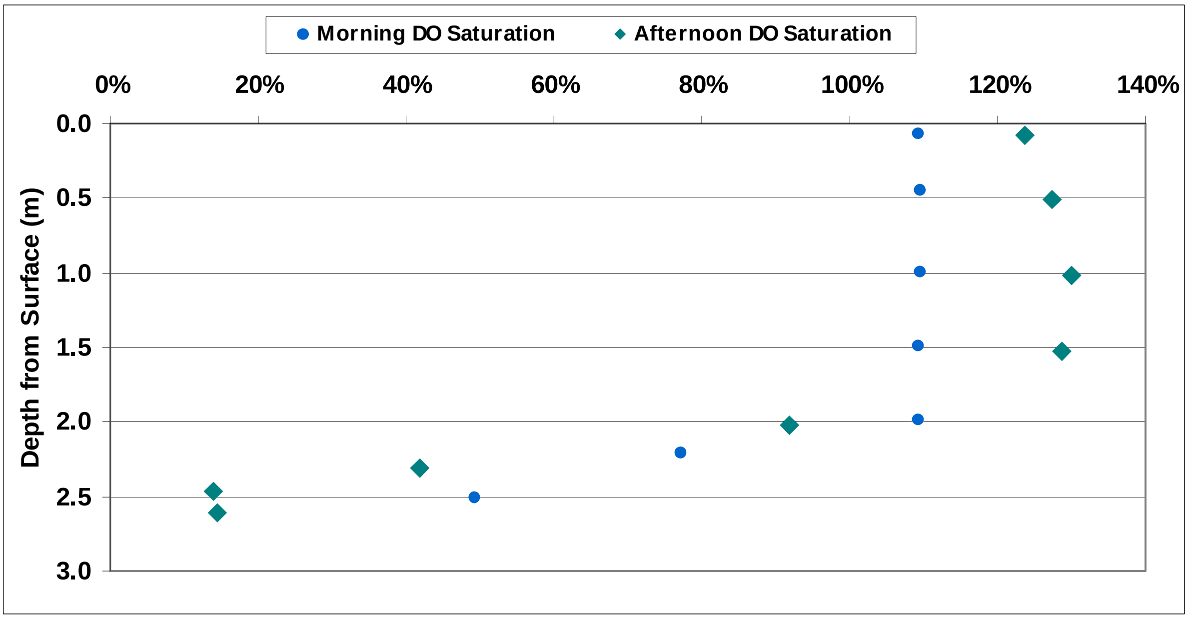

Figure G-40. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 10:10 a.m. and 3:00 p.m. on August 11, 2010

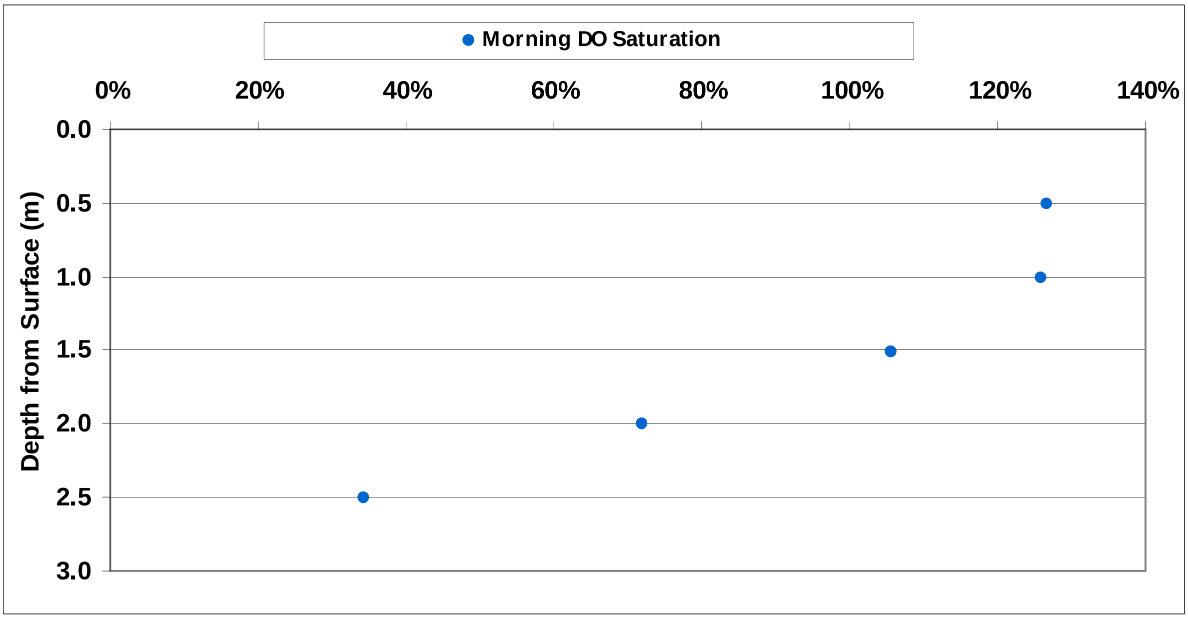

Figure G-41. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 8:45 a.m. on September 29, 2010

G.9.2 MONITORING RELATED TO METALS IMPAIRMENTS

Metals data collected at Legg Lake (pink triangle, Figure G-33), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-107. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to three meters. The range of the 43 dissolved copper samples was between less than 10 µg/L and 97 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to three meters. The 43 samples collected ranged in concentration from less than 1 µg/L to 70 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Legg Lake states that copper and lead were not supporting the assessed uses: 43 measurements had a maximum lead concentration of 70 µg/L, a maximum copper concentration of 97 µg/L, and a maximum zinc concentration of 134 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-107. Legg Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/6/1992	0	17	<1
	1.5	35	<1
	2.1	41	1
7/6/1992	0	16	N/A
	1.3	22	<1
	1.6	27	2
7/6/1992	0	18	1
	1.4	42	1
	1.8	26	<1
8/12/1992	0	27	5
	1.5	<10	1
	2.5	30	1
8/12/1992	0	32	7
	2	27	8
8/12/1992	0	41	14
	1.5	37	2
9/21/1992	0	<10	<1
	1.5	<10	<1
	2.5	<10	<1
10/8/1992	0	<10	<1
	1.5	<10	<1
	2.5	<10	<1
11/3/1992	0	<10	1
	1.5	15	1
	3	32	2
12/15/1992	0	<10	<1
	2	<10	<1
1/13/1993	0	<10	<1
	2	12	<1
2/3/1993	0	<10	<1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
3/4/1993	0	14	<1
	1.5	<10	18
	2.5	<10	27
4/13/1993	0	97	5
	1.5	84	<1
	2.5	78	<1
5/5/1993	0	<10	21
	2	<10	22
	3	<10	<1
6/8/1993	0	<10	70
	1.5	<10	28
	2.5	<10	17

On July 18, 2007, the county of Los Angeles contracted with AquaBio Environmental Technologies to perform sediment sampling near two storm drain inlets in North Lake (Table G-108).

Table G-108. July 18, 2007 County of Los Angeles Sediment Monitoring Data in North Lake

Station	Copper (mg/kg)	Lead (mg/kg)	Zinc (mg/kg)
North Lake near the east storm drain	<5.0	5.7	24
North Lake near the west storm drain	11	<5.0	31

Table G-109 presents 45 additional metal samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between February 2009 and September 2010 in North, Center, and Legg lakes. Samples were collected at locations LEGG-1, LEGG -2, LEGG -4, LEGG -5, LEGG -6, LEGG -8, LEGG -9 and LEGG -10. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.2 µg/L; copper concentrations were between <0.4 µg/L and 3.5 µg/L; lead concentrations ranged from <0.1 µg/L to 1.0 µg/L; and zinc concentrations were <0.1 µg/L to 14.5 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 68.05 mg/L to 280 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-109 were weighted more heavily in the assessment.

Table G-109. Metals Data for the 2008-2010 Legg Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	2/3/2009	LEGG 1/2	106.7	0.2	0.9	<0.1	0.4	average of Legg 1 replicates and Legg 2
RB	2/3/2009	LEGG 4	174.3	<0.2	1.1	0.2	1.2	average of duplicates
RB	2/3/2009	LEGG 5 / 6	181.55	<0.2	1.6	0.2	0.6	average of sites 5 & 6
RB/EPA	7/14/2009	LEGG 1/2	96	<0.3	0.6	<0.1	2.1	
RB/EPA	7/14/2009	LEGG 4	155.7	<0.2	0.6	0.1	3.3	
RB/EPA	7/14/2009	LEGG 5	138.5	<0.2	0.6	0.1	1.6	
RB/EPA	7/14/2009	LEGG 8	97.6	<0.2	0.5	0.1	<0.1	
RB/EPA	7/14/2009	LEGG 9	158.7	<0.2	0.5	0.1	<0.1	
RB/EPA	7/14/2009	LEGG 10	136.2	<0.2	0.5	0.1	<0.1	average of replicates and duplicate
County	12/8/2009	LEGG-1	133.75	<0.2	3.5	0.2	14.5	average of replicates
County	12/8/2009	LEGG-10	198.5	<0.2	0.5	<0.1	<0.1	
County	12/8/2009	LEGG-4	195.9	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	LEGG-6	202.1	<0.2	1.2	<0.1	<0.1	
County	12/8/2009	LEGG-8	155.4	<0.2	0.7	<0.1	0.9	
County	12/8/2009	LEGG-9	188	<0.2	0.5	<0.1	<0.1	average of duplicates
EPA	12/16/2009	LEGG-1	117.55	<0.2	2.3	0.2	11.3	average of replicates
EPA	12/16/2009	LEGG-10	211.1	<0.2	1.4	0.2	9.2	average of duplicates
EPA	12/16/2009	LEGG-4	166.2	<0.2	1.4	0.1	6.6	
EPA	12/16/2009	LEGG-6	200.6	<0.2	0.6	0.2	<0.1	
EPA	12/16/2009	LEGG-8	140.5	<0.2	1.1	0.1	2.7	
EPA	12/16/2009	LEGG-9	170.5	<0.2	1.3	0.1	10.9	
County	1/28/2010	LEGG-1	68.05	<0.2	1.8	0.1	<0.1	average of replicates
County	1/28/2010	LEGG-10	188	<0.2	0.5	0.1	<0.1	
County	1/28/2010	LEGG-4	118.6	<0.2	1.1	<0.1	<0.1	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
County	1/28/2010	LEGG-6	190.2	<0.2	0.6	0.1	<0.1	
County	1/28/2010	LEGG-8	82.4	<0.2	2	<0.1	<0.1	average of duplicates
County	1/28/2010	LEGG-9	121.8	<0.2	1	0.1	<0.1	
County	2/17/2010	LEGG-1	103.8	<0.2	0.9	0.06	0.2	
County	2/17/2010	LEGG-10	184.2	<0.2	0.6	0.16	0.8	
County	2/17/2010	LEGG-4	128.1	<0.2	0.8	0.06	3.55	average of replicates
County	2/17/2010	LEGG-6	184.9	<0.2	0.6	0.16	2.1	
County	2/17/2010	LEGG-8	79.1	<0.2	1.15	0.055	1.05	average of duplicates
County	2/17/2010	LEGG-9	129	<0.2	0.8	0.05	4	
EPA / RB	8/11/2010	LEGG-1	260	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-10	220	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-4	220	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-6	220	NA	2	<1	NA	
EPA / RB	8/11/2010	LEGG-8	240	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-9	220	NA	<1	<1	NA	
EPA / RB	9/29/2010	LEGG-1	240	NA	1.2	<1	NA	
EPA / RB	9/29/2010	LEGG-10	240	NA	1.7	<1	NA	
EPA / RB	9/29/2010	LEGG-4	250	NA	2.4	<1	NA	
EPA / RB	9/29/2010	LEGG-6	280	NA	1.9	<1	NA	
EPA / RB	9/29/2010	LEGG-8	270	NA	2.15	<1	NA	
EPA / RB	9/29/2010	LEGG-9	250	NA	2.4	<1	NA	

RB = Regional Board
EPA = USEPA
County = County Los Angeles

USEPA also collected three sediment samples during August 2010 to further evaluate lake conditions. Table G-110summarizes the lead and copper concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target and zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-110. Sediment Metals Data for the August 2010 Legg Lakes Sampling Event

Organi zation	Date	Station ID	Copper (µg/g)	Lead (µg/g)	Notes
EPA	08/11/2010	LEGG-8	135	76	Average of duplicates
EPA	08/11/2010	LEGG-9	110	60	
EPA	08/11/2010	LEGG-10	52	20	

G.10 Monitoring Data for Puddingstone Reservoir

Sampling has occurred intermittently from 1992 to 2009. In addition, fish tissue data are available for 1986 to 2007. Figure G-42 shows the locations of historic and recent monitoring.

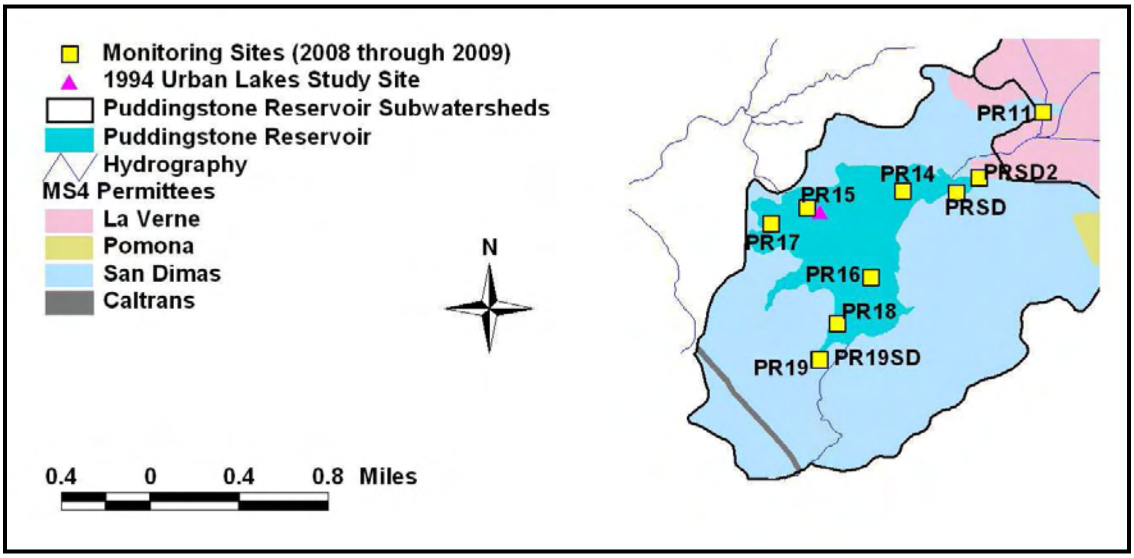

Figure G-42. Puddingstone Reservoir Monitoring Sites

G.10.1 Monitoring Related to Nutrient Impairments

Puddingstone Reservoir was monitored for water quality in 1992 and 1993 in support of the Urban Lakes Study near the center of the northern half of the lake (pink triangle, Figure G-42) (Table G-111). TKN ranged from 0.3 mg/L to 6.9 mg/L although concentrations greater than 1.2 mg/L only occurred at depths greater than or equal to 8 meters. Ammonium ranged from 0.1 mg/L to 5.3 mg/L with 39 measurements less than the reporting limit; concentrations did not exceed 0.2 mg/L except at depths greater than or equal to 8 meters. The upper range of these concentrations are above both the chronic and acute targets (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Each of the 75 measurements of nitrite was less than the reporting limit, and 23 nitrate samples were less than the reporting limit. The maximum concentration of nitrate observed was 2 mg/L. Forty-nine of 75 samples of orthophosphate were less than the reporting limit, and the maximum concentration observed was 1.7 mg/L. Total phosphorus was similar with 45 measurements less than the reporting limit and a maximum observed concentration of 1.3 mg/L. Concentrations of neither orthophosphate nor total phosphorus exceeded 0.2 mg/L except at depths greater than or equal to 14 meters. pH ranged from 7.4 to 9.0, and TOC ranged from 2.8 mg/L to 8.2 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 4 µg/L to 22 µg/L with an average of 13 µg/L; however, the raw data have not been located.

Table G-111. Puddingstone Reservoir 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/28/1992	0	0.8	<0.01	<0.01	0.2	<0.01	<0.01	7.5	6.7	225
	3	0.8	<0.01	<0.01	0.1	<0.01	<0.01	8.1	5.4	224
	6.5	0.6	<0.01	<0.01	0.1	<0.01	<0.01	7.9	8.2	228
	10	0.9	0.2	<0.01	0.1	<0.01	<0.01	7.8	5.9	231
	13.5	1.3	0.8	<0.01	0.1	0.2	0.2	7.7	5.9	222
	17	2.9	2.5	<0.01	0.1	0.9	0.8	7.5	5.6	235
7/28/1992	0	1	0.1	<0.01	0.1	<0.01	<0.01	8.6	6.4	228
	2	1.1	<0.01	<0.01	0.1	<0.01	<0.01	8.7	5.6	225
	4.5	0.8	<0.01	<0.01	0.1	<0.01	<0.01	8.4	6.9	236
	7	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8	6.6	223
7/28/1992	0	1	<0.01	<0.01	0.1	<0.01	<0.01	8.7	6.1	229
	2.5	1.2	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	6.3	236
	5.5	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8.2	6.4	229
	7.5	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8	8.2	239
	9.5	0.9	0.3	<0.01	<0.01	0.1	<0.01	7.9	6.7	238
9/1/1992	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.1	6.2	190
	4	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.3	181
	8	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	204
	11	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.1	6.1	197
	14	2.8	1.9	<0.01	<0.01	0.6	0.5	7.6	6.1	211
	17	5.1	4	<0.01	<0.01	1.3	1.3	7.4	6.4	201
10/6/1992	0	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	6.3	185
	4	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	6.1	216
	8	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	202
	11	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	185
	14	0.8	0.3	<0.01	<0.01	<0.01	<0.01	7.8	6.6	220
	17	6.9	5.3	<0.01	<0.01	1.7	1.1	7.4	6.7	193
11/5/1992	0	0.8	0.2	<0.01	0.4	0.1	0.1	7.9	6	197
	3	0.8	0.1	<0.01	0.4	<0.01	0.1	7.9	6	188
	6	0.7	0.2	<0.01	0.4	<0.01	0.1	7.9	5.9	204
	9	1.1	0.2	<0.01	0.4	0.1	0.2	7.9	6.1	186

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	12	0.9	0.2	<0.01	0.4	0.1	0.1	7.9	5.9	195
	15	0.9	0.3	<0.01	0.4	0.1	0.1	7.9	5.9	200
12/17/1992	0	1.1	0.1	<0.01	0.6	<0.01	<0.01	7.9	5.7	211
	3.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	209
	6.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	6.3	205
	9.8	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	210
	12.5	0.7	<0.01	<0.01	0.5	<0.01	<0.01	7.9	5.8	209
	16.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	207
1/27/1993	0	0.9	0.2	<0.01	2	0.1	0.2	7.9	4.8	196
	5	0.9	0.2	<0.01	2	0.1	0.2	7.9	4.6	198
	10	0.8	0.2	<0.01	2	0.1	0.2	7.9	4.8	199
	15	0.8	0.2	<0.01	2	0.1	0.2	7.9	4.5	208
	20	1	0.3	<0.01	1.9	0.1	0.2	7.9	4.7	216
	25	1.2	<0.01	<0.01	1.8	0.2	0.2	7.9	5.1	194
2/11/1993	0	1.2	<0.01	<0.01	1.7	0.1	0.2	8.1	4.4	181
	3	0.9	0.1	<0.01	1.8	0.2	0.2	8.1	3.9	177
	6	0.6	0.1	<0.01	1.7	0.1	0.2	8.1	4.1	187
	9	0.6	0.1	<0.01	1.7	0.1	0.1	8.1	4.4	186
	12	0.7	0.1	<0.01	1.7	0.1	-	8.1	4.2	208
	15	0.6	0.2	<0.01	1.7	0.2	0.2	8.1	4.4	193
3/11/1993	0	0.6	<0.01	<0.01	1.4	<0.01	<0.01	8.6	3.6	221
	2	0.6	<0.01	<0.01	1.4	<0.01	<0.01	8.7	3.3	225
	5	0.4	<0.01	<0.01	1.7	<0.01	<0.01	8.3	3.2	228
	10	0.5	0.1	<0.01	1.7	<0.01	<0.01	8.2	3	224
	15	0.3	0.2	<0.01	1.6	<0.01	<0.01	8.1	2.8	223
	19	0.3	0.2	<0.01	1.6	<0.01	<0.01	8.2	2.9	233
4/14/1993	0	1.1	<0.01	<0.01	0.7	<0.01	<0.01	9	4.4	236
	2	1	<0.01	<0.01	0.7	<0.01	<0.01	8.9	3.5	230
	6	0.3	<0.01	<0.01	1.1	<0.01	<0.01	8.1	2.8	237
	10	0.5	<0.01	<0.01	1.5	<0.01	<0.01	8	2.8	245
	14	0.4	<0.01	<0.01	1.6	<0.01	<0.01	7.9	2.9	224
	17	0.5	<0.01	<0.01	1.8	<0.01	<0.01	7.9	2.9	211

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
5/11/1993	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	4.1	213
	5	1.1	0.1	<0.01	<0.01	<0.01	<0.01	8.2	3.6	218
	8	1.3	0.3	<0.01	<0.01	<0.01	<0.01	7.9	3.9	231
	11	1.4	0.2	<0.01	0.8	<0.01	<0.01	7.9	3.4	235
	14	0.8	0.3	<0.01	0.9	0.1	<0.01	7.9	3.6	231
	18	1.3	0.7	<0.01	0.3	0.3	0.3	7.9	3.8	227
6/10/1993	0	1.2	<0.01	<0.01	<0.01	<0.01	0.2	8.9	4.6	238
	3.5	1.1	<0.01	<0.01	<0.01	<0.01	0.1	8.5	3.7	218
	7.5	0.8	<0.01	<0.01	<0.01	<0.01	0.1	8.3	4	224
	10	0.8	<0.01	<0.01	<0.01	<0.01	0.1	8.2	3.6	242
	14	0.5	0.4	<0.01	0.3	0.2	0.1	8	3.7	236
	17	1.5	1.1	<0.01	<0.01	0.4	0.4	8	3.7	234

The 1996 Water Quality Assessment Report does contain summary information regarding the DO impairment which was listed as not supporting the aquatic life use. DO was measured 187 times with concentrations ranging from 0.1 mg/L to 14.9 mg/L. However, the accompanying database does not contain these measurements so no information regarding location, time, depth, or temperature can be compared. There are some temperature and pH measurements in the database that were collected from December 1977 through March 1978. Temperature ranged from 11.1 ºC to 11.7 ºC, and pH ranged from 6.6 to 7.6.

More recent monitoring of nutrients in Puddingstone Reservoir occurred on November 18, 2008 at four locations as well as one site located on Live Oak Wash above the mouth (Figure G-42). All samples of ammonia, TKN, nitrate, nitrite, orthophosphate, total phosphate, and total suspended solids collected at the four lake stations were below the reporting limits of 0.1 mg/L, 1 mg/L, 0.1 mg/L, 0.1 mg/L, 0.4 mg/L, 0.5 mg/L, and 10 mg/L, respectively. Total dissolved solids in the lake ranged from 217 mg/L to 251 mg/L. At the Live Oak Wash site (PR11), the following concentrations were observed: ammonia 0.215 mg/L, TKN 1.87 mg/L, TSS 50 mg/L, TDS 761 mg/L, nitrate 3.31 mg/L, and nitrite 0.131 mg/L. Samples of orthophosphate and total phosphate at this site were less than the reporting limit. Chlorophyll *a* ranged from 11.3 µg/L to 21.4 µg/L.

Field data for the November 2008 monitoring event are summarized in Table G-112. The sampling pump broke after sampling at site PR-15. Water quality samples at the other three sites were collected approximately 4 inches below the surface of the water.

Table G-112. Field Data for the November 18, 2008 Monitoring Event at Puddingstone Reservoir

Station	Time	Depth (m)	Temperature ºC	pH	Secchi Depth (m)
PR-14	13:15	Surface	18.5	8.3	1.2
PR-15	11:00	0.4	18.5	8.5	5.6
		3.0	18.5	8.2	
		6.1	18.0	8.0	
PR-16	15:20	Surface	18.0	8.3	Not reported
PR-17	16:30	Surface	17.5	8.3	Not reported

Puddingstone Reservoir was sampled in February and July in 2009 by USEPA and the Regional Board. The field notes report that approximately 300 gallons of chlorine are pumped into the swim beach area each week during the summer. The edges of the lake are sometimes treated for weeds. The location of the swim beach is not reported. Table G-113 lists the nutrient related measurements collected at stations PR-15 and PR-16 in Puddingstone Reservoir. Samples were collected from a depth of 1.5 meters at locations PR-15 and PR-16 in the winter, and sampled at a depth of 0.35 meters at both locations in the summer. Secchi depths were 0.76 meters at all locations in the winter and 0.71 meters at all locations sampled in the summer. Ideally, samples would have been collected from half the Secchi depth, rather than twice the Secchi depth to reflect average conditions over the photic zone. The summer samples were collected in this area, but winter samples were collected below the photic zone. Nitrogen species had relatively low concentrations at both locations in both seasons. Total phosphorus was slightly elevated with an average concentration of 0.11 mg/L in February and 0.08 mg/L in July. Chlorophyll *a* measurements were relatively high and ranged from 66.1 µg/L to 113.5 µg/L in February. The summer chlorophyll *a* levels were much less, at an average of 26.2 µg/L.

Table G-113. 2009 Water Quality Monitoring at Puddingstone Reservoir

Date	Sample Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2/24/2009	PR-15	10:10	1.7	0.04	0.05	0.26	0.062	0.121	113.5	0.76
	PR-15 (dup.)	10:30	1.3	0.03	0.02	0.02	0.016	0.114	94.8	0.76
	PR-16	12:15	1.3	0.03	0.02	0.02	0.016	0.098	66.1	0.76
7/16/2009	PR-15	9:00	0.98	<0.03	<0.01	<0.01	<0.0075	0.041	27.3	0.66
	PR-16	10:00	1.1	<0.03	<0.01	<0.01	<0.0075	0.164	25.1	0.71
	PR-16 (dup.)	10:00	1.1	<0.03	NA	NA	NA	0.048	NA	0.71

Supplemental water quality samples were collected from Puddingstone Reservoir. Table G-114 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Sites 15 and 16.

Table G-114. Supplemental Water Quality Monitoring for In-lake Samples in Puddingstone Reservoir

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Bicarb-onate (mg/L)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/24/2009	PR-15	10:10	40.1	30.2	114	102	224	10.3	5.8	5.8
	PR-15 (dup.)	10:30	40.0	30.3	120	112	118	NA	5.7	6.6
	PR-16	12:15	40.3	30.8	114	98	196	12.7	6.0	6.2
7/16/2009	PR-15	9:00	47.1	34.6	90	NA	258	NA	6.0	7.6
	PR-16	10:00	47.0	34.4	92	NA	246	NA	6.1	8.0

Profile data collected at stations PR-15 and PR-16 on February 24, 2009 are shown in Figure G-43 and Figure G-44, respectively. Measurement depths were limited by cable length to approximately 13 meters. Specific conductivity is constant with depth at both locations. Over 3 to 4 meters of depth, DO decreases from over 6 mg/L at the surface to 0 mg/L. pH ranges from 7.6 to 9.4 at each station. Temperature at these two stations ranges from 11.3 ºC to 14.6 ºC. **Note that field operators found DO readings suspicious and have since sent meter off for repair (Greg Nagle, USEPA Region IX, personal communication, 5/22/09).**

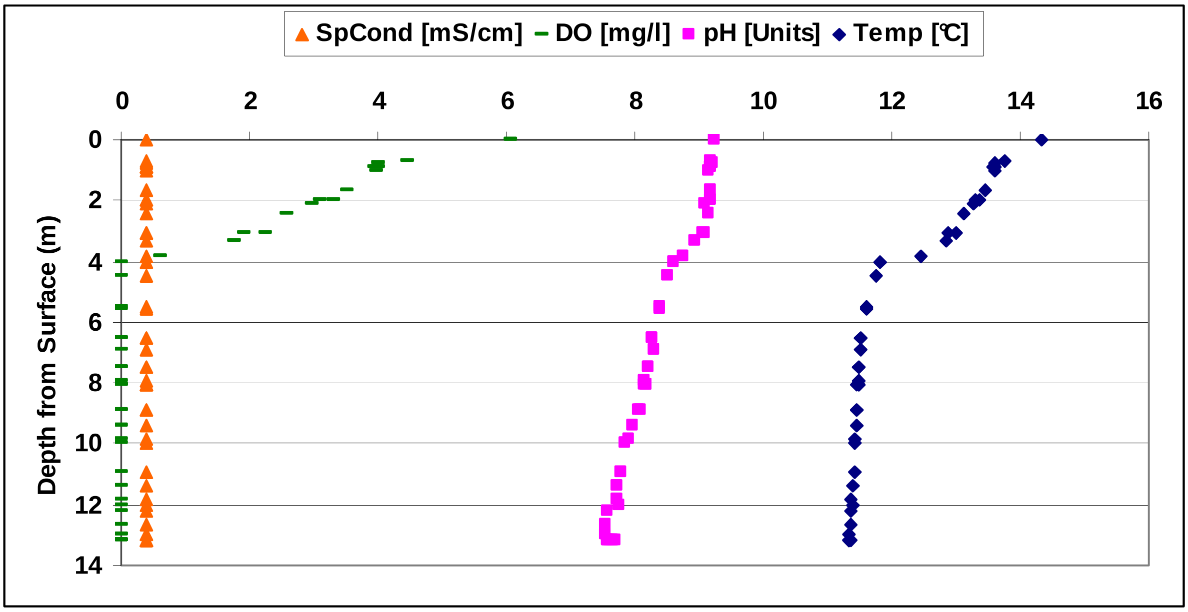

Figure G-43. Profile Data Collected at PR-15 on February 24, 2009

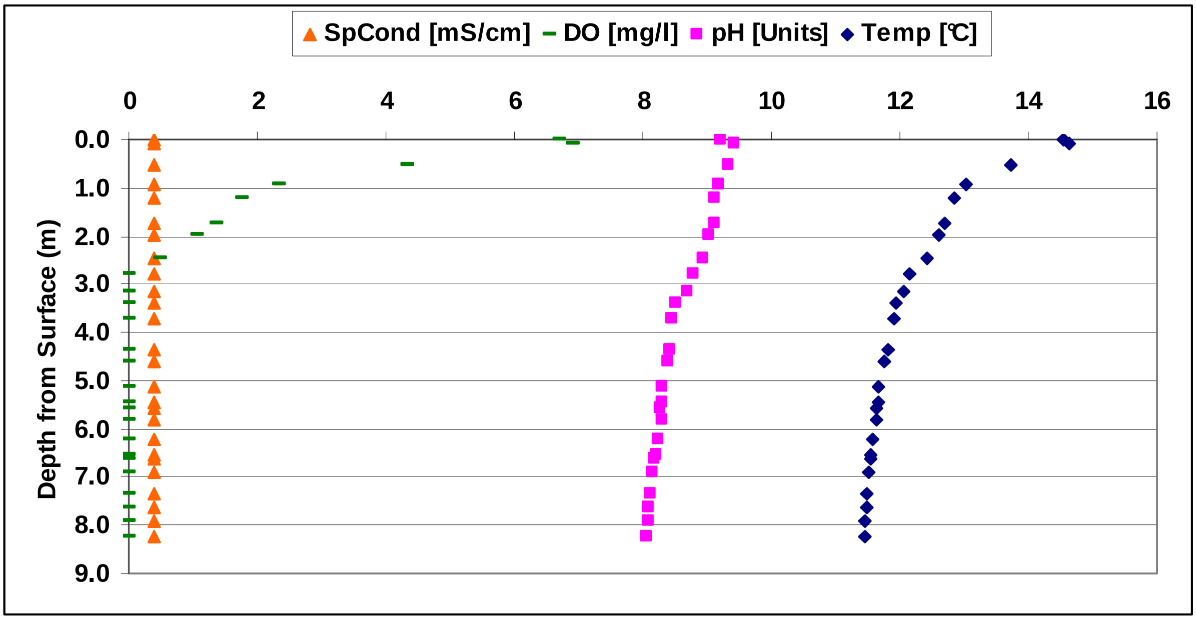

Figure G-44. Profile Data Collected at PR-16 on February 24, 2009

Profile data were also collected on the stations on July 16, 2009, shown in Figure G-45 and Figure G-46. Specific conductivity is constant with depth at both locations, similar to the data collected in January. The DO decreased between 3 and 4 meters of depth, although it remained around 1.8 mg/L instead of dropping to 0 mg/L as it did in January. Temperature at these two stations ranges from 13.0 ºC to 27.1 ºC. The pH ranges from 7.6 to 8.9 at each station.

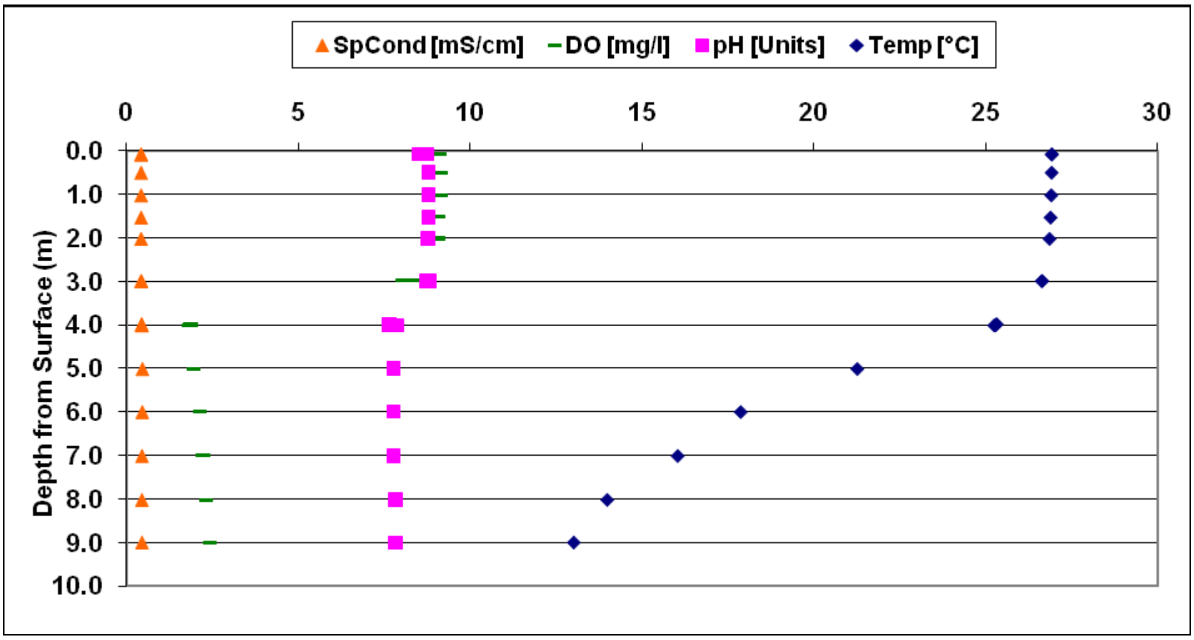

Figure G-45. Profile Data Collected at PR-15 on July 16, 2009

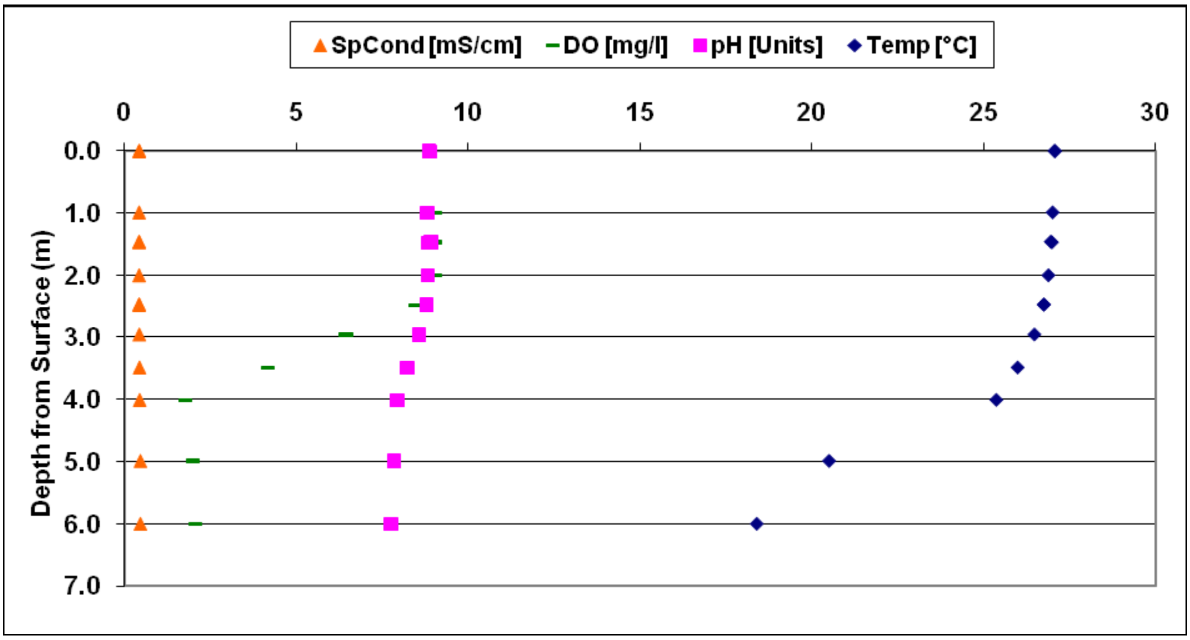

Figure G-46. Profile Data Collected at PR-16 on July 16, 2009

G.10.2 MONITORING RELATED TO MERCURY IMPAIRMENT

Mercury data have been collected in the Puddingstone Reservoir watershed since 1986. Fish tissue concentrations were measured four times under the TSMP from 1986 to 1999. The San Gabriel Watershed Council (SGWC) collected fish tissue measurements in 2006 and 2007, and the Regional Board collected samples in 2004 and 2007 (Davis et al., 2008). In-lake water column concentrations were measured as part of the Urban Lakes Study (UC Riverside, 1994) in 1992. The Regional Board sampled in-lake and tributary water column and sediment mercury concentrations in 2008 and 2009.

G.10.2.1 In-Lake Water Quality Monitoring

G.10.2.1.1 Water Column Measurements

In-lake water column mercury concentrations were measured in July and September 1992 as part of the Urban Lakes Study. All 21 measurements were less than the detection limit of 0.5 µg/L (500 ng/L). As the detection limit of this dataset is 10 times higher than the water quality criterion for mercury (50 ng/L), it is difficult to assess compliance in terms of a water column concentration.

In November 2008, the Regional Board sampled Puddingstone Reservoir for total mercury concentrations in the water column. Water column concentrations ranged from 1.2 ng/L to 1.6 ng/L and were more than one order of magnitude less than the water quality standard. Duplicates were measured at two sites. Samples were processed with EPA method 1631Em, which has a minimum detection limit of 0.5 ng/L.

In February 2009, the Regional Board and USEPA sampled two lake stations for both total and methylmercury. Station PR15 was sampled at a depth of 15.24 meters; the total depth at this location was 17.68 meters. Station PR16 was sampled at a depth of 1.5 meters; the total depth was 8.23 meters. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; concentrations ranged from 1.67 ng/L to 2.52 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L; concentrations ranged from 0.081 ng/L to 0.127 ng/L. The percent of mercury in the methyl form ranged from 4.8 to 5.2.

These two stations were resampled in July 2009. Both samples were collected from a depth of approximately 0.34 m. Concentrations of methyl and total mercury at both stations were less than those observed during the winter sampling event. Methylmercury ranged from 0.025 ng/L to 0.027 ng/L and total mercury ranged from 0.34 ng/L to 0.35 ng/L. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; methyl mercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Table G-115 presents the water column sampling results for Puddingstone Reservoir.

Table G-115. In-lake Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
PR14	11/18/2008	13:15	NA	1.5	NA
PR15		11:00	NA	1.6	NA
PR16		15:20	NA	1.2	NA
PR17		16:30	NA	1.4	NA
PR17 (duplicate)		16:30	NA	1.4	NA
PR15	2/24/2009	10:10	0.127	2.44	10.3
PR15 (duplicate)		10:10	NA	2.52	NA
PR16		12:15	0.081	1.67	12.6
PR15	7/16/2009	9:00	0.027	0.35	9.3
PR16		10:15	0.025	0.34	9.3
PR16 (duplicate)		10:15	NA	0.26	NA

Supplemental water quality data were also collected during the February 2009 event. Table G-116 summarizes the results. Note that the sampling depth for the supplemental data collected at PR-15 was 5 ft, which is different than the sampling depth used to obtain the mercury and TSS measurements.

Table G-116. Supplemental Water Quality Monitoring for In-lake Samples in Puddingstone Reservoir

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Bicarb-onate (mg/L))	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/24/2009	PR15	10:00	40.1	441.8	117	107	221	5.75	4.20
	PR16	12:15	40.3	4,233.5	114	98	196	6.0	4.95
	PRSD	13:10	252.9	162.1	218	218	NA	12.5	10.80
	PR11	14:30	166.9	88.6	122	66	610	6.5	2.96

G.10.2.1.1 Sediment Samples

In February 2009, the Regional Board and USEPA sampled sediment mercury concentrations at two locations in Puddingstone Reservoir. Total mercury concentration in the sediment samples ranged from 125 µg/kg to 165 µg/kg on a dry weight basis. Methylmercury concentrations ranged from 0.263 µg/kg to 0.502 µg/kg in the sediments. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.61 µg/kg to 4.72 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.020 µg/kg to 0.021 µg/kg. Detection limits were adjusted to account for sample aliquot size.

In July 2009, the Regional Board and USEPA resampled sediment mercury concentrations at these two locations. Total mercury concentration in the sediment samples ranged from 121 µg/kg to 145 µg/kg on a dry weight basis. Methylmercury concentrations ranged from 0.246 µg/kg to 0.330 µg/kg in the sediments. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.24 µg/kg to 4.12 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.030 µg/kg to 0.037 µg/kg. Detection limits were adjusted to account for sample aliquot size.

Table G-117 shows the sediment mercury concentrations measured in Puddingstone Reservoir.

Table G-117. In-lake Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR15	2/24/2009	11:00	0.502	165	42.46	859.96
PR15 (duplicate)		11:00	NA	136	26.65	846.61
PR16		12:45	0.263	125	39.73	816.26
PR15	7/16/2009	9:00	0.246	121	23.48	34.56
PR16		10:15	0.330	125	28.78	34.42
PR16 (duplicate)		10:15	NA	145	31.39	NA

G.10.2.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in Puddingstone Reservoir since 1986 by the TSMP, SGWC, and SWAMP. Table G-118 presents the fish tissue mercury concentrations on a wet weight basis. Concentrations range from 0.114 ppm to 0.744 ppm. Twelve individual common carp ranging in length from 395 mm to 687 mm were also analyzed for mercury during the 2004 sampling. Mercury concentrations ranged from 0 ppm to 0.092 ppm and were not considered in the fish tissue versus length mercury regression analysis as a conservative assumption. The applicable fish tissue guideline for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-118. Largemouth Bass Fish Tissue Mercury Concentrations Measured in Puddingstone Reservoir

Program	Date	Number in Sample	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	5/6/1986	6	302	0.200
TSMP	6/11/1991	6	380	0.510
TSMP	4/28/1992	6	386	0.420
TSMP	8/10/1999	6	345	0.371
SWAMP	9/22/2004	1	465	0.449
SWAMP	9/22/2004	1	349	0.365
SWAMP	9/22/2004	1	390	0.311
SWAMP	9/22/2004	1	429	0.39
SWAMP	9/22/2004	1	355	0.384
SWAMP	9/22/2004	1	380	0.369
SWAMP	9/22/2004	1	311	0.152
SWAMP	9/22/2004	1	324	0.271
SWAMP	9/22/2004	1	326	0.149
SWAMP	9/22/2004	1	374	0.228
SWAMP	9/22/2004	1	430	0.292
SWAMP	9/22/2004	1	520	0.499
SGWC	11/2/2006	5	150	0.328
SGWC	6/6/2007	16	350	0.224
SWAMP	Summer 2007	1	365	0.744
SWAMP	Summer 2007	1	375	0.451
SWAMP	Summer 2007	1	385	0.713
SWAMP	Summer 2007	1	351	0.346
SWAMP	Summer 2007	1	370	0.417
SWAMP	Summer 2007	1	367	0.463
SWAMP	Summer 2007	1	387	0.623
SWAMP	Summer 2007	1	371	0.311
SWAMP	Summer 2007	1	317	0.229
SWAMP	Summer 2007	1	365	0.532
SWAMP	Summer 2007	1	432	0.723
SWAMP	Summer 2007	1	598	0.535

Program	Date	Number in Sample	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	1	258	0.253
SWAMP	Summer 2007	1	255	0.158
SWAMP	Summer 2007	1	220	0.114
SWAMP	Summer 2007	1	200	0.115

Figure G-47 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length. As expected, fish tissue mercury concentrations increase with length. Concentrations exceed 0.22 ppm in all individual or composite samples greater than 345 mm. Twenty-three individual and five composite samples exceed the fish tissue target; five individual samples and one composite had concentrations less than the target.

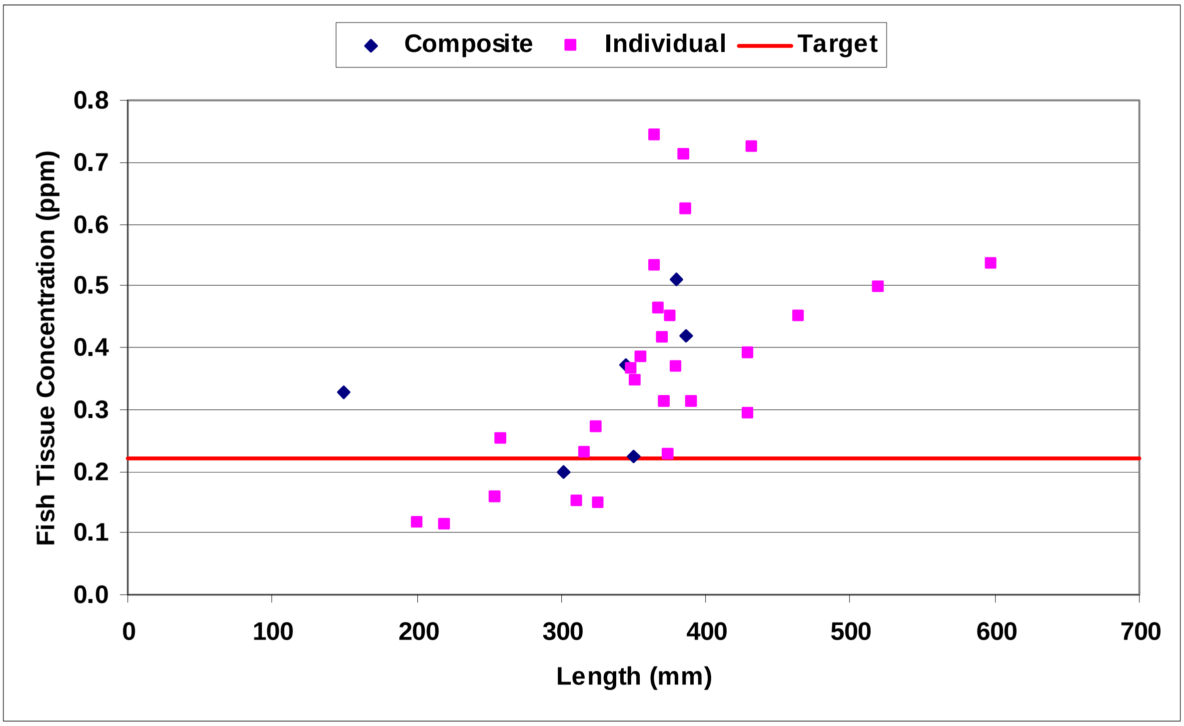

Figure G-47. Mercury Concentrations in Largemouth Bass in Puddingstone Reservoir

G.10.2.3 Tributary/Inflow Monitoring

G.10.2.1.3 Water Column Measurements

In February 2009, USEPA and the Regional Board sampled Live Oak Wash and the storm drain near the campground for total and methylmercury in the water column. The total mercury concentration measured from these two inputs ranged from 2.65 ng/L to 3.52 ng/L. Total mercury was analyzed with EPA Method 1631 with a detection limit of 3.03 ng/L. Methylmercury concentrations ranged from less than the detection limit to 0.043 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. The percent of mercury in the methyl form was 1.2 percent in the sample where methylmercury was greater than the detection limit.

Inflow water column measurements were collected again in the summer of 2009. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; methyl mercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. Concentrations of methyl and total mercury in Live Oak Wash were 0.553 ng/L and 4.24 ng/L, respectively. Concentrations measured in PRSD2 were 0.046 ng/L and 7.55 ng/L, respectively. [Note that storm drain PRSD was not flowing during this sampling event.]

Table G-119 shows the tributary and storm drain water column measurements for Puddingstone Reservoir.

Table G-119. Tributary/Inflow Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
PR11	2/24/2009	14:30	0.043	3.52	5.8
PRSD		13:10	<0.020	2.65	5.7
PR11	7/16/2009	11:45	0.553	4.24	3.6
PRSD2		13:10	0.046	7.55	3.8

G.10.2.1.3 Sediment Samples

During the February 2009 monitoring event, a sediment sample was collected from Live Oak Wash (PR11). The storm drain near the campground (PRSD) was not sampled for sediment because the only solid material evident at the discharge was leaves. The total mercury concentration of the Live Oak Wash sample was 52.9 µg/kg. The methylmercury concentration was less than the detection limit . Total mercury was analyzed with EPA Method 1631 with a detection limit of 2.61 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.011 µg/kg.

In July 2009, sediment samples were collected from four inlet locations. Concentrations of methyl and total mercury at Live Oak Wash were 1.71 µg/kg and 73.1 µg/kg, respectively. Concentrations were also measured in the overland flow ditch (PR19) and storm drain (PR19SD) present on the south side of the reservoir. Concentrations of methyl and total mercury from the ditch were 0.068 µg/kg and 34.3 µg/kg. the storm drain had concentrations of 0.940 µg/kg and 66.2 µg/kg, respectively. Concentrations measured at PRSD2 were 1.14 µg/kg and 50.4 µg/kg, respectively. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 1.28 µg/kg to 3.03 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.011 µg/kg to 0.025 µg/kg. Detection limits were adjusted to account for sample aliquot size.

Table G-120 presents the sediment concentrations measured in the inputs to Puddingstone Reservoir. Concentrations are reported on a dry weight basis.

Table G-120. Inflow Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR11	2/24/2009	14:30	<0.011	52.9	74.63	79.95
PR11	7/16/2009	11:45	1.71	73.1	35.59	98.92
PR19		14:05	0.068	34.3	81.19	73.02

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR19SD		14:10	0.940	66.2	37.62	138.93
PRSD2		13:10	1.14	50.4	34.58	163.86

G.10.3 MONITORING RELATED TO ORGANOCHLORINE PESTICIDES AND PCBS IMPAIRMENTS

An OC Pesticides and PCBs TMDL has been developed for Puddingstone Reservoir. The reservoir is impaired by DDT, chlordane, and PCBs. The Regional Board, UCLA, SWAMP, and TSMP report organic data for Puddingstone from several different media. Levels of OC Pesticides and PCBs have been analyzed in the water column, lake sediment, suspended sediments, fish, porewater and suspended sediment in the porewater. The existing data for chlordane, DDT, dieldrin, and PCBs are summarized in this section. Puddingstone Reservoir is not listed for a dieldrin impairment, however dieldrin data are included for potential future needs and because nearby lakes (Echo and Peck Road Park Lakes) are impaired by this pesticide.

G.10.3.1 Water Column Data Observed in Puddingstone Reservoir

Water sampling was conducted for the UCLA study in the fall of 2008 at PR11, PR-14, and PR-15. The only analyte quantified was PCB-5 (17.95 ng/L) at PR-15. Results are shown in Table G-121.

Table G-121. Water Column Measurements at Puddingstone Reservoir in Fall 2008

	PR-11			PR-11 (field dup)			PR-14			PR-15		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	(ng/L)											
Chlordane-gamma	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
Chlordane-alpha	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
4,4'-DDE	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
4,4'-DDD	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
4,4'-DDT	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
Dieldrin	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
PCB 5	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	17.95
PCB 18	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 31	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 52	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 44	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 66	1.50	15.00	ND	1.52	15.23	3.66*	1.52	15.23	ND	1.57	15.71	ND
PCB 101	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 87	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 151	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 110	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 153	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 141	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 138	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 187	1.50	15.00	ND	1.52	15.23	5.72*	1.52	15.23	ND	1.57	15.71	ND

Contaminant	PR-11			PR-11 (field dup)			PR-14			PR-15		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)											
PCB 183	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 180	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 170	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 206	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND

*Results above detection limit but below reporting limit.

The Regional Board collected water samples from several stations on November 18, 2008 and collaborated in sampling efforts with USEPA on February 24, 2009 and July 16, 2009. On November 18, 2008 samples were collected at PR-11, PR-14, PR-15, PR-16 and PR-17. A duplicate sample was taken at Station PR-17. The collected samples were analyzed for Aroclor PCBs, PCBs, and chlorinated pesticides. The Aroclor PCBs tested for included the following congeners: 1016, 1221, 1232, 1242, 1248, 1254, and 1260. No Aroclor PCBs or chlorinated pesticides were detected at any of the sampled locations. Only one PCB congener was quantified in the water samples; PCB-201 was detected at 555.1 ng/L at PR-15. This concentration is well above the criteria for the protection of aquatic life and human health. The results of the November 18th monitoring are shown in Table G-122.

Table G-122. Water Column Measurements at Puddingstone Reservoir on November 18, 2008

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
Chlordane-alpha	ND	ND	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	ND	ND	1.0
2-4’DDD	ND	ND	ND	ND	ND	ND	1.0
2-4’DDE	ND	ND	ND	ND	ND	ND	1.0
2-4’DDT	ND	ND	ND	ND	ND	ND	1.0
4-4’DDD	ND	ND	ND	ND	ND	ND	1.0
4-4’DDE	ND	ND	ND	ND	ND	ND	1.0
4-4’DDT	ND	ND	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	ND	ND	1.0
Aroclor 1016	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1221	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1232	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1242	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1248	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1254	ND	ND	ND	ND	ND	ND	10.0

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
Aroclor 1260	ND	ND	ND	ND	ND	ND	10.0
PCB003	ND	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	ND	1.0
PCB119	ND	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
PCB149	ND	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	555.1	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	ND	1.0

A water sample was collected during field monitoring by the Regional Board on February 24, 2009 at a storm drain flowing to Puddingstone River (PR-SD). The sample was tested for PCBs only (not chlordane, DDTs, or dieldrin). No PCBs were detected in the sample. The detection limit for each PCB congener was 1 ng/L.

On July 16, 2009, water samples were collected at PR-11, PR-15, PR-16, PR-SD2. A duplicate sample was collected at PR-16. Samples were analyzed for chlorinated pesticides and PCB congeners. No analytes were detected in any of the samples (Table G-123).

Table G-123. Water Column Measurements at Puddingstone Reservoir on July 16, 2009

Contaminant (ng/L)	PR-11	PR-15	PR-16		PR-SD2	MDL
			Results	Duplicate		
Chlordane-alpha	ND	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	ND	1.0
2-4'DDD	ND	ND	ND	ND	ND	1.0
2-4'DDE	ND	ND	ND	ND	ND	1.0
2-4'DDT	ND	ND	ND	ND	ND	1.0
4-4'DDD	ND	ND	ND	ND	ND	1.0
4-4'DDE	ND	ND	ND	ND	ND	1.0
4-4'DDT	ND	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	ND	1.0
PCB003	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PR-11	PR-15	PR-16		PR-SD2	MDL
			Results	Duplicate		
PCB119	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	1.0

G.10.3.2 Porewater Data Observed in Puddingstone Reservoir

Porewater and TSS from porewater were analyzed in Puddingstone Reservoir in fall 2008 as part of the UCLA study. PR-14 and PR-15 were sampled, as shown in Table G-124 (see Stenstrom et al., 2009 for raw data). Chlordane, DDT, and dieldrin were not detected in any of the samples. PCB-31 was detected in the porewater at PR-14 and PR-15 and in the suspended sediment at PR-14, but not at reportable levels (DNQ).

Table G-124. Porewater Measurements at Puddingstone Reservoir in Fall 2008

Contaminant (ng/L)	Porewater (ng/L)			TSS in Porewater (µg/kg)		
	PR-14	PR-15	MDL	PR-14	PR-15	MDL
Chlordane	ND	ND	15	ND	ND	0.2-0.53
DDT	ND	ND	30	ND	ND	0.4-1.06
Dieldrin	ND	ND	30	ND	ND	0.4-1.06
Total PCBs	DNQ[1]	DNQ[1]	15	DNQ[1]	ND	0.2-0.53

[1] PCB-31 was detected at less than reporting level (150 ng/L for porewater and 3.01 µg/kg for TSS in porewater).

G.10.3.3 Fish Tissue Levels Observed in Puddingstone Reservoir

Concentrations of the organochlorides and PCBs in fish from Puddingstone Reservoir are shown below in Table G-125. The common carp in Puddingstone Reservoir had the highest average concentrations of Aroclor PCBs, chlordane, DDT, and dieldrin. The chlordane and DDT average concentrations for all fish species were above the FCGs. In common carp samples, the average chlordane concentration was 119.6 ppb and the average DDT level was 232.8 ppb. The average concentration in bullhead and largemouth bass for chlordane was 46.5 and 10.5 ppb, and 71.0 and 20.9 ppb for DDTs, respectively. Levels of PCBs were 60.2, 125.5, and 17.2 ppb for bullhead, common carp, and largemouth bass, respectively. Dieldrin concentrations were non-detect for bullhead and 4.6 and 1.2 ppb for common carp and largemouth bass, respectively.

Table G-125. OC Pesticides and PCBs Fish Tissue Data for Puddingstone Reservoir

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
TSMP	Aroclor PCBs	4/28/1992	Largemouth Bass	65	386	1,268.7
TSMP	Aroclor PCBs	5/6/1986	Common Carp	590	566	4474
TSMP	Aroclor PCBs	8/10/1999	Largemouth Bass	13	345	816.6
TSMP	Aroclor PCBs	6/16/1987	Common Carp	160	557	362.2
TSMP	Aroclor PCBs	6/22/1988	Brown Bullhead	66	315	538.7
TSMP	Aroclor PCBs	6/16/1987	Bullhead	ND	282	350.1
TSMP	Aroclor PCBs	6/11/1991	Largemouth Bass	54	380	1,030.4
TSMP	Aroclor PCBs	5/6/1986	Largemouth Bass	ND	302	509.7
TSMP	Chlordane	4/28/1992	Largemouth Bass	31.7	386	1,268.7
TSMP	Chlordane	5/6/1986	Common Carp	460	566	4474
TSMP	Chlordane	8/10/1999	Largemouth Bass	2.8	345	816.6
TSMP	Chlordane	6/16/1987	Common Carp	193.5	557	362.2
TSMP	Chlordane	6/22/1988	Brown Bullhead	48.5	315	538.7
TSMP	Chlordane	6/16/1987	Bullhead	44.4	282	350.1
TSMP	Chlordane	6/11/1991	Largemouth Bass	16.1	380	1,030.4
TSMP	Chlordane	5/6/1986	Largemouth Bass	10.4	302	509.7

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
SWAMP	Chlordane	Summer 2007	Largemouth Bass	9.29	366	NA
SWAMP	Chlordane	Summer 2007	Largemouth Bass	4.97	365	NA
SWAMP	Chlordane	9/22/2004	Largemouth Bass	12.43	397.6	799.6
SWAMP	Chlordane	9/22/2004	Largemouth Bass	5.95	343	563.1
SWAMP	Chlordane	9/22/2004	Largemouth Bass	13.55	456.6	1,464.6
SWAMP	Chlordane	9/22/2004	Largemouth Bass	7.31	342.6	581.6
SWAMP	Chlordane	9/22/2004	Common Carp	1.17	420.7	1,203.9
SWAMP	Chlordane	9/22/2004	Common Carp	27.25	632.7	3,795
SWAMP	Chlordane	9/22/2004	Common Carp	19.98	593.7	2,631
SWAMP	Chlordane	9/22/2004	Common Carp	15.60	669	4,354.7
TSMP	DDT	4/28/1992	Largemouth Bass	36	386	1,268.7
TSMP	DDT	5/6/1986	Common Carp	880	566	4474
TSMP	DDT	8/10/1999	Largemouth Bass	10.7	345	816.6
TSMP	DDT	6/16/1987	Common Carp	358	557	362.2
TSMP	DDT	6/22/1988	Brown Bullhead	72	315	538.7
TSMP	DDT	6/16/1987	Bullhead	70	282	350.1
TSMP	DDT	6/11/1991	Largemouth Bass	25	380	1,030.4
TSMP	DDT	5/6/1986	Largemouth Bass	16	302	509.7
SWAMP	DDT	Summer 2007	Largemouth Bass	10.8	365	NA
SWAMP	DDT	Summer 2007	Largemouth Bass	30.77	366	NA
SWAMP	DDT	9/22/2004	Largemouth Bass	33.72	397.6	799.6
SWAMP	DDT	9/22/2004	Largemouth Bass	15.561	343	563.1
SWAMP	DDT	9/22/2004	Largemouth Bass	35.34	456.6	1,464.6
SWAMP	DDT	9/22/2004	Largemouth Bass	19.42	342.6	581.6
SWAMP	DDT	9/22/2004	Common Carp	2.51	420.7	1,203.9
SWAMP	DDT	9/22/2004	Common Carp	69.357	632.7	3,795
SWAMP	DDT	9/22/2004	Common Carp	47.66	593.7	2,631
SWAMP	DDT	9/22/2004	Common Carp	39.082	669	4,354.7
TSMP	Dieldrin	4/28/1992	Largemouth Bass	ND	386	1,268.7
TSMP	Dieldrin	5/6/1986	Common Carp	12	566	4474
TSMP	Dieldrin	8/10/1999	Largemouth Bass	ND	345	816.6
TSMP	Dieldrin	6/16/1987	Common Carp	5	557	362.2
TSMP	Dieldrin	6/22/1988	Brown Bullhead	ND	315	538.7
TSMP	Dieldrin	6/16/1987	Bullhead	ND	282	350.1
TSMP	Dieldrin	6/11/1991	Largemouth Bass	ND	380	1,030.4
TSMP	Dieldrin	5/6/1986	Largemouth Bass	ND	302	509.7
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.73	397.6	799.6
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	0.858	343	563.1

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.58	456.6	1,464.6
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.16	342.6	581.6
SWAMP	Dieldrin	9/22/2004	Common Carp	0.704	420.7	1,203.9
SWAMP	Dieldrin	9/22/2004	Common Carp	4.34	632.7	3,795
SWAMP	Dieldrin	9/22/2004	Common Carp	3.35	593.7	2,631
SWAMP	Dieldrin	9/22/2004	Common Carp	2.48	669	4,354.7
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.68	366	NA
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	ND	365	NA
SWAMP	PCB	9/22/2004	Largemouth Bass	29.108	397.6	799.6
SWAMP	PCB	9/22/2004	Largemouth Bass	16.024	343	563.1
SWAMP	PCB	9/22/2004	Largemouth Bass	35.87	456.6	1,464.6
SWAMP	PCB	9/22/2004	Largemouth Bass	17.85	342.6	581.6
SWAMP	PCB	9/22/2004	Common Carp	6.461	420.7	1203.9
SWAMP	PCB	9/22/2004	Common Carp	49.304	632.7	3,795
SWAMP	PCB	9/22/2004	Common Carp	36.799	593.7	2,631
SWAMP	PCB	9/22/2004	Common Carp	28.314	669	4,354.7
SWAMP	PCB	Summer 2007	Largemouth Bass	19	366	NA
SWAMP	PCB	Summer 2007	Largemouth Bass	8	365	NA
Regional Board	Total Detectable DDTs	11/3/2006	Bass	25.6	NA	NA
Regional Board	Total Detectable DDTs	11/3/2006	Bass	10.1	NA	NA
Regional Board	Total Detectable PCBs	11/3/2006	Bass	ND	NA	NA
Regional Board	Total Detectable PCBs	11/3/2006	Bass	3.3	NA	NA
Regional Board	Chlordane	11/3/2006	Bass	1.1	NA	NA
Regional Board	Chlordane	11/3/2006	Bass	ND	NA	NA
Regional Board	Dieldrin	11/3/2006	Bass	ND	NA	NA
Regional Board	Dieldrin	11/3/2006	Bass	ND	NA	NA

ND = non-detect

G.10.3.4 Sediment Data Observed in Puddingstone Reservoir

Sediment samples from Puddingstone Reservoir were collected in the fall of 2008 by UCLA at PR-14 and PR-15. A field duplicate was collected at PR-14 and laboratory duplicates were performed for each sample. Chlordane-gamma was detected at unreportable levels at PR-14 (laboratory duplicate of field duplicate). DDT and dieldrin were not detected in any sample. Four PCB congeners were detected at PR-14 (laboratory duplicate of field duplicate): PCB-5, PCB-31, PCB-66, and PCB-138. The concentration of PCB-5 was 6.78 µg/kg dry weight, and the concentration of PCB-31 was 12.67 µg/kg dry weight, these were the only reportable PCB congeners. PCBs were not detected at PR-15. The results and detection and reporting limits for each contaminant are shown in Table G-126.

Table G-126. OC Pesticides and PCBs Measurements in Sediment at Puddingstone Reservoir in Fall 2008

	PR-14			PR-14 (lab dup)			PR-14 (field dup)			PR-14 (lab dup of field dup)			PR-15			PR-15 (lab dup)		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	µg/kg dry suspended solids																	
Chlordane-gamma	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	0.14*	1.58	15.83	ND	1.58	15.84	ND
Chlordane-alpha	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
4,4'-DDE	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
4,4'-DDD	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
4,4'-DDT	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
Dieldrin	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
PCB 5	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	6.78	1.58	15.83	ND	1.58	15.84	ND
PCB 18	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 31	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	12.67	1.58	15.83	ND	1.58	15.84	ND
PCB 52	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 44	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 66	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	1.03*	1.58	15.83	ND	1.58	15.84	ND
PCB 101	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 87	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 151	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 110	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 153	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 141	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 138	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	0.97*	1.58	15.83	ND	1.58	15.84	ND
PCB 187	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 183	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 180	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 170	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 206	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND

*Results above detection limit, but below reporting limit.

Sediment samples were collected by the Regional Board and USEPA on July 16, 2009 at PR-11, PR-15, PR-16, PR-19, PR-19SD, and PR-SD2. Chlordane was quantified at all stations, between 1.1 and 6.5 µg/kg dry weight. The chlordane levels at PR-11 and PR-15 were above the TEC CBSQG (3.24 µg/kg dry weight). DDT was only detected at PR-19; however, DDE was detected at almost all stations. PCBs were also detected at all locations except PR-19. Table G-127 shows the results of the July 16, 2009 monitoring.

Table G-127. OC Pesticides and PCBs Measurements in Sediment at Puddingstone Reservoir on July 16, 2009

Contaminant (µg/kg dry weight)	PR-11	PR-15	PR-16	PR-19	PR-19SD	PR-SD2	MDL
Chlordane	6.5	4.1	2.4	1.1	2.6	2.2	1
DDE	5.2	18.6	11.8	6.1	8.5	ND	1
DDT	ND	ND	ND	1.7	ND	ND	1
Dieldrin	ND	ND	ND	ND	ND	ND	1
PCB099	1	1.3	1.6	ND	ND	ND	1
PCB101	1.45	1.45	1.8	ND	ND	ND	1
PCB110	1.4	1.2	1.3	ND	ND	1	1
PCB118	ND	1.2	ND	ND	ND	ND	1
PCB119	ND	ND	ND	ND	193.7	ND	1
PCB138	ND	1.8	ND	ND	1	ND	1
PCB153	1.4	1.8	ND	ND	ND	ND	1
PCB174	ND	ND	1.1	ND	ND	ND	1
PCB180	2.1	1.7	1.5	ND	ND	ND	1
Total PCBs	7.4	10.5	7.3	ND	194.7	1.0	1

G.10.3.5 Suspended Sediment Data Observed in Puddingstone Reservoir

Samples of suspended solids were collected at PR-11, PR-14, and PR-15 in the fall of 2008 by UCLA. Chlordane-gamma was detected at unreportable levels at PR-11. In each sample except one of the PR-11 duplicates, PCBs were detected below reporting limits. The individual PCBs detected at each station and other results of the fall 2008 TSS analysis are shown in Table G-128.

Table G-128. OC Pesticides and PCBs Measurements in Suspended Sediment at Puddingstone Reservoir in Fall 2008

Contaminant	PR-11			PR-11 (dup 1)			PR-11 (dup 2)			PR-14			PR-15		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry suspended solids														
Chlordane-gamma	29.07	290.70	ND	2.38	23.79	10.30*	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
Chlordane-alpha	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
4,4'-DDE	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
4,4'-DDD	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND

Contaminant	PR-11			PR-11 (dup 1)			PR-11 (dup 2)			PR-14			PR-15		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry suspended solids														
4,4'-DDT	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
Dieldrin	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
PCB 5	29.07	290.70	132.56*	2.38	23.79	7.77*	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 18	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 31	29.07	290.70	ND	2.38	23.79	14.75*	2.11	21.14	ND	22.11	221.13	61.18*	36.23	362.32	ND
PCB 52	29.07	290.70	256.22*	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 44	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 66	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	62.36*
PCB 101	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 87	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 151	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 110	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 153	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 141	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 138	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	98.23*
PCB 187	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 183	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 180	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 170	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 206	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND

*Results are above detection levels but below reporting levels.

In the fall of 2008, a TSS sample was collected at PR-11 during a wet weather event (Table G-129). A composite sample from the event did not detect any of the pollutants. A grab sample from PR-11 was collected 90 minutes into the wet weather event also had no detectable results. Water column samples were also collected during this event (a time series composite and a single time point sample), but not analyzed.

Table G-129. Wet Weather OC Pesticides and PCBs Measurements in Suspended Sediment at Puddingstone Reservoir in Fall 2008

Contaminant (µg/kg dry suspended solids)	PR-11 Storm Composite	Composite MDL	PR-11 Storm @ 1.5 hours	Grab Sample MDL
Chlordane	ND	1.57	ND	2.70
DDT	ND	3.14	ND	5.39
Dieldrin	ND	3.14	ND	5.39
Total PCBs	ND	1.57	ND	2.70

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G.11 Monitoring Data for Santa Fe Dam Park Lake

Monitoring data relevant to the impairments of Santa Fe Dam Park Lake are available from 1992, 1993, 2009, and 2010. Figure G-48 shows the historical and recent monitoring locations for these lakes.

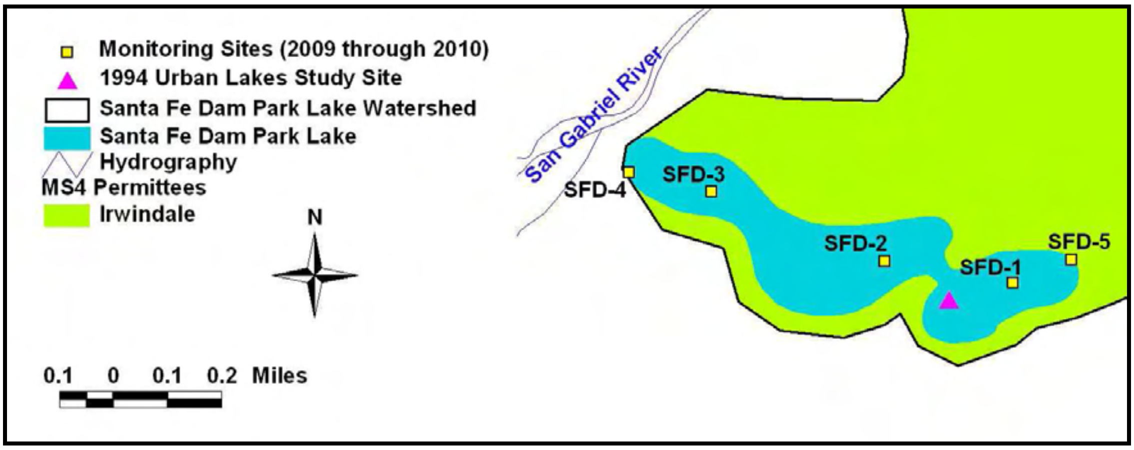

Figure G-48. Santa Fe Dam Park Lake Monitoring Sites

G.11.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

In 1992 and 1993, Santa Fe Dam Park Lake was monitored for water quality as part of the Urban Lakes Study (Table G-130). The station was located in the southeast end of the lake near the spillway (pink triangle, Figure G-48) (UC Riverside, 1994). TKN ranged from 0.3 mg/L to 1.1 mg/L. Ammonium generally ranged from 0.1 mg/L to 0.2 mg/L with 21 measurements less than the reporting limit and one measurement of 0.4 mg/L collected at a depth of 2 meters. The upper range of these concentrations are above the chronic target, but below the acute target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All 37 samples of nitrite were less than the reporting limit, and the majority of nitrate samples (32) were less than the reporting limit; the maximum observed nitrate concentration was 0.2 mg/L. All orthophosphate and total phosphorus concentrations were less than the reporting limit except one total phosphorus concentration which measured 0.1 mg/L. pH ranged from 8.0 to 9.6, and TOC ranged from 2.3 mg/L to 3.4 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 1 µg/L to 29 µg/L with an average of 13 µg/L; the raw data were not available.

Table G-130. Santa Fe Dam Park Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
8/10/1992	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	9.1	3.3	256
	2	0.9	0.4	<0.01	<0.01	<0.01	<0.01	9.2	3.3	279
	3.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3	296
8/10/1992	0	0.9	0.2	<0.01	<0.01	<0.01	<0.01	8.9	2.7	274
	2.5	0.9	0.1	<0.01	0.1	<0.01	<0.01	8.7	2.9	346
8/10/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3.3	268
	2.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3.3	309
9/10/1992	0	1.1	0.1	<0.01	<0.01	<0.01	<0.01	8.9	3.1	284
	2	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3	287
	3.5	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	2.9	281
10/13/1992	0	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.7	2.6	286
	2	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.7	2.7	316
	3.5	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.6	2.8	301
11/3/1992	0	0.7	0.1	<0.01	0.2	<0.01	<0.01	8.7	3.1	231
	1.5	0.7	0.1	<0.01	0.1	<0.01	<0.01	8.8	3.1	252
	2.5	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8.7	3.2	282
12/10/1992	0	0.6	<0.01	<0.01	0.1	<0.01	<0.01	8.7	2.6	286
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.8	284
	3.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.8	327
1/14/1993	0	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	2.7	181
	2	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.8	183
	3.5	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.8	189
2/3/1993	0	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8	2.4	221
	2	0.6	0.2	<0.01	<0.01	<0.01	<0.01	8.2	2.6	251
	3	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8.2	2.3	229
3/9/1993	0	0.6	0.1	<0.01	<0.01	<0.01	<0.01	8.1	2.7	212
	2	0.7	0.1	<0.01	<0.01	<0.01	<0.01	8.3	2.5	201
	3.5	0.7	0.1	<0.01	<0.01	<0.01	<0.01	8.3	2.6	223
4/14/1993	0	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	3.4	247
	1.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	2.5	235
	2.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	2.7	256

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
5/25/1993	0	0.4	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3	257
	1.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3.2	248
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3.3	246
6/21/1993	0	0.3	<0.01	<0.01	<0.01	<0.01	0.1	9.5	2.7	249
	1.5	0.3	<0.01	<0.01	<0.01	<0.01	<0.01	9.6	2.9	252
	2.5	0.3	<0.01	<0.01	<0.01	<0.01	<0.01	9.6	3.1	242

The 1996 Water Quality Assessment Report states that pH was partially supporting the aquatic life use and not supporting the contact recreation and secondary drinking water uses. Ninety-five measurements of pH were taken, ranging from 7.5 to 9.6. The associated database did not contain the raw data for these samples.

On March 3 and August 3, 2009, USEPA and the Regional Board sampled water quality in the Santa Fe Dam Park Lake (Table G-131). The field notes indicate that water is pumped from an underground well to fill the lake every night. The well water enters the lake via a rock stream about 50 ft from SFD-4. Potable water is also input at SFD-3 from the Valley County Water District. During the swimming season, lake water is treated with chlorine several days a week. The chlorine is mixed with lake water in a pump house. Three samples were collected in the lake during both sampling events. During the winter sampling, two shoreline samples were collected on the western and eastern ends of the lakes. In the summer, the well water was sampled. Overall, both nitrogen and phosphorus levels were very low. Chlorophyll *a* concentrations in the lake did not exceed 20.5 μg/L; a shoreline sample had a chlorophyll *a* concentration of 25.8 μg/L. In August, chlorophyll *a* was below the detection level. The field notes report that the lake was very green in August and the Secchi depth readings were shallow, indicative of algal production. A less common chlorophyll structure, e.g. Chlorophyll *b*, could be present in the lake. The average depths at SFD-1 and SFD-2 were 2.93 and 3.02 meters, respectively. The depth at SFD-3 averaged 2.5 meters.

Table G-131. 2009 In-lake and Shoreline Water Column Measurements for Santa Fe Dam Park Lake

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll *a* (μg/L)	Secchi Depth (m)
3/3/2009	**In-lake Samples**									
	SFD-1	10:00	1.1	<0.03	0.04	0.1	<0.0075	0.025	20.5	0.61
	SFD-2	10:40	1.1	0.05	0.04	0.08	<0.0075	0.021	14.4	0.84
	SFD-3	11:10	0.84	0.03	0.04	0.06	<0.0075	0.03	16.7	0.84
	Shoreline Samples									
	SFD-4	12:40	0.98	0.03	0.03	0.04	<0.0075	0.028	14.0	NA
	SFD-4 (duplicate)	13:00	1.1	<0.03	0.03	0.04	<0.0075	0.028	11.6	NA
	SFD-5	13:30	0.98	0.03	0.03	0.14	<0.0075	0.036	25.8	NA

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll *a* (µg/L)	Secchi Depth (m)
8/3/2009	**In-lake Samples**									
	SFD-1	10:45	0.58	<0.03	0.04	<0.01	<0.0075	0.027	<1	0.61
	SFD-2	9:20	0.47	<0.03	<0.01	<0.01	<0.0075	0.036	<1	0.56
	SFD-3	8:40	<0.46	<0.03	<0.01	<0.01	<0.0075	0.050	<1	0.46
	Well sample									
	Well SFD-1	13:30	<0.456	<0.03	<0.01	2.985	0.016	NS	NS	NA

Supplemental water quality samples were collected from Santa Fe Dam Park Lake. Table G-132 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data measured in the lake. Temperature and pH measurements reported in the field notes are also shown in this table. The additional chloride added to the lake in the summer is apparent in the higher concentrations measured during the August sampling event. The average chloride concentration in the lake during August was 35.4 mg/L and 28.1 mg/L in March. Temperature, alkalinity, total hardness, total dissolved solids, and TSS also increased during the summer. The pH range remained similar for both sampling periods. The TOC was slightly lower in the summer, between 3.5 and 3.7 mg/L; the winter range of TOC was 4.0 to 5.2 mg/L.

Table G-132. 2009 Supplemental Water Quality Monitoring for Santa Fe Dam Park Lake

Date	Location	Time	Chloride (mg/L)	Temper-ature (^{o}C)	pH	Total Alka-linity (mg/L)	Total Hardness as $CaCO_3$ (mg/L)	TDS (mg/L)	TSS (mg/L)	TOC (mg/L)
3/3/2009	**In-lake Samples**									
	SFD-1	10:00	28.18	15.0	8.6	118	107.5	284	6.9	4.0
	SFD-2	10:40	27.76	15.0	8.7	118	104.9	314	5.2	4.6
	SFD-3	11:10	28.30	15.0	8.7	114	103.0	314	6.7	5.0
	Shoreline Samples									
	SFD-4	12:40	27.88	16.0	8.7	114	103.1	290	8.5	5.2
	SFD-4 (duplicate)	13:00	27.88	16.0	8.7	112	102.0	292	8.2	4.5
	SFD-5	13:30	27.79	16.0	8.7	116	101.8	286	10.5	4.7
8/3/2009	**In-lake Samples**									
	SFD-1	10:45	35.63	28.5	8.8	126	131.3	286	9.5	3.5
	SFD-2	9:20	35.23	27.5	8.7	124	133.1	306	9.6	3.7
	SFD-3	8:40	35.23	27.2	8.7	122	131.3	316	14.8	3.5

On May 4, 2009, Clean Lakes Inc. was contracted by the Los Angeles County Department of Parks and Recreation to conduct baseline water quality monitoring (Table G-133) of Santa Fe Dam Park Lake to determine if aquatic weed or algal growth controls were needed. Three locations were sampled in the lake at a depth of approximately 1 ft below the water surface. The location numbering and locations correspond to SFD-1, SFD-2, and SFD-3 monitored by the Regional Board and USEPA.

Table G-133. In-lake Water Column Measurements for Santa Fe Dam Park Lake (5/4/09)

Date	Location	Time	NH_3-N (mg/L)	NO_{2+3}-N (mg/L)	Total P (mg/L)	COD (mg/L)	TSS (mg/L)	Secchi Depth (m)
5/4/2009	SFD-1	9:55	0.24	0.29	<0.01	14	<5	1.7
	SFD-2	10:13	0.47	0.21	<0.01	16	<5	1.4
	SFD-3	10:24	0.17	0.18	<0.01	17	<5	1.5

Four types of alkalinity were also monitored at these locations (Table G-134). Total alkalinity at each station was approximately 140 mg/L in the bicarbonate form.

Table G-134. Alkalinity Measurements for Santa Fe Dam Park Lake (5/4/09)

Date	Location	Time	Total Alkalinity (mg/L)	Carbonate Alkalinity (mg/L)	Bicarbonate Alkalinity (mg/L)	Hydroxide Alkalinity (mg/L)
5/4/2009	SFD-1	9:55	142	< 1	142	< 1
	SFD-2	10:13	142	< 1	142	< 1
	SFD-3	10:24	140	< 1	140	< 1

Clean Lakes, Inc. conducted depth profiles at each location in Santa Fe Dam Park Lake on May 4, 2009 (Table G-135). The pH ranged from 7.39 to 7.96 at all locations and depths. DO ranged from 5.54 mg/L to 8.27 mg/L at all stations and depths with the exception of the bottom reading at station SFD-1 where the DO was 3.72 mg/L. Depth measurements were between 3.18 and 3.25 meters, and Secchi depth readings were between 1.35 and 1.65 meters.

Table G-135. Profile Data Collected in Santa Fe Dam Park Lake (5/4/09)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
SFD-1	9:50	0.27	22.44	7.96	8.27	1.65	3.20
		0.67	22.29	7.86	8.17		
		1.35	22.01	7.78	8.03		
		2.01	21.91	7.73	7.73		
		2.66	21.25	7.64	7.18		
		3.29	21.18	7.58	5.54		
		3.48	21.21	7.55	3.72		
SFD-2	9:30	0.30	22.27	7.67	8.19	1.35	3.18

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
		0.69	22.16	7.64	8.04		
		1.31	21.97	7.6	7.94		
		1.97	24.54	7.59	7.87		
		2.66	21.43	7.52	7.29		
		3.31	21.16	7.43	6.21		
		3.44	24.15	7.39	5.82		
SFD-3	10:20	0.31	21.86	7.59	7.97	1.52	3.25
		0.67	21.81	7.56	7.73		
		1.34	21.45	7.53	7.62		
		2.02	21.3	7.52	7.47		
		2.67	21.18	7.51	7.56		
		3.30	21.17	7.51	7.44		
		3.55	21.17	7.49	7.35		

Profile data for these three sites was also collected by the Regional Board on August 3, 2009 and is listed in Table G-136. The profile data for SFD-1 is shown in Figure G-49. The temperature at this site ranged from 26.3 to 28.5 °C and the pH ranged from 7.45 to 8.75. The DO was greatest at one meter of depth, it ranged from 1.75 to 12.24 mg/L. Morning and afternoon data were collected from SFD-2 and SFD-3, shown in Figure G-50 and Figure G-51. At both sites, the afternoon temperature and DO were slightly higher, especially at the surface. At both stations, below two meters of depth there was no temperature difference between morning and afternoon. Field data were also collected for the groundwater source during this event. After purging the line for approximately 10 minutes, the pH was 7.59 and the temperature was 18.4 ºC.

Table G-136. Profile Data Collected in Santa Fe Dam Park Lake (8/3/09)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
SFD-1	10:45	0.10	28.50	8.75	10.74	0.61	2.95
		0.50	28.36	8.73	11.83		
		1.00	28.03	8.73	12.24		
		1.50	27.67	8.55	9.75		
		2.00	27.23	8.39	8.20		
		2.50	26.72	7.83	3.79		
		3.00	26.33	7.45	1.75		
SFD-2	9:30	0.07	27.52	8.59	8.58	0.56	2.90
		0.49	27.53	8.74	9.94		
		0.99	27.50	8.74	10.06		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
		1.50	27.46	8.71	10.05		
		1.99	27.44	8.68	9.80		
		2.50	27.33	8.28	6.24		
		2.99	26.89	7.98	4.56		
		2.98	26.91	7.94	3.96		
SFD-2	16:30	0.09	29.09	8.96	11.83	0.56	2.90
		0.50	29.12	8.85	11.87		
		1.01	29.01	8.86	11.84		
		1.50	28.43	8.78	11.12		
		1.99	27.39	8.50	8.46		
		2.49	27.13	8.09	5.11		
		3.01	26.85	7.93	3.13		
SFD-3	8:45	0.48	27.17	8.55	9.59	0.46	2.36
		0.98	27.18	8.65	10.01		
		2.00	27.15	8.64	9.98		
		0.10	27.20	8.73	10.21		
		1.50	27.18	8.64	10.09		
SFD-3	16:00	0.09	28.91	9.02	12.00	0.46	2.36
		0.09	28.90	8.99	12.00		
		0.49	28.82	9.00	12.11		
		1.00	28.50	9.01	12.19		
		1.51	27.85	8.95	12.34		
		2.01	27.28	8.74	10.89		

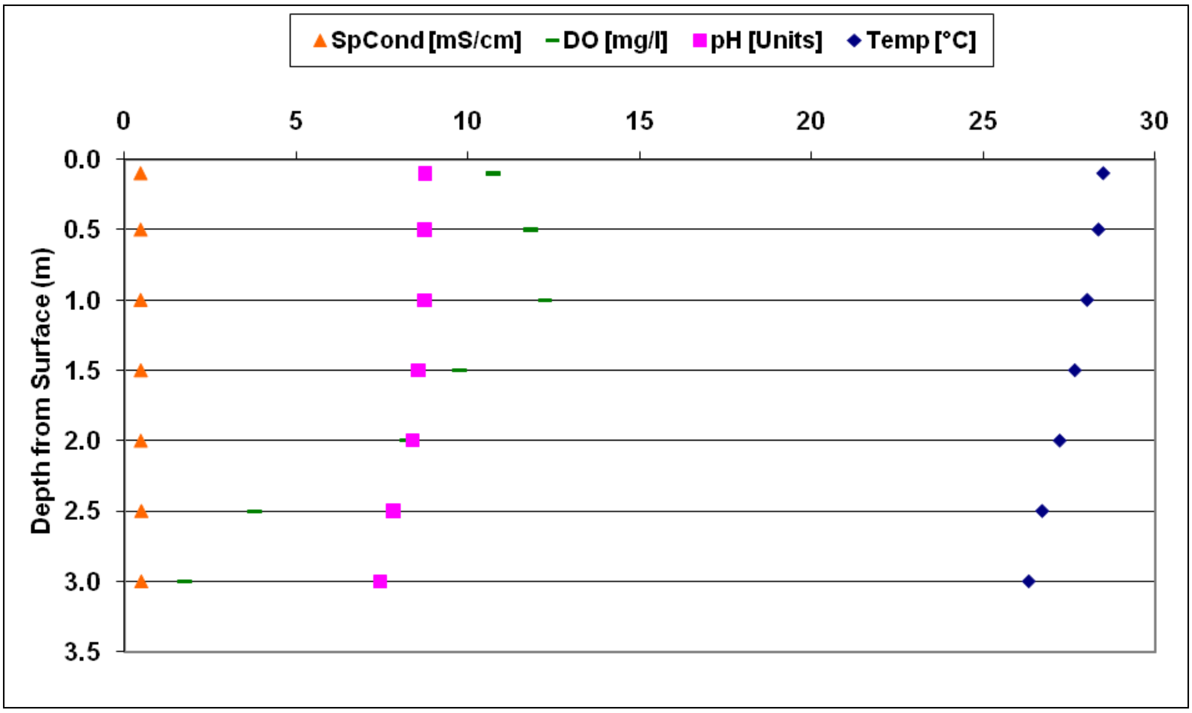

Figure G-49. Profile Data Collected at SFD-1 on August 3, 2009

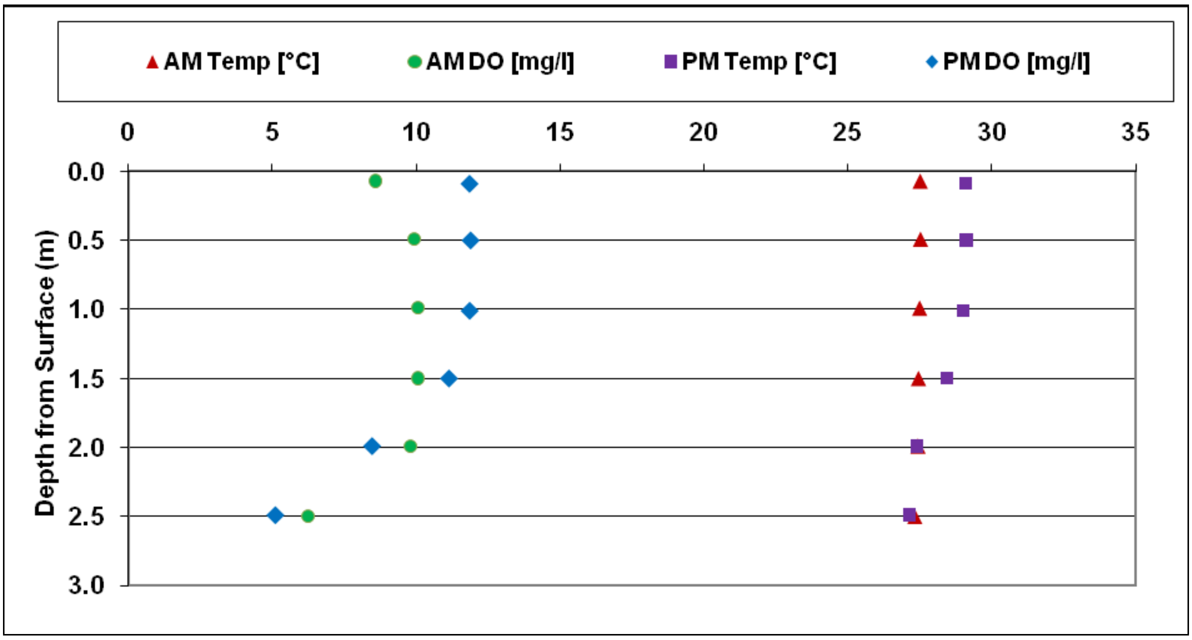

Figure G-50. Profile Data Collected at SFD-2 on August 3, 2009

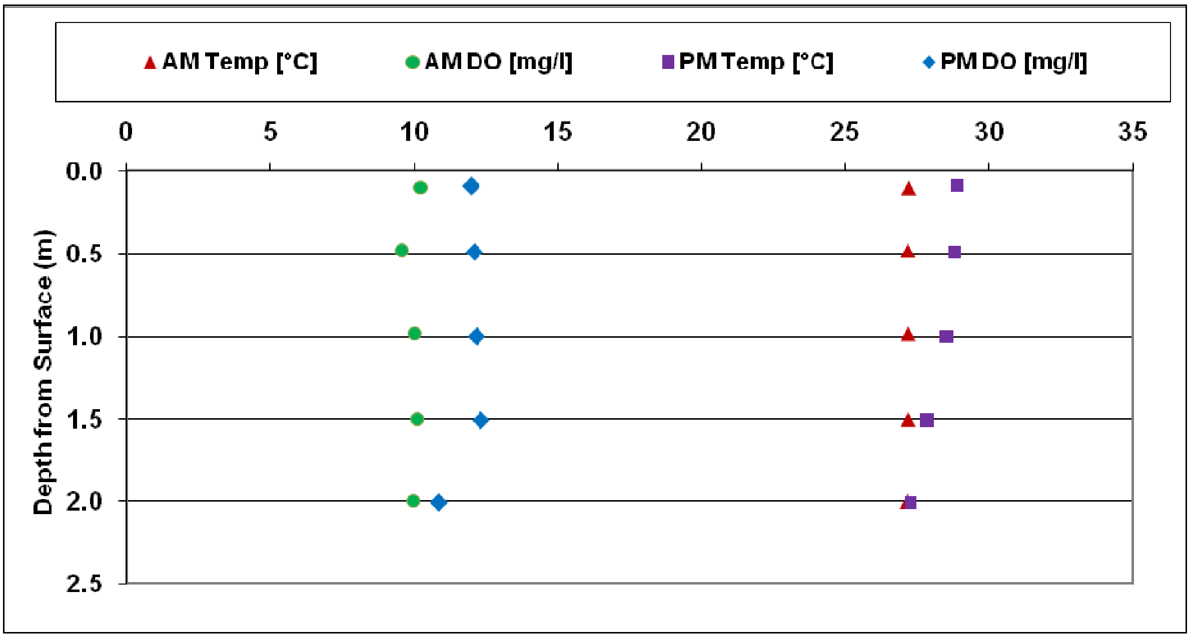

Figure G-51. Profile Data Collected at SFD-3 on August 3, 2009

The DO differences from morning and afternoon can further be analyzed by the DO saturation levels. The saturation at SFD-2 was highest in the afternoon, reaching a maximum of 157 percent at the surface. In the morning, the surface DO was at 110 percent and 129 percent between 0.5 and 2 meters of depth. The DO below 2 meters of depth was between 80 and 40 percent in the morning and afternoon. The DO saturation profile for SFD-2 is shown in Figure G-52.

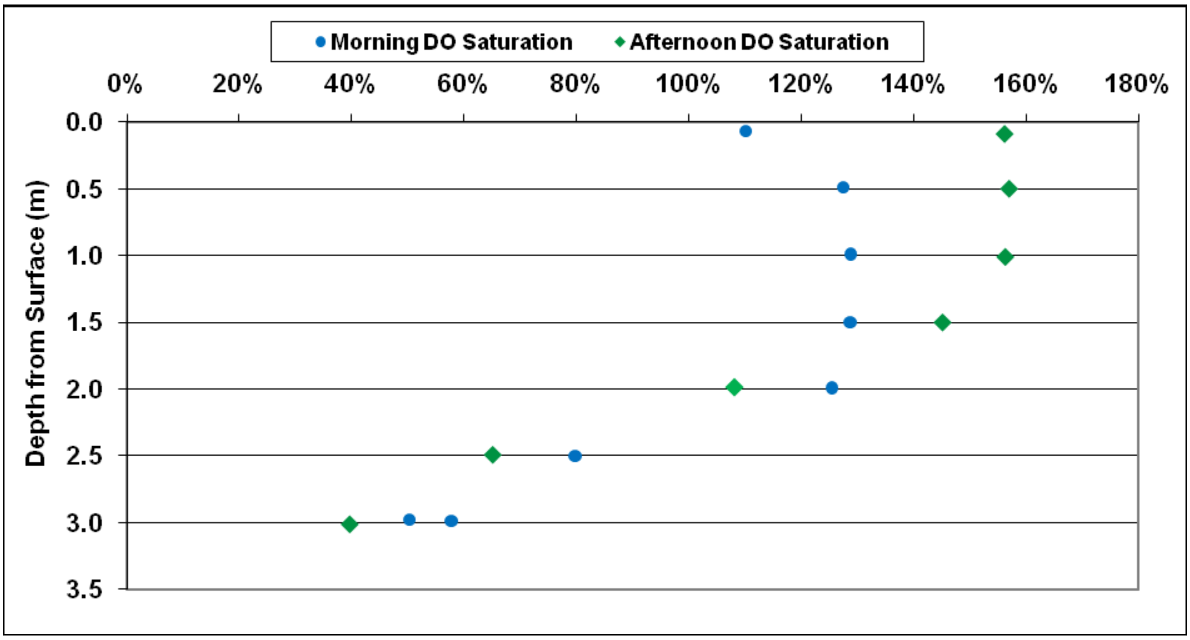

Figure G-52. DO Saturation from Profile Data Collected at SFD-2 on August 3, 2009

The DO saturation at SFD-3 had spatial and temporal patterns similar to those at SFD-2, shown in Figure G-53. The maximum DO was in the afternoon near the surface, 159 percent. The DO saturation in the morning had very little variation with depth and ranged from 122-130 percent. The DO saturation in the afternoon was around 159 percent between 1.5 meters of depth and the surface. The lowest saturation percent in the afternoon was 139, measured at 2.0 meters of depth.

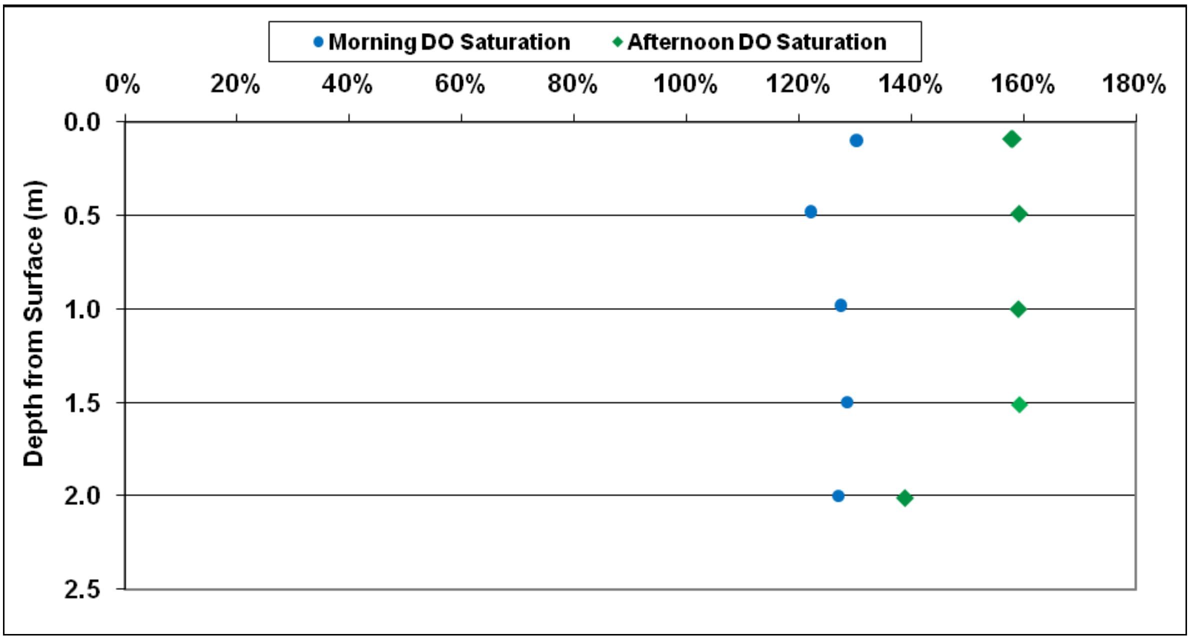

Figure G-53. DO Saturation from Profile Data Collected at SFD-3 on August 3, 2009

During the December 2009 monitoring event, profile measurements were collected at two in-lake stations. Additional single measurements were collected at one shoreline site. Table G-137 summarizes the field data collected during this event.

Table G-137. Field Data Collected At Santa Fe Dam Park Lake on December 14, 2009

Station	Depth (m)	pH	ORP (mV)	Temp (C)	DO (mg/L)	Cond (mS/cm2)
SFD-1	.5	8.63	105.6	12.75	10.2	0.537
	1	8.70	106.0	12.72	9.8	0.538
	1.5	8.73	105.4	12.65	9.2	0.541
	2	8.73	107.8	12.54	8.15	0.544
	2.5	8.63	108.3	12.41	6.1	0.550
SFD-3	.5	8.82	82.4	12.24	9.66	0.542
	1	8.89	85.0	12.22	9.55	0.542
	1.5	8.90	87.0	12.08	9.17	0.543
	2	8.90	87.4	12.06	8.98	0.543
	2.5	8.88	88.9	11.99	8.45	0.544
	2.8	8.87	88.9	12.00	8.07	0.546

Station	Depth (m)	pH	ORP (mV)	Temp (C)	DO (mg/L)	Cond (mS/cm2)
SFD-4	Surface	8.08	97.0	13.24	10.8	0.542
SFD-5	Surface	8.65	86.7	14.15	11.0	0.540

USEPA sampled Santa Fe Dam Park Lake again on August 12, 2010 (Table G-138). Secchi depth ranged from 0.61 m to 0.762 m. In-lake samples of TKN ranged from less than the detection limit of 0.47 mg-N/L to 0.594 mg-N/L. Ammonia samples at SFD-1 and SFD-3 were less than the detection limit of 0.03 mg-N/L, and nitrite samples were both detected at 0.035 mg-N/L. Nitrate concentrations were less than the detection limit (0.01 mg-N/L) at SFD-3 and 0.097 mg-N/L at SFD-1. Orthophosphate measurements at both sites were less than the detection limit of 0.0075 mg-P/L; total phosphorus concentrations ranged from 0.023 mg-P/L to 0.129 mg-P/L. Chlorophyll *a* concentrations ranged from 18.4 µg/L to 22.7 µg/L. Ammonia and nitrite concentrations in the groundwater were similar to those in the lake. TKN and nitrate in the groundwater sample were 1.11 mg-N/L and 1.62 mg-N/L, respectively. Orthophosphate concentration of the groundwater was 0.036 mg-P/L; total phosphorus was less than the detection limit of 0.0165 mg-P/L. Chlorophyll *a* concentration was less than the detection limit of 1.2 µg/L.

Table G-138. 2010 In-lake Water Column Measurements for Santa Fe Dam Park Lake

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll *a* (µg/L)	Secchi Depth (m)
8/12/2010	SFD-1	11:00	0.594	<0.03	0.035	0.097	<0.0075	0.129	22.7	0.762
8/12/2010	SFD-3	11:40	<0.47	<0.03	0.035	<0.01	<0.0075	0.0228	18.4	0.61
8/12/2010	SFD Well	12:40	1.11	<0.03	0.036	1.62	0.036	<0.0165	<1.2	NA

Supplemental water quality data for the August 2010 sampling event are shown in Table G-139.

Table G-139. 2010 Supplemental Water Quality Measurements for Santa Fe Dam Park Lake

Date	Location	Time	Chloride (mg/L)	Temper-ature (°C)	pH	Total Alka-linity (mg/L)	Total Hardness as $CaCO_3$ (mg/L)	TDS (mg/L)	TSS (mg/L)	TOC (mg/L)
8/12/2010	SFD-1	11:00	35.1	25.85	8.72	156	92.5	260	8.50	4.32
8/12/2010	SFD-3	11:40	36.4	25.93	8.73	150	92.9	222	10.8	4.11
8/12/2010	SFD Well	12:40	19.7	18.71	7.81	162	NA	228	<0.5	<2.0

During the August 2010 monitoring event, 24-hr temperature/pH/DO/conductivity probes were deployed at SFD-1 and SFD-3 (Figure G-54 and Figure G-55, respectively). The diurnal sampler placed at SFD-1 measured pH values ranging from 8.75 to 8.97 and DO concentrations ranging from 8.3 mg/L to 9.9 mg/L. At SFD-3, diurnal measurements of pH ranged from 8.82 to 8.97, and DO concentrations ranged from 8.9 mg/L to 11.3 mg/L.

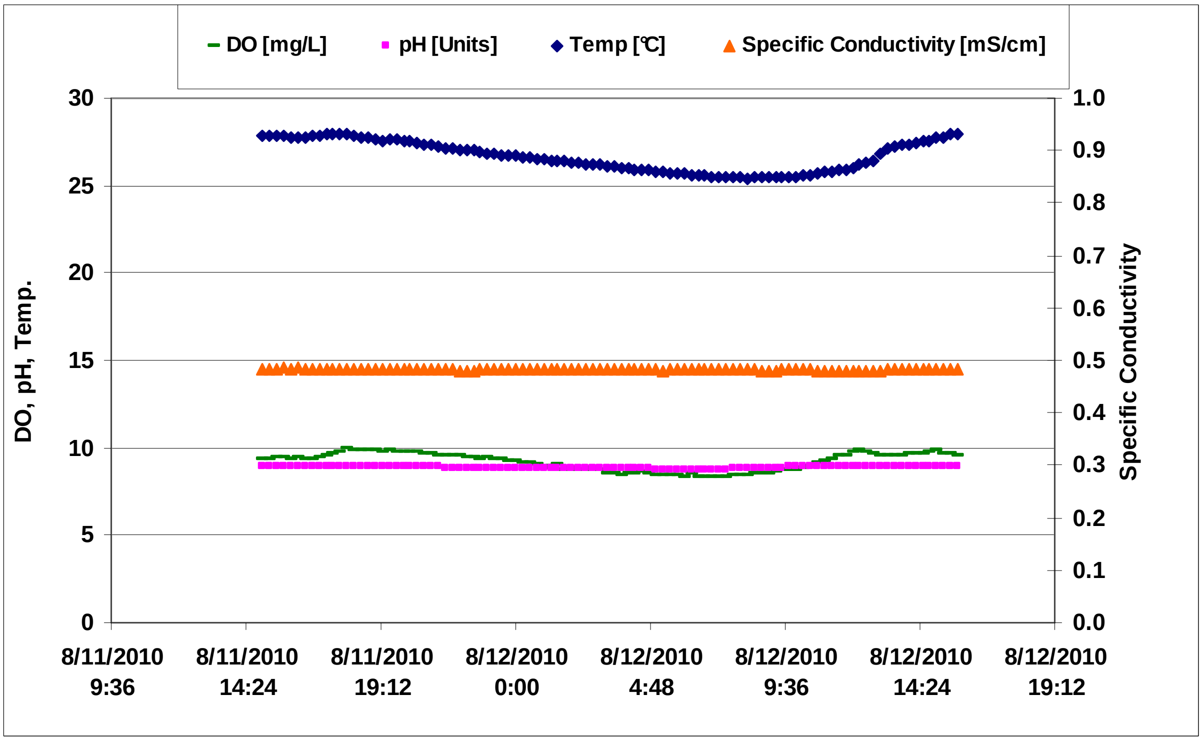

Figure G-54. Profile Data Collected at SFD-1 on August 12, 2010

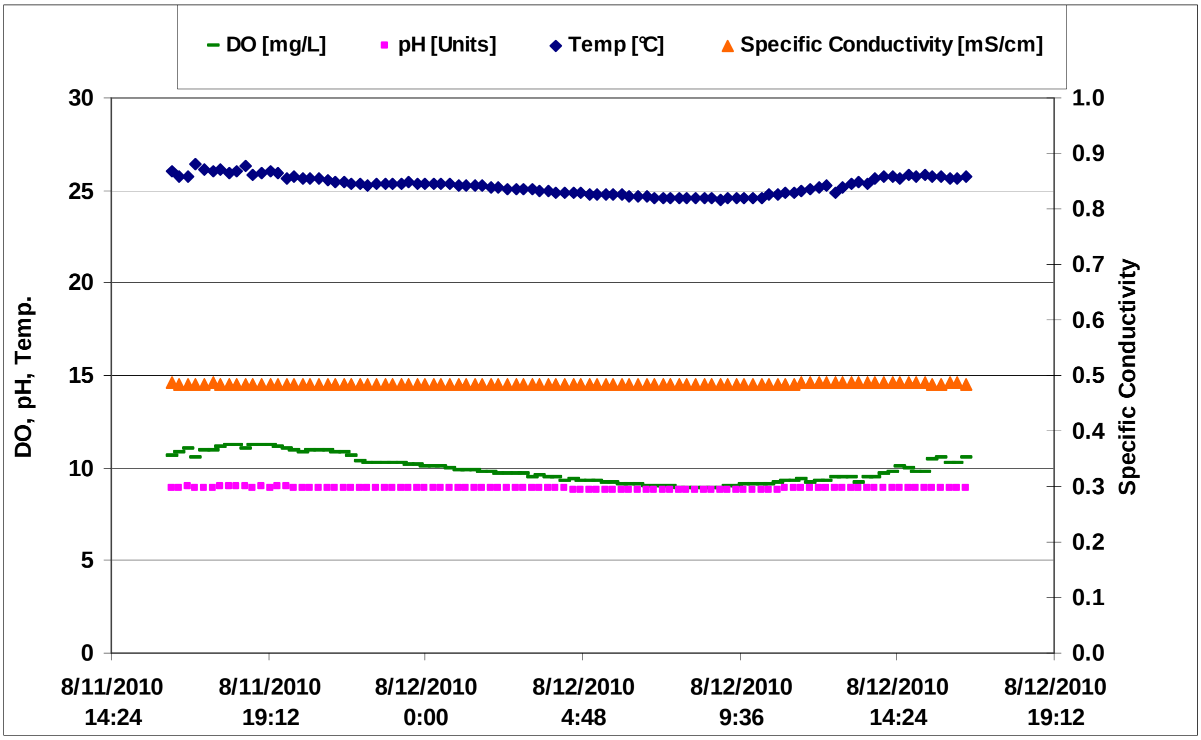

Figure G-55. Profile Data Collected at SFD-3 on August 12, 2010

Depth-profile data were also collected during this water sampling event. Table G-140 summarizes the depth-profile data collected at SFD-1 and SFD-3.

Table G-140. Profile Data Collected in Santa Fe Dam Park Lake (8/12/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
SFD-1	10:40	0.03	26.13	8.49	8.29	0.488	157
		0.53	25.85	8.72	8.62	0.487	158
		0.97	25.54	8.71	8.75	0.488	158
		1.45	25.41	8.69	8.66	0.488	158
		1.96	25.32	8.67	8.52	0.489	157
		2.54	24.3	8.56	8.29	0.488	157
SFD-2	11:25	0.06	26.07	8.73	8.33	0.488	145
		0.46	25.93	8.73	8.49	0.488	144
		1	24.85	8.75	8.93	0.486	144
		1.59	24.64	8.74	8.97	0.485	144
		1.97	24.52	8.73	8.87	0.485	143

Sediment samples were also collected during the August 2010 monitoring event. Table G-141 summarizes these data.

Table G-141. August 12, 2010 Sediment Monitoring Data for Santa Fe Dam Park Lake

Loca -tion	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hard-ness (mg/kg)
SFD-1	11:00	903	7.21	1.79	1.90	0.621	739	2.89	1.08	25.0	48,600
SFD-1D	11:40	1,150	10.4	1.79	1.90	0.584	750	2.86	0.308	24.6	36,200
SFD-3	12:40	855	8.28	1.51	1.60	0.461	842	2.31	0.371	29.9	36,000

G.11.2 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 Santa Fe Dam Park Lake was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Santa Fe Dam Park Lake is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Santa Fe Dam Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-142. The station was located in the southeast end of the lake near the spillway (pink triangle, Figure G-48) (UC Riverside, 1994). Sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 3.5 meters. The range of the 34 dissolved copper samples was between less than 10 µg/L and 56 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 3.5 meters. The 34 samples collected ranged in concentration from less than 1 µg/L to 51 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Santa Fe Dam Park Lake states that copper and lead were not supporting the assessed uses: 37 measurements had a maximum lead concentration of 51 µg/L, a maximum copper concentration of 56 µg/L, and a maximum zinc concentration of 65 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-142. Santa Fe Dam Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
8/10/1992	0	18	<1
	2	18	10
	3.5	13	3
8/10/1992	0	18	2
	2.5	19	2
8/10/1992	0	22	2
	2.5	21	2
9/10/1992	0	<10	2
	2	<10	<1
	3.5	<10	<1
10/13/1992	0	<10	15
	2	<10	4
	3.5	<10	<1
11/3/1992	0	27	3
	1.5	20	2
	2.5	56	2
1/14/1993	0	<10	1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	3.5	<10	<1
2/3/1993	0	<10	<1
	2	<10	<1
	3	<10	<1
3/9/1993	0	<10	8
	2	<10	23
	3.5	<10	2
4/14/1993	0	<10	9
	1.5	<10	37
	2.5	<10	<1
5/25/1993	0	<10	18
	1.5	<10	36
	2.5	<10	12
6/21/1993	0	<10	<1
	1.5	<10	8
	2.5	<10	51

Table G-143 presents 32 additional water column metals samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between March 2009 and August 2010. Samples were collected at locations SFD-1, SFD -2, SFD -3, SFD -4, and SFD-5. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05µg/L, and dissolved zinc detection limit of 0.1 µg/L to 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L; copper concentrations were between 0.6 µg/L and 2.76 µg/L; lead concentrations ranged from < 0.05 µg/L to 0.1 µg/L; and zinc concentrations were <0.1 µg/L to 2.9 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 86 mg/L to 133.2 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-143 were weighted more heavily in the assessment.

Table G-143. Water Column Metals Data for the 2008-2010 Santa Fe Dam Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	3/3/2009	SFD-2 / 3	103.95	<0.2	1.7	<0.1	0.1	average of stations 2 and 3
RB	3/3/2009	SFD-1	106.8	<0.2	1.8	<0.1	0.1	average of replicates
RB	3/3/2009	SFD-4	102.55	<0.2	1.5	<0.1	<2.4	average of duplicates
RB	3/3/2009	SFD-5	101.8	<0.2	1.9	<0.1	0.1	
RB/EPA	8/3/2009	SFD 1	131.3	<0.2	1.9	<0.1	1.9	
RB/EPA	8/3/2009	SFD 2 / 3	132.175	<0.2	1	0.1	0.1	average of replicates and then of sites 2 and 3
RB/EPA	8/3/2009	SFD 4	133.2	<0.2	1.1	0.1	1.1	
RB/EPA	8/3/2009	SFD 5	132.7	<0.2	1.8	0.1	2	
EPA/County	11/17/2009	SFD 4	89.9	<0.2	0.9	<0.1	1.1	
EPA/County	11/17/2009	SFD 5	92.5	<0.2	0.9	<0.1	1.4	
EPA/County	11/17/2009	SFD 3	91.6	<0.2	1	<0.1	1.5	averaged with dup & field filtered
EPA/County	11/17/2009	SFD 1	91.8	<0.2	0.8	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 1	93.55	<0.2	1.4	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 3	89.7	<0.2	1	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 4	91.4	<0.2	0.6	<0.1	<0.1	
County	12/8/2009	SFD 5	87.8	<0.2	1.5	<0.1	0.7	
EPA	12/14/2009	SFD 1	89.35	<0.2	0.7	<0.1	<0.1	average of replicates
EPA	12/14/2009	SFD 3	88.3	<0.2	0.7	<0.1	<0.1	average of replicates
EPA	12/14/2009	SFD 4	90.2	<0.2	0.8	<0.1	<0.1	
EPA	12/14/2009	SFD 5	86	<0.2	0.7	<0.1	<0.1	
County	1/28/2010	SFD 1	101.4	<0.2	0.9	<0.1	<0.1	average of replicates & duplicate

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
County	1/28/2010	SFD 3	100.2	<0.2	0.9	<0.1	<0.1	
County	1/28/2010	SFD 4	100	<0.2	1	<0.1	<0.1	
County	1/28/2010	SFD 5	103.5	<0.2	0.9	<0.1	<0.1	
County	2/17/2010	SFD 1	109.1	<0.2	1.1	0.065	0.65	average of duplicate
County	2/17/2010	SFD 3	110.5	<0.2	1.1	0.07	2.7	
County	2/17/2010	SFD 4	113.1	<0.2	1.15	0.06	1.95	average of replicates
County	2/17/2010	SFD 5	112	<0.2	1.2	0.06	2.9	
EPA	8/12/2010	SFD 1	92.5	<0.2	1.03	<0.05	<0.1	
EPA	8/12/2010	SFD 3	92.9	<0.2	2.76	<0.05	<0.1	
EPA	8/12/2010	SFD 4	NA	<0.2	0.879	<0.05	2.06	
EPA	8/12/2010	SFD 5	NA	<0.2	1.05	<0.05	<0.1	

RB = Regional Board

EPA = USEPA

County = County of Los Angeles

USEPA also collected two sediment samples during the month of August 2010 to further evaluate lake conditions. Table G-144 summarizes the copper and lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target and zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-144. Sediment Metals Data for the August 2010 Santa Fe Dam Park Lake Sampling Event

Organization	Date	Station ID	Copper (mg/kg)	Lead (mg/kg)	Notes
EPA	8/12/2010	SFD 1	14.7	1.76	Average of duplicates
EPA	8/12/2010	SFD 3	5.92	1.49	

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G.12 Monitoring Data for Lake Sherwood

Fish tissue monitoring data relevant to the impairments of Lake Sherwood are available from 1991 top 2007, while water and sediment quality data are available for 2009. Figure G-56 shows the historical and recent monitoring locations for Lake Sherwood.

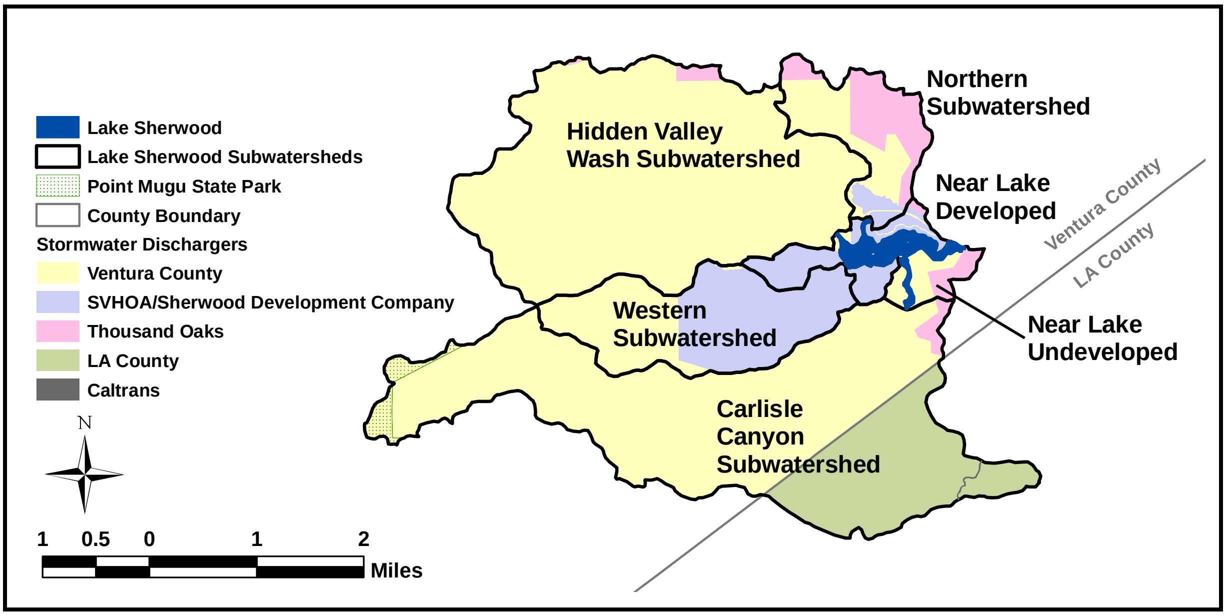

Figure G-56. Lake Sherwood Monitoring Sites

G.12.1 Monitoring Related to Mercury Impairment

Mercury data have been collected in the Lake Sherwood watershed since 1991. Fish tissue concentrations were measured three times under the Toxic Substances Monitoring Program (TSMP) from 1991 to 1997 and by the Regional Board in 2007 (Davis et al., 2008). USEPA and the Regional Board also sampled in-lake and tributary water column and sediment mercury concentrations during two events in 2009. Figure G-56 shows the locations of the water quality monitoring stations.

G.12.1.1 In-Lake Water Quality Monitoring

G.12.1.1.1 Water Column Measurements

USEPA and the Regional Board sampled one station in Lake Sherwood for total and methylmercury in February and July 2009. During the February event, the total depth at this location was 5 meters; samples were collected from 3 meters below the surface. A representative of the Lake Sherwood home owner's association (HOA) provided a boat and accompanied the sampling team. The HOA representative would not allow the sampling team to anchor the boat during sampling, so the engine was left running. The in-lake February sample may therefore be contaminated from the exhaust of the outboard motor. During the July event, samples were collected from a depth of 1 m, and the total depth at this site was 7.8 m. The boat was anchored during this event with the engine turned off.

Table G-145 compares the February and July 2009 water column concentrations observed in Lake Sherwood. In February, the total mercury concentration was 3.32 ng/L, and the methylmercury concentration was 0.189 ng/L or 5.7 percent. In July, the total mercury concentration was 0.75 ng/L and the methylmercury concentration was 0.329 ng/L. The percent of mercury in the methyl form in July was 44 percent. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Supplemental water quality data are included in Table G-146.

Table G-145. In-lake Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
SL-In-lake	2/25/2009	10:00	0.189	3.32	7.1
SL-In-lake	7/13/2009	9:00	0.329	0.75	5.3

Table G-146. Supplemental Water Quality Monitoring for In-lake Samples in Lake Sherwood

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
SL-In-lake	2/25/2009	10:00	73.73	180.03	202	664	6.0
SL-In-lake	7/13/2009	9:00	73.23	200.06	240	752	6.75

Profile data were collected at station SL-In-lake on February 25, 2009 (Figure G-57). Specific conductivity is constant with depth. DO decreases from over 9 mg/L at the surface to 0 mg/L at a depth of 3 meters. pH ranges from 8.0 to 8.6, and temperature ranges from 11.1 ºC to 13.0 ºC. **Note that field operators found DO readings suspicious and have since sent meter off for repair (Greg Nagle, USEPA Region 9, personal communication, 5/22/09).**

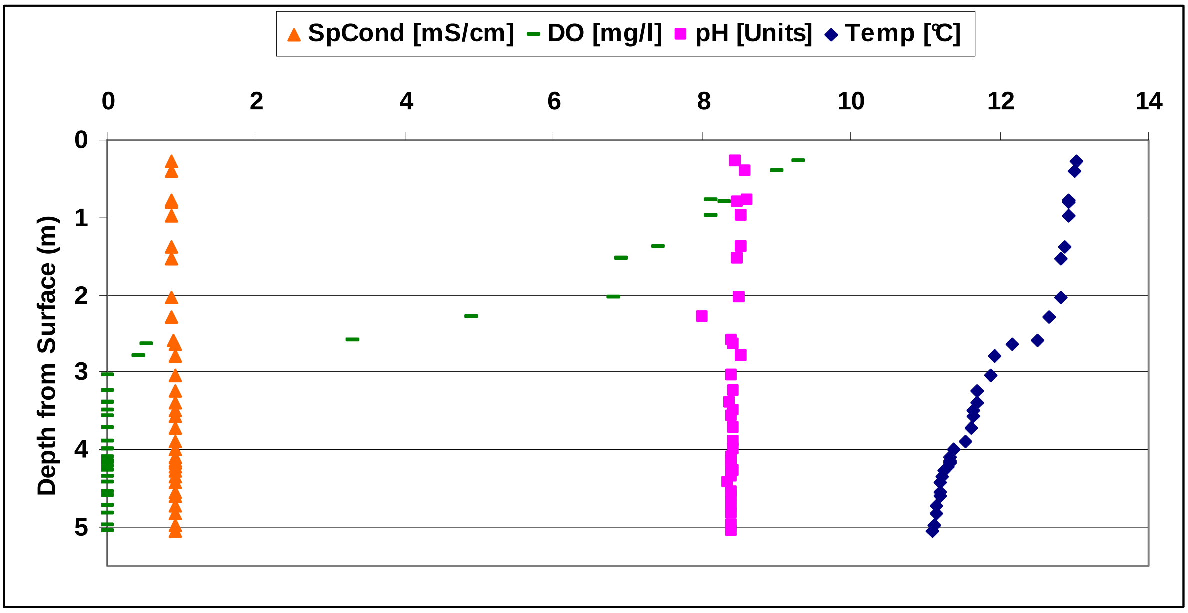

Figure G-57. Profile Data Collected at SL-In-lake on February 25, 2009

Profile data were also collected at station SL-In-lake on July 13, 2009 (Figure G-58). Specific conductivity remained constant with depth. DO decreases from over 10 mg/L at the surface to almost 0 mg/L at a depth of 8 meters. The DO meter was repaired for these readings. pH ranges from 7.5 to 8.8, and temperature ranges from 21.1 ºC to 25.9 ºC

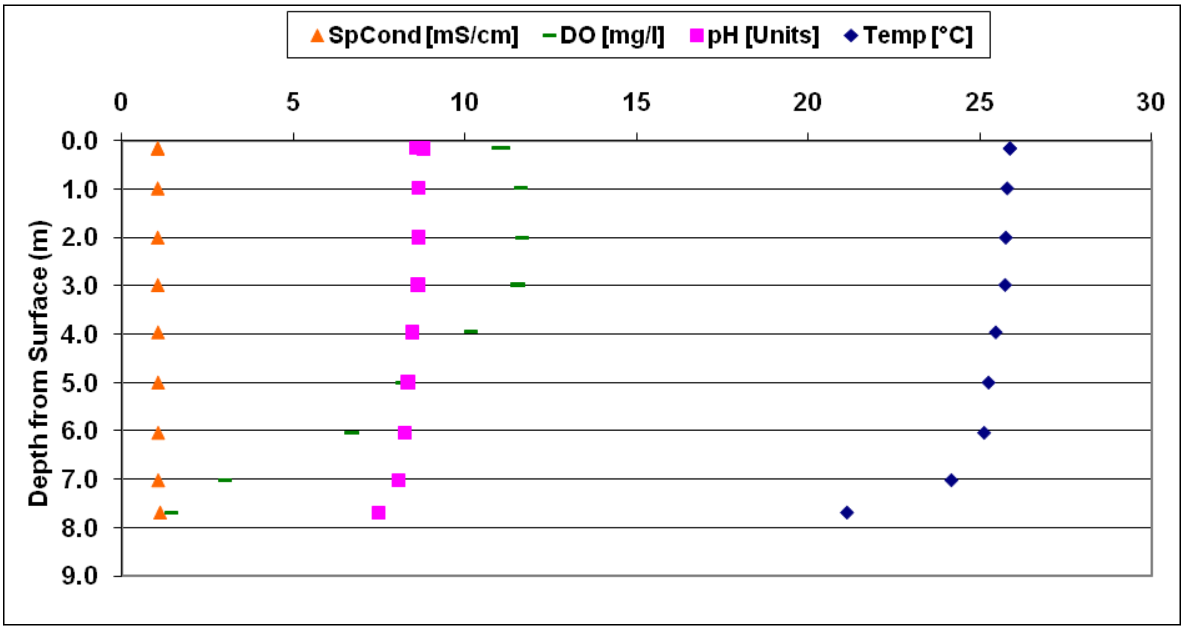

Figure G-58. Profile Data Collected at SL-In-lake on July 13, 2009

G.12.1.1.1 Sediment Samples

USEPA and the Regional Board collected sediment samples from Lake Sherwood to measure total and methylmercury concentrations in sediment. In February, total mercury was analyzed with EPA Method 1631 with a detection limit of 4.96 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.022 µg/kg. The concentrations of total and methylmercury were 470 µg/kg and 0.685 µg/kg, respectively. In July, total mercury was analyzed with EPA Method 1631 with a detection limit of 15.9 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.025 µg/kg. The concentrations of total and methylmercury were 388 µg/kg and 0.599 µg/kg, respectively.

In-lake sediment mercury concentrations for Lake Sherwood are presented in Table G-147. Supplemental data are presented in Table G-148. Concentrations are reported on a dry weight basis.

Table G-147. In-lake Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)
SL-In-lake	2/25/2009	10:00	0.685	470	36.70
SL-In-lake	7/13/2009	9:00	0.599	388	33.96

Table G-148. Supplemental Sediment Data for In-lake Samples in Lake Sherwood

Location	Date	Time	Sulfate (mg/kg)	Total Organic Carbon (percent of dry weight)
SL-In-lake	2/25/2009	10:00	481.93	3.38
SL-In-lake	7/13/2009	9:00	218.99	5.15

G.12.1.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in Lake Sherwood since 1991. The TSMP sampled individual fish three times. The SWAMP sampled individual fish during the summer of 2007 and April 2010. The Sherwood Valley HOA sampled five individual fish in 2007 as well (Weston Solutions, 2007); length data were not retained during analysis. Fillet and liver tissue were analyzed. Table G-149 presents the fish tissue mercury concentrations on a wet weight basis; liver concentrations are not included. Concentrations range from 0.214 ppm to 1.6 ppm. The applicable fish tissue guideline for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-149. Fish Tissue Mercury Concentrations Measured in Lake Sherwood Large Mouth Bass

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	4/22/1991	356	0.700
TSMP	4/21/1992	286	1.600
TSMP	7/17/1997	349	0.214
SWAMP	Summer 2007	205	0.219
SWAMP	Summer 2007	242	0.239
SWAMP	Summer 2007	261	0.325
SWAMP	Summer 2007	284	0.236
SWAMP	Summer 2007	305	0.362
SWAMP	Summer 2007	321	0.322
SWAMP	Summer 2007	365	0.802
SWAMP	Summer 2007	345	0.751
SWAMP	Summer 2007	353	0.601
SWAMP	Summer 2007	318	0.444
SWAMP	Summer 2007	328	0.464
SWAMP	Summer 2007	349	0.504
SWAMP	Summer 2007	339	0.607
SWAMP	Summer 2007	386	0.552
SWAMP	Summer 2007	418	0.802

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	452	0.665
Sherwood HOA	Summer 2007	Length data not available	0.465
Sherwood HOA	Summer 2007	Length data not available	0.670
Sherwood HOA	Summer 2007	Length data not available	0.319
Sherwood HOA	Summer 2007	Length data not available	0.284
Sherwood HOA	Summer 2007	Length data not available	0.409
SWAMP	4/19/2010	417	1.02
SWAMP	4/19/2010	385	0.664
SWAMP	4/19/2010	374	0.824
SWAMP	4/19/2010	368	0.994
SWAMP	4/19/2010	357	1.09

Piscivorous fish tend to have increased mercury tissue concentrations with age. Figure G-59 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. As expected, fish tissue mercury concentrations increase with length. All fish specimens with a mean or individual length greater than 205 mm exceed the fish tissue target of 0.22 mg/kg, with the exception of one sample that had a concentration of 0.214 ppm and a length of 349 mm. Of the samples with corresponding length data, 22 exceeded the fish tissue target.

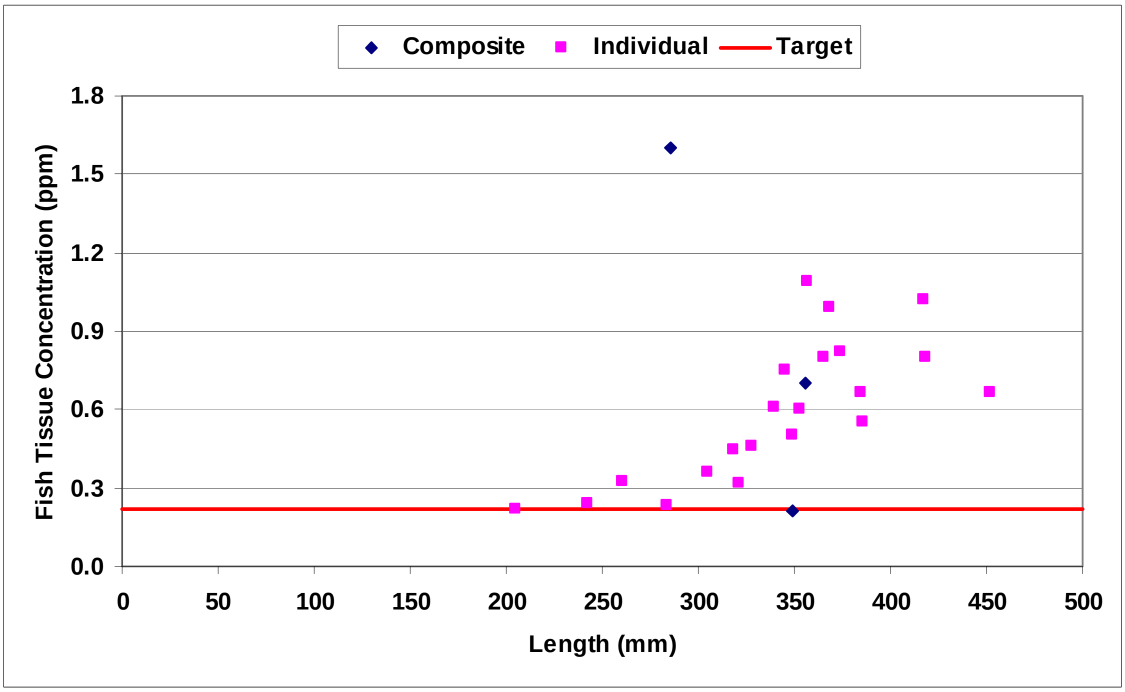

Figure G-59. Mercury Concentrations in Largemouth Bass in Lake Sherwood

SWAMP also collects data on mercury concentration in redear sunfish. Table G-150 provides composite results for redear sunfish collected in April 2010.

Table G-150. Composite Fish Tissue Mercury Concentrations Measured in Lake Sherwood Redear Sunfish

Program	Date	Average Fish Length (mm)	Number of Fish per Composite	Total Mercury Concentration (ppm wet weight)
SWAMP	4/19/2010	289	5	0.140
SWAMP	4/19/2010	291	5	0.185
SWAMP	4/19/2010	291	5	0.169

G.12.1.3 Tributary/Inflow Monitoring

G.12.1.1.3 Water Column Measurements

In February 2009, USEPA and the Regional Board sampled water column total and methylmercury concentrations from two tributaries and one storm drain. However, the temperature requirements for the methylmercury sample collected at the storm drain (SL-8) were not met, so the measured methylmercury concentration may be compromised. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. The two tributary samples (SL-3 and SL-6) had total mercury concentrations ranging from 2.96 ng/L to 6.00 ng/L. The storm drain (SL-8) had a higher total mercury concentration of 23.9 ng/L. Methylmercury concentrations in the tributary samples ranged from 0.157 ng/L to 0.216 ng/L. Methylmercury in the storm drain sample was an order of magnitude lower, but this sample was compromised and may not be accurate.

Inflow water column measurements were collected again in the summer of 2009. The tributary at SL-6 was not flowing, so a sample was not collected during the July event. The storm drain at SL-8 had methyl and total mercury concentrations of 0.096 ng/L and 54.0 ng/L, respectively. The creek flowing through the golf course community was sampled at SL-3; a duplicate sample was analyzed for total mercury. The forebay at the outlet of Hidden Valley Wash (SL-7) had methyl and total mercury concentrations of 3.41 ng/L and 11.3 ng/L, respectively. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 0.15 ng/L to 0.73 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Table G-151 presents the results of the water column mercury and TSS concentrations measured in the tributaries and storm drains to Lake Sherwood. The tributary flowing through the mountainous subwatershed that discharges to the south side of the lake had the lowest concentrations of methyl and total mercury during the winter sampling event; this tributary was not flowing during the summer event. The highest concentrations of total mercury were observed in storm drain SL-8. Methylmercury concentrations were highest in the forebay at the outlet of Hidden Valley Wash. This site (SL-7) was identified as a potential methylation hot spot based on sediment samples collected in February 2009 (see discussion in Section G.12.1.1.3. Table G-152 presents the supplemental water quality data.

Table G-151. Tributary/Inflow Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
SL-3	2/25/2009	13:00	0.157	6.00	1.0
SL-6		11:00	0.216	2.96	2.9
SL-8		11:45	0.025[1]	23.9	2.2
SL-8	7/13/2009	10:00	0.096	54.0	5.1
SL-3		8:55	0.536	4.58	2.1
SL-3D		8:55	NA	4.63	NA
SL-7		10:15	3.41	11.3	20.3

[1] Temperature requirements for methylmercury analysis not met.

Table G-152. Supplemental Water Quality Monitoring for Inflow Samples for Lake Sherwood

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
SL-3	2/25/2009	13:00	134.61	384.7	262	1,094	4.7
SL-6		11:00	55.68	146.65	206	578	4.1
SL-8		11:45	180.97	488.44	238	1,310	5.6
SL-8	7/13/2009	10:00	156.33	271.61	248	1036	5.5
SL-3		8:55	190.1	573.81	346	1628	5.2
SL-7		10:15	76.46	181.58	192	714	7.7

G.12.1.1.3 Sediment Samples

Sediment samples were collected from three tributaries (SL-3, SL-6, and SL-7) and one storm drain (SL-5) during the February 2009 monitoring event. Samples SL-3 and SL-6 represented flowing water through developed and undeveloped areas, respectively. Total and methylmercury sediment concentrations at these two sites were much lower than site SL-7, which was intended to represent the Hidden Valley Wash tributary. This tributary appears to be piped under Janss Road prior to discharging to Lake Sherwood. The outlet of the pipe is beneath the surface of a stagnant backwater area adjacent to the Lake. The sediment mercury concentration of this sample may be more reflective of a wetland area than the sediment being delivered from the upland areas draining to Hidden Valley Wash. This is particularly true of the methylmercury sediment concentration which is an order of magnitude greater than those measured in the other inputs or Lake Sherwood itself. Though the methyl and total mercury concentrations at site SL-7 may not be accurate for estimating loading from Hidden Valley Wash, they do identify a potential location of high rates of methylation that may be increasing the bioavailability of mercury to the aquatic life in Lake Sherwood. Typical hotspots for methylation include wetlands, where sediments alternate between wet and dry conditions. Based on two reconnaissance events conducted for Lake Sherwood, this backwater area undergoes both dry (January 2009) and wet/stagnant (February 2009) periods.

In July 2009, sediment samples were collected from four locations. Duplicate total mercury samples were collected at SL-3. The lowest total mercury concentrations (approximately 60 µg/kg) were observed at

SL-PR (upstream of SL-7 on Hidden Valley Wash at Potrero Road) and SL-5. The highest total mercury concentrations were measured at SL-3 and SL-7. Methylmercury concentrations ranged from 0.397 µg/kg to 0.657 µg/kg at SL-3, SL-5, and SL-7 with the highest concentration (0.696) measured at SL-8. Concentrations were much lower at SL-PR and were equivalent to the detection limit for that sample.

Sediment mercury concentrations collected from the inputs and adjacent area of Lake Sherwood are presented in Table G-153. Concentrations are reported on a dry weight basis. Table G-154 presents the supplemental sediment quality data.

Table G-153. Inflow Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)
SL-3	2/25/2009	13:00	0.269	92.7	77.70
SL-6		11:00	0.136	129	75.90
SL-5		13:15	0.145	51.0	82.62
SL-7[1]		08:30	2.53	243	74.25
SL-3	7/13/2009	8:55	0.397	392	30.30
SL-3D		8:55	NA	265	34.45
SL-5		9:45	0.657	62.9	96.80
SL-7		10:15	0.453	275	73.18
SL-PR		10:50	0.009	60.3	98.82
SL-8		10:00	0.696	63.3	74.52

[1] This sample is likely not representative of the sediment methylmercury concentrations delivered from Hidden Valley Wash.

Table G-154. Supplemental Sediment Data for Inflow Samples to Lake Sherwood

Location	Date	Time	Sulfate (mg/kg	Total Organic Carbon (percent of dry weight)
SL-3	2/25/2009	13:00	157.54	0.24
SL-6		11:00	125.09	0.58
SL-5		13:15	92.76	1.44
SL-7		08:30	108.98	1.67
SL-3	7/13/2009	8:55	1,106.74	10.15
SL-5		9:45	903.53	3.93
SL-7		10:15	93.23	0.68
SL-PR		10:50	9.3	1.64
SL-8		10:00	41.69	2.35

G.13 Monitoring Data for Westlake

Monitoring data relevant to the impairments of Westlake Lake are available for 1992, 1993, 2009, and 2010. Figure G-60 shows the historical and recent monitoring locations for Westlake Lake.

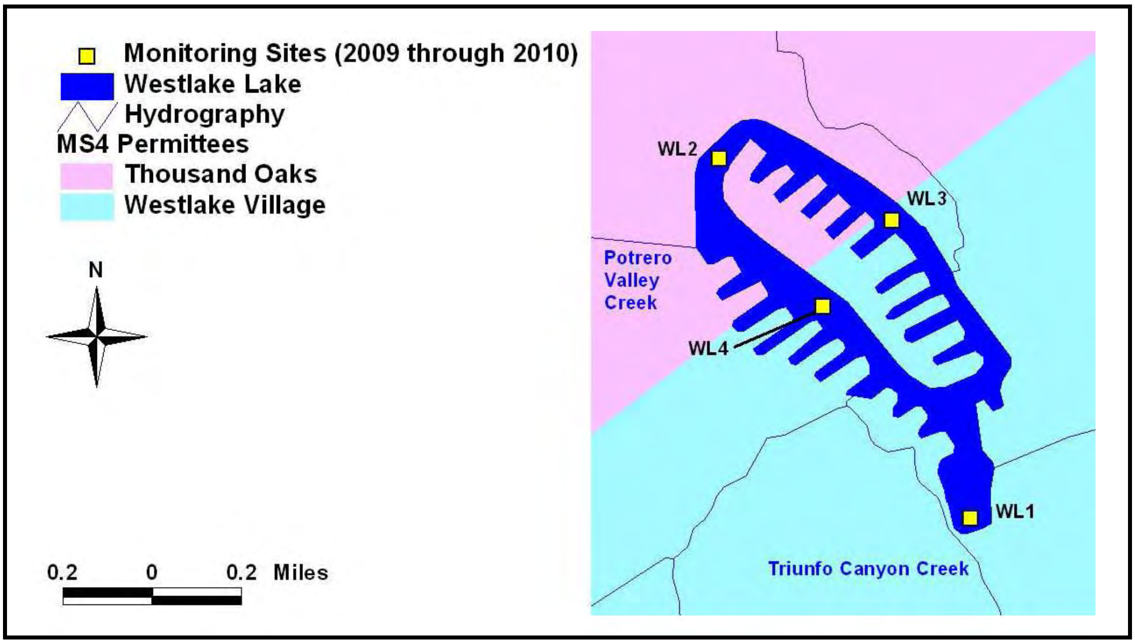

Figure G-60. Westlake Lake Monitoring Sites

G.13.1 MONITORING RELATED TO METALS IMPAIRMENT

In 1996 Westlake Lake was impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Westlake Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Westlake Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are presented in Table G-155. Samples were collected near the outlet of the lake (WL1) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to six meters. The range of the 52 dissolved copper samples was between less than 10 µg/L and 56 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to six meters. The 52 samples collected ranged in concentration from less than 1 µg/L to 91 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Westlake Lake states that copper and lead were not supporting the assessed uses (copper has since been delisted): 52 measurements had a maximum lead concentration of 91 µg/L, a maximum copper concentration of 56 µg/L, and a maximum zinc concentration of 12 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-155. Westlake Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
8/3/1992	0	21	<1
	2	21	<1
	4	19	1
	6	13	4
8/3/1992	0	25	2
	2	21	<1
8/3/1992	0	42	<1
	2.5	28	2
8/18/1992	0	47	<1
	2.5	47	<1
	4	36	<1
	6	21	<1
9/23/1992	0	55	22
	1.5	33	6
	4	25	4
	6	21	2
10/14/1992	0	48	<1
	2	46	<1
	4	46	<1
	6	44	<1
11/10/1992	0	24	1
	2	34	1
	4	37	1
	6	54	2
12/14/1992	0	31	11
	1.5	29	5
	3	42	6
	6	56	13
1/20/1993	0	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	2	<10	<1
	4	<10	<1
	6	<10	<1
2/24/1993	0	<10	<1
	2	<10	<1
	4	12	<1
	6	<10	<1
3/10/1993	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
4/19/1993	0	26	1
	2.5	30	33
	3.5	27	8
	4.5	18	6
5/19/1993	0	30	91
	2.5	31	27
	4.5	26	9
	6.5	19	17
6/28/1993	0	36	19
	2	33	2
	4	29	<1
	6	29	<1

Table G-156 presents 24 additional metals samples that were collected by USEPA and the Regional Board between March 2009 and October 2010. Samples were collected at locations WL-1, WL-2, WL-3, and WL-4. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.4 µg/L; copper concentrations ranged from 2.5 µg/L to 8.9 µg/L; lead concentrations were between <0.05 µg/L and 0.065 µg/L; and zinc concentrations ranged from <0.1 µg/L to 5.45 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 231 mg/L to 477 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-156 were weighted more heavily in the assessment.

Table G-156. Metals Data for the 2009-2010 Westlake Lake Sampling Events

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
3/26/2009	WL 1	348.58	<0.2	5.78	<0.05	1.18	average of duplicates and replicates
3/26/2009	WL 2	353.60	<0.2	5.50	<0.05	1.40	
3/26/2009	WL 3	343.80	<0.2	6.00	<0.05	2.20	
3/26/2009	WL 4	347.70	<0.2	5.60	<0.05	0.80	
7/17/2009	WL 1	469.77	<0.2	6.87	0.05	0.57	average of duplicates and replicates
7/17/2009	WL 2	477.00	<0.2	8.90	<0.05	<0.10	
7/17/2009	WL 3	466.00	<0.2	7.30	<0.05	0.40	
7/17/2009	WL 4	469.25	<0.2	8.15	<0.05	<0.10	average of replicates
12/17/2009	WL 1	382.6	<0.2	4.9	0.055	5.2	average of replicates
12/17/2009	WL 2	382.9	<0.2	4.4	0.065	5.45	average of duplicates
12/17/2009	WL 3	351	<0.2	4.8	0.05	4.7	
12/17/2009	WL 4	388.2	<0.2	4.1	0.05	1.3	
1/26/2010	WL 1	246.1	<0.2	3.4	<0.05	1.55	average of replicates
1/26/2010	WL 2	243.3	<0.2	2.5	<0.05	4.05	average of duplicates
1/26/2010	WL 3	231.5	<0.2	3.25	<0.05	1.6	
1/26/2010	WL 4	256.3	<0.2	2.8	<0.05	0.6	
8/13/2010	WL 1	333	0.409	4.46	<0.05	2.61	
8/13/2010	WL 2	334	<0.2	4.10	<0.05	<0.1	
8/13/2010	WL 2D	334	0.407	4.10	<0.05	<0.1	
8/13/2010	WL 3	332	ND	4.08	<0.05	0.748	
8/13/2010	WL 4	331	ND	4.14	<0.05	<0.1	
10/1/2010	WL 1	337	<0.2	5.96	<0.05	<0.1	
10/1/2010	WL 2	335	<0.2	5.95	<0.05	<0.1	
10/1/2010	WL 3	328	<0.2	4.99	<0.05	<0.1	
10/1/2010	WL 4	335	<0.2	6.34	<0.05	<0.1	

Note: all sampling performed by the Regional Board and/or USEPA.

USEPA also collected two sediment samples during August 2010 to further evaluate lake conditions. Table G-157 summarizes the lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-157. Sediment Metals Data for August 2010 West Lake Sampling Event

Organization	Date	Station ID	Lead (mg/kg)	Notes
EPA	08/13/2010	WL1	31.1	
EPA	08/13/2010	WL2	83.1	Average of duplicates

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G.14 References

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Appendix H. Methodology for Organochlorine Pesticides and PCBs TMDL Development

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H.1 Introduction

USEPA Region IX is establishing Total Maximum Daily Loads (TMDLs) for impairments in nine lakes in the Los Angeles Region (Figure H-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

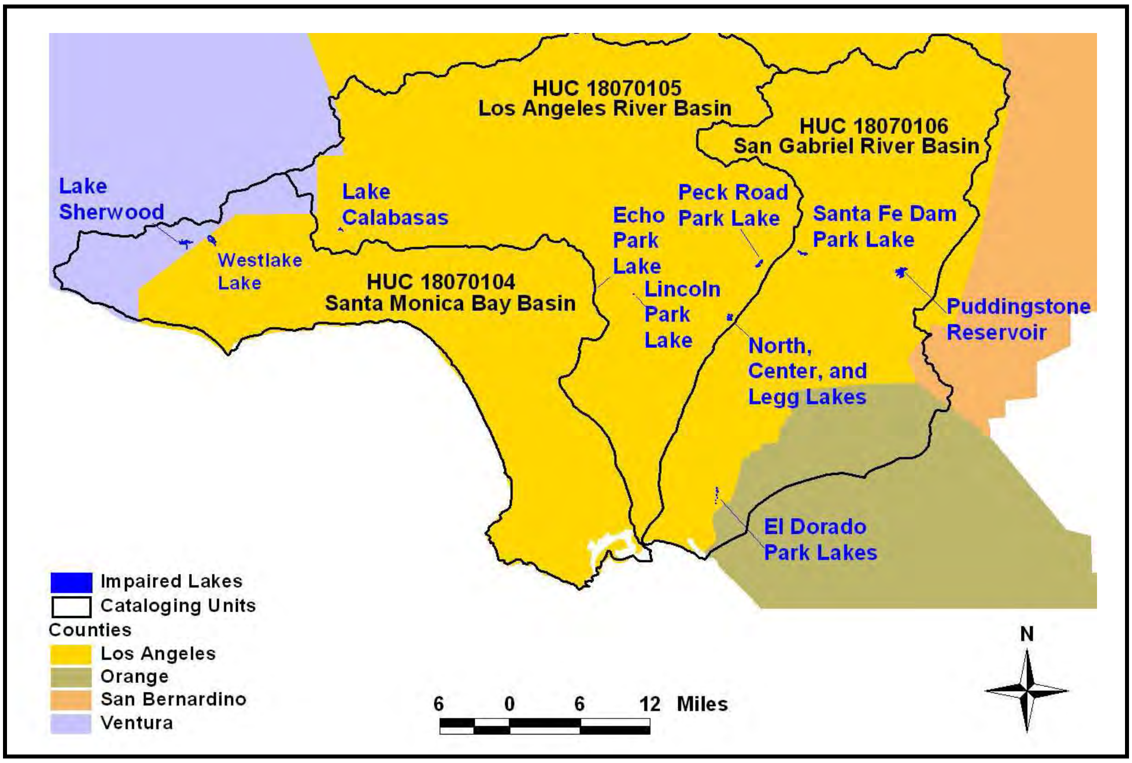

Figure H-1. Location of 10 TMDL Lakes in the Los Angeles Region

Three of these waterbodies are listed as impaired by Organochlorine (OC) Pesticides and PCBs due to elevated fish tissue concentrations: Echo Park Lake, Peck Road Park Lake, and Puddingstone Reservoir. Puddingstone Reservoir was listed for fish tissue concentrations of chlordane, DDT, and PCBs in 1996 and 1998 based on data collected by the Toxic Substance Monitoring Program (TSMP). The listings were carried over to the 2008-2010 303(d) list. The TSMP fish data were also used as the basis for listing PCBs in Echo Park Lake and chlordane and DDT in Peck Road Park Lake. These listings began in 1996 and were also listed on the 1998, 2002, 2006, and 2008-2010 303(d) lists. Recently collected data revealed other impairments not included in the 2008-2010 303(d) listings, but requiring remedial efforts. PCB and dieldrin impairments were identified in Peck Road Park Lake, a dieldrin impairment was identified in Puddingstone Reservoir, and chlordane and dieldrin impairments were found in Echo Park Lake based on fish tissue contamination found in 2004, 2007, and/or 2010 data collected for the Surface Water Ambient Monitoring Program (SWAMP) study. The basis for listings in each lake is shown in Table H-1.

The TMDLs developed for fish tissue contaminations will also reduce OC Pesticides and PCBs in the sediment and water. This appendix discusses the methods used to calculate TMDLs based on the measured tissue concentrations observed in each waterbody. The lake-specific chapters describe data, results, and allocations associated with Echo Park Lake, Peck Road Park Lake, and Puddingstone Reservoir.

Table H-1. OC Pesticides and PCBs Impairments in Los Angeles Region Lakes

Lake	Chlordane	DDT	Dieldrin	Total PCBs
Echo Park Lake	○		○	●
Peck Road Park Lake	●	●	○	○
Puddingstone	●	●	○	●

● Impairment included in both the consent decree and 2008-2010 303(d) list.

○ Impairment identified by new data analyses (after the 2008-2010 303(d) list data cutoff).

H.2 Conceptual Model

Storage in the sediment accounts for the major fraction of OC Pesticides and PCBs in most lake systems. The cycling of OC Pesticides and PCBs between sediment, benthic biota, and aquatic organisms is illustrated in the conceptual model in Figure H-2. The figure illustrates the direct uptake of pollutants by filter feeders and benthic organisms (via adsorption or ingestion) and the indirect uptake of pollutants in fish by consumption of contaminated benthic organisms. Most of the OC Pesticides and PCBs mass that is not incorporated in the aquatic lifecycle will travel through a settling-resuspension cycle of lake particulates. Other transport cycles are also shown in Figure H-2 or described in the Linkage Analysis discussion (Section H.4).

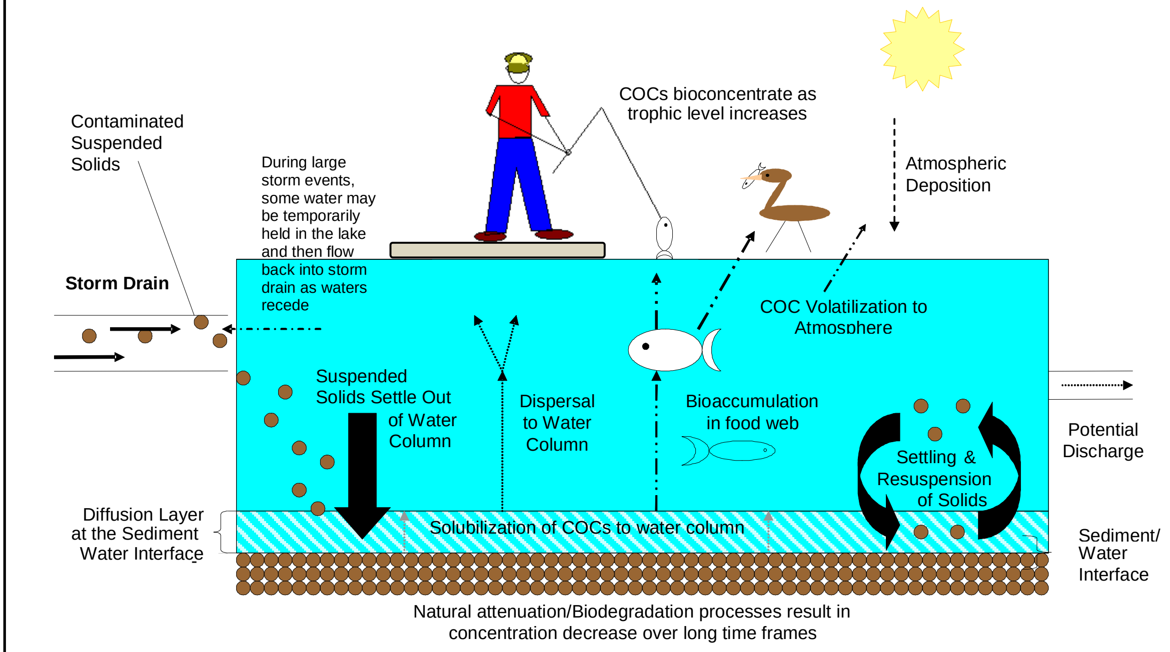

Figure H-2. Conceptual Model for OC Pesticides and PCBs Mobilization

The remainder of this section provides a summary of brief background information on the pollutants addressed in the TMDLs for organic compounds impairments. For the OC Pesticides and PCBs, the general uses and sources of the chemical are explained. There is an abundance of literature for each contaminant's history, chemical characteristics, and toxicological effects, which are rudimentarily summarized for this background.

H.2.1 ORGANOCHLORINE PESTICIDES

Organochlorine (OC) pesticides describes a large collection of pesticides synthetically generated and composed of an organic chemical with at least one chlorine atom. OC pesticides include aldrin, chlordane, DDT, dicofol, dieldrin, endosulfan, endrin, heptachlor, mirex, and toxaphene. The use of OC pesticides was widespread between 1940-1980, at which point, most OCs were banned in the United States (Kalkhoff and Van Metre, 2009). This group of pesticides is often referred to as legacy pesticides, as they continue to persist in the environment long after their initial entry. The OC pesticides addressed for the lake TMDLs are chlordane, dichlorodiphenyltrichloroethane (DDT), and dieldrin. Many of the OC pesticides, including the chemicals of concern here, are nonpolar and highly lipophilic (Connell, 2005), giving them a propensity to bioaccumulate in fats (lipids) in fish tissue.

H.2.1.1 DDT

Dichlorodiphenyltrichloroethane (DDT) is a synthetic organochlorine insecticide once used throughout the world to control insects. Technically DDT consists of two isomers, 4,4'-DDT and 2,4'-DDT, of which the former is the most toxic. In the environment, DDT breaks down to form two related compounds: DDD (tetrachlorodiphenylethane) and DDE (dichlorodiphenyl-dichloroethylene). The sum of DDT, DDD, and DDE is referred to as total DDTs. DDT and its degradation products are colorless crystalline solids and exhibit physical properties of low water solubility and high lipophilicity, which play a key role in its environmental fate (LARWQCB, 2009a; LARWQCB, 2009b). DDT became widely used as a pesticide in 1939. During World War II, its use was focused on controlling disease-carrying insects, such as mosquitoes and lice (USEPA, 1975). DDT for agricultural and commercial uses started after 1945. Use of DDT peaked in 1959, at which time approximately 80 million pounds were being applied annually. In California, DDT was widely used for control of both agricultural and disease-carrying pests (Mischke et al., 1985). In 1963 the California Department of Food and Agriculture (CDFA) declared DDT a restricted material. The last year that substantial amounts of DDT were applied in California was 1970, when roughly 1.2 million pounds of DDT were applied, primarily to agricultural areas (Mischke et al., 1985).

The overall use of DDT started to decline in the early 1970s because of restrictions and reporting uses, in addition to the developed resistance of the pests that were previously sensitive to DDT (USEPA, 1975). Furthermore, new more effective pesticides had been developed, and there was growing public concern over adverse human and environmental health effects from DDT exposure (USEPA, 1975). Even though domestic usage of DDT has been banned for more than 30 years, there are still widespread environmental impairments caused by DDT and DDT-associated degradation products.

Because DDT exhibits such low water solubility, it is mainly concentrated in soils and will bind strongly to the organic fraction of sediments (Walker et al., 2001). DDT has an estimated half-life in soil of two to sixteen years (Connell, 2005). DDT is transported to surface waterbodies through the sediment and erosion runoff. DDT in the water column will remain partitioned to sediment or other organic mediums (living organisms).

DDT is also highly lipophilic and will accumulate in the fatty tissues of exposed wildlife and biomagnify as it moves through the food chain to reach the primary predator (NPIC, 1999). The ability of DDT to biomagnify is one of the primary environmental concerns of this pollutant because the exposure increases from one trophic level to another.

H.2.1.2 Chlordane

Chlordane is a white solid pesticide that was first registered and approved for agricultural and non-agricultural uses in the United States in 1948. Chlordane is actually a generally encompassing term used to describe technical chlordane, the common pesticide formula which is composed of over 50 different closely-related compounds. Technical chlordane includes heptachlor, nonachlor, chlordane and similar chemicals. The true chlordane compound composes roughly 40 percent of the technical mixture in two isomers: alpha-chlordane and gamma-chlordane (NPIC, 2001).

Non-agricultural uses of chlordane included treating pests in residential lawns and gardens as well as structural pests such as termites. Chlordane was used on a variety of agricultural crops including corn, citrus, deciduous fruits and nuts, and vegetables. USEPA banned the use of chlordane on all food crops, lawns, and gardens in 1978. It was still registered as a termiticide until 1988, when USEPA expanded the chlordane ban to all uses (USEPA, 2009a).

As an organochlorine pesticide, chlordane has similar properties to DDT. It has low water solubility, a strong binding affinity to soil particles, and is persistent in the environment, with a half-life in soils of approximately four years (EXTOXNET, 1996). Soils historically treated with chlordane can continue to be a present source of chlordane in the environment and contaminated soils can be transported to waterbodies via runoff. Moreover, chlordane will bioaccumulate in the fat tissue of exposed organisms and is considered highly toxic to fish and freshwater invertebrates (NPIC, 2001; EXTOXNET, 1996).

H.2.1.3 Dieldrin

Dieldrin is a man-made organochlorine pesticide product, but can also be produced through the natural and metabolic degradation of aldrin, another organochlorine pesticide (USEPA, 2008). Dieldrin was originally developed as an alternative to DDT and mainly used between 1950 and 1970. It was applied to structures for termite control and used in agriculture for control of soil insects such as corn rootworms, cutworms, and locusts in citrus, corn, and cotton crops (ATSDR, 2002; USEPA, 2008). Use of dieldrin peaked in 1966 at one million pounds and dropped to 670,000 pounds in 1970, during the same period the use of aldrin dropped from 19 million pounds to 10.5 million pounds (USEPA, 1980).

In 1970, all registered uses for both pesticides were cancelled by the US Department of Agriculture (USDA), but the USEPA lifted the cancellation under the authority of the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) in 1972 for deep ground insertions for termite control, nursery clippings of roots and tops of non-food plants, and moth-proofing. In 1974, the manufacturing of aldrin and dieldrin was suspended, and in 1987 all uses of dieldrin were cancelled (USEPA, 2008).

Dieldrin is resistant to biotic and abiotic degradation, becomes sequestered in the soil with time, and therefore persists in the environment. The half life in soils is between six months and three years (Connell, 2005; Alexander, 1999). Similar to the other organochlorine pesticides, dieldrin also has a strong affinity to soil particles and lipids and a low solubility in water. The most common exposure routes of dieldrin are from living in houses treated with dieldrin to control termites and consumption of root crops, fish, and seafood. Dieldrin has a wide range of suspected negative effects in living organisms. Most often in humans, dieldrin damages functions of the nervous system (ASTDR, 2002).

H.2.2 POLYCHLORINATED BIPHENYLS

Polychlorinated biphenyls (PCBs) consist of two phenyl rings with from one to ten chlorine atoms attached. Individual PCB compounds, referred to as congeners, vary in the number and placement of the chlorine atoms. There are a total of 209 possible congeners, which vary in physical properties and toxicity (ATSDR, 2001). Some commercial mixtures of PCBs are known by the trade name Aroclor. Most PCBs are oily liquids or waxy solids, and some can exist as a vapor in air (ATSDR, 2001; USEPA, 2009b). There are no natural sources of PCBs.

PCBs were manufactured in the U.S. from 1929 until production was banned in 1979. The cumulative production of PCBs in the United States from 1930 to 1985 is estimated at 1.4 billion pounds (USEPA, 2010). PCBs were used for a variety of applications and functions, including coolants and lubricants in transformers, capacitors, and other electrical equipment; heat transfer and hydraulic fluids; fluorescent light ballasts; cable insulation and thermal insulation; adhesives and tape; varnishes, surface coatings and paints; caulking; plastics; and carbonless copy paper (USEPA, 2009b). Useful characteristics, such as non-flammability, chemical stability, and insulating ability, resulted in use of PCBs for myriad industrial and commercial purposes (USEPA, 2009b).

Prior to the 1979 ban on manufacturing, PCBs were released into the environment during their production and various uses (USEPA, 2009b). USEPA regulates PCBs under the Toxic Substances Control Act (TSCA), which generally bans the manufacture, use, and distribution in commerce of the chemicals in products at concentrations of 50 parts per million or more. TSCA allows USEPA to authorize certain continued uses of PCBs, such as to rebuild existing electrical transformers during the transformers' useful life, which may be 30 years or more. PCBs are also still present in older materials made prior to 1979, such as paint, and caulking (USEPA, 1999).

PCBs enter the environment through improper disposal of industrial waste; releases or leachate from abandoned manufacturing areas and waste sites; and leaks and/or improper dumping of materials containing PCBs. Global cycling of PCBs occurs when they volatilize from soils and/or surface waters, are transported into the atmosphere, and are then redeposited to land and surface waters (USEPA, 1999; ATSDR, 2001). This process plays an important role in the transport and deposition of PCBs to surface waters (USEPA, 1999; USEPA, 2009b).

PCBs have low water solubility and are highly lipophilic, with variation dependent on the characteristics of the individual congeners (USEPA, 1999). PCBs bind strongly to soils and natural organic matter, which can be transported to surface waters through runoff (USEPA, 1999). Because of their high lipophilicity, PCBs are stored in the fat tissue of exposed organisms and bioaccumulate through the food chain. Bioconcentration factors generally increase with chlorine content of the congeners. Because PCBs concentrate in the food chain, a small concentration in water or sediment can produce a significant environmental impact.

PCBs are resistant to abiotic and biotic degradation and the resistance increases as the chlorination of the compound increases. Historical loads of PCBs, stored in lake sediments, can continue to contaminate the aquatic food chain for many decades.

H.3 Source Assessment

The OC Pesticides and PCBs addressed in these TMDLs are no longer in production and use is either banned or strictly limited. For this reason, loading to the lakes is expected to have declined over time, and the historic loads that are stored in lake sediment appear to be the major source of bioaccumulation in fish. Nonetheless, any ongoing loads must also be addressed in the TMDL.

Sources of chlordane, DDT, dieldrin, and PCBs that cause contamination in a waterbody may include both point and nonpoint sources. Federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which waste load allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). Continuing loads of OC Pesticides and PCBs into the lakes is from permitted stormwater discharges by municipalities. Chlordane, DDT and dieldrin are expected to be in highest concentrations near agricultural land, on which pesticides and insecticides were used heavily. Older industrial sites are more likely to contain PCBs where they were used or integrated into substances such as coolants, lubricants, and surface coatings. Older residential areas are also potential sources of PCBs and organochlorine pesticides.

H.3.1 POINT SOURCES

Discharges that occur at one or more defined points, such as a pipe or storm drain outlet, are defined as point sources. Most point sources are regulated through the NPDES permitting process.

H.3.1.1 MS4 Permittees

In 1990, USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed into the Municipal Separate Storm Sewer Systems (MS4) by stormwater runoff, or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

OC pesticides and PCB loads from urban stormwater runoff and associated sediment are estimated from monitoring data collected from the lake sediments near drainage inputs (Appendix G, Monitoring Data) and simulated sediment loads from a previously developed LSPC model of the San Gabriel and Los Angeles river basins (Appendix D, Wet Weather Loading) (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. The LSPC model results and estimated sediment loading for each contributing MS4 system are described in further detail in Appendix D (Wet Weather Loading).

Because OC Pesticides and PCBs are strongly sorbed to sediment, loading and transport during dry weather flow is assumed to be insignificant. Therefore, loading estimates are based on sediment delivery and no separate load calculation is performed for dry weather flows.

H.3.1.2 Other NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). Loading of OC Pesticides and PCBs from non-MS4 NPDES discharges is expected to be negligible because the contaminants addressed in these TMDLs are no longer in use and have been banned for over 20 years. To quantify OC Pesticides and PCBs loading from non-MS4 discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the Los Angeles and San Gabriel river basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the three OC Pesticides and PCBs-impaired lakes. OC Pesticides and PCBs loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake-specific sections of this report.

H.3.1.3 Additional Inputs

One of the lakes addressed by these TMDLs has supplemental water additions from groundwater wells or potable water that maintain its lake level. Access and monitoring data for these inputs are limited and no specific OC Pesticides and PCBs analyses are available. OC Pesticides and PCBs loading from unknown inputs are encompassed in the calculated loading because the loadings are based on the observed data, which capture all sources upstream of the monitoring station.

H.3.2 NONPOINT SOURCES

OC Pesticides and PCBs loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

H.3.2.1 Watershed Loading

OC Pesticides and PCBs loads from areas that do not drain to an MS4 system are also estimated from monitoring data collected from the lake sediments near drainage inputs (Appendix G, Monitoring Data) and simulated sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output and estimated sediment loading for areas that do not discharge to an MS4.

H.3.2.2 Atmospheric Deposition

The atmospheric deposition of OC Pesticides and PCBs on the watershed is accounted for in the annual runoff loads. The direct net deposition of OC Pesticides and PCBs (on the lake surfaces) is estimated to be minimal in comparison to the indirect loading. The surface area of each impaired lake is only a small portion of the total draining area for each lake. The lake surface area of Peck Road Park Lake represents only 0.37 percent of the total surface area draining to the lake. The atmospheric deposition from the remaining drainage area for the lake (99.63 percent) is accounted for in the annual runoff loads collected in the MS4 system. The area for direct deposition for Echo Park Lake and Puddingstone Reservoir is 1.8 percent and 3.1 percent of the total draining surface area (respectively). Moreover, research of OC Pesticides and PCBs exchange between waterbodies and atmosphere has demonstrated a recent shift in equilibriums that causes OC Pesticides and PCBs to be expelled from the lake and into the atmosphere under conditions of declining loads. The volatilization of OC Pesticides and PCBs may be greater than the direct atmospheric deposition into the lake (Manodori, et al., 2007; Thomann and Di Toro, 1983). Thus, direct deposition of OC Pesticides and PCBs to the lake surface is not evaluated as a loading source in these TMDLs.

H.3.2.3 OC Pesticides and PCBs Stored in Lake Sediment

Historical loading of OC Pesticides and PCBs has resulted in storage of these contaminants in lake sediment. In most cases, this legacy storage appears to be the major source of OC Pesticides and PCBs in the food chain. Benthic macroinvertebrates accumulate OC Pesticides and PCBs from the sediment and are consumed by sediment foraging fish, which in turn are consumed by higher trophic level fish, resulting in bioconcentration of OC Pesticides and PCBs.

The sediment stores of OC Pesticides and PCBs do not constitute an ongoing load and are thus not amenable to a traditional load allocation in mass per time units. Instead, a target concentration is assigned to achieve FCGs based on a BSAF analysis (see Section H.4.1).

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H.4 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of each impaired lake. The loading capacity is used to estimate the TMDL, and allocate that load to permitted, point sources (wasteload allocations) and nonpoint sources (load allocations). The TMDL also contains a Margin of Safety.

The OC Pesticides and PCBs TMDLs for the three lakes assess watershed loading into the lakes using monitoring data and sediment loads simulated by a previously developed LSPC model calibrated for the Los Angeles and San Gabriel river basins. The simulated sediment loads are based on the characteristics of the watershed land uses and incorporate dry, normal, and wet conditions for the Los Angeles area. The LSPC model is discussed in further detail in Appendix D (Wet Weather Loading) and by Tetra Tech (2004, 2005).

For many of the OC Pesticides and PCBs impairments, concentrations in water and sediment meet applicable criteria for those media, but concentrations in fish exceed FCGs. The CTR criteria for the protection of human health are designed to protect against elevated fish tissue concentrations due to bioaccumulation from the water column, but do not address bioaccumulation from the sediment. The consensus-based TEC targets are designed to protect against direct toxicity to benthic organisms, but explicitly do not consider food chain bioaccumulation. Therefore, a separate linkage analysis is needed to determine the sediment exposure concentration that will achieve FCGs.

Lake sediments are often the predominant source of OC pesticides and PCBs in the water column. The bottom sediment serves as a sink for organic compounds that can be recycled through the aquatic life cycle. OC Pesticides and PCBs have long half-lives in sediment and water and decay will not be a significant mechanism of reduction. Incoming loads of OC Pesticides and PCBs will mainly be adsorbed to particulates in stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

H.4.1 Representation of Bioaccumulation from Sediment

A linkage between the OC Pesticides and PCBs concentrations in sediment and the concentration in the impaired fish species is established using an empirical relationship based on a biota-sediment accumulation factor (BSAF).

Bioaccumulation of OC Pesticides and PCBs from contaminated sediment is described using biota-sediment accumulation factors (BSAFs). The BSAF describes the pollutant ratio between sediment and aquatic biota. The species and environmental factors are accounted for by normalizing the ratio to the fraction of organic carbon in sediments and fraction of lipids in the biota:

$$BSAF = \frac{C_{biota}/f_l}{C_{sed}/f_{OC}},$$

where C_{biota} is pollutant concentration in the benthic organism or benthic community, C_{sed} is the pollutant concentration in the sediment, f_l is the fraction of lipids in the biota, and f_{OC} is the fraction of organic carbon in the sediment.

Typical BSAF values are provided by Wong et al. (2001). Measurements of contaminants in sediment and fish from hundred of sites in the United States were compiled for data between 1992 and 1995 and analyzed by Wong et al. (2001). There were several different fish taxa included in the analysis; most (88 percent) of the samples were benthic species (carp, white sucker, channel catfish, etc.), but some pelagic fish were included (trout and bass) in the calculation. The BSAF values for the selected pollutants are shown in Table H-2.

Table H-2. Typical BSAF[1] Values

Pollutant	BSAF
Chlordane	2.9
DDT[2]	1.1
Dieldrin	3.4
Total PCBs	2.4

[1]Typical values from Wong et al. (2001).
[2]Based on o'p-DDT.

The BSAF can be used to determine the associated equilibrium sediment concentration, as shown below:

$$BSAF = \frac{C_{biota}/f_l}{C_{sed}/f_{OC}} \text{; therefore } C_{sed} = f_{OC} \cdot \frac{C_{biota}}{f_1} \div BSAF$$

The maximum allowable sediment concentration that can exist without causing impairment to the fish is determined using the FCGs for C_{biota}. The difference between the existing sediment concentration and the maximum allowable concentrations are compared to determine necessary reductions.

$$C_{biota} = FCG$$

$$C_{sed-target} = f_{OC} \cdot \frac{FCG}{f_1} \div BSAF$$

The allowable fraction of existing sediment concentration is then simply

$$\frac{C_{sed-target}}{C_{sed}} = \frac{FCG}{C_{biota}}$$

The sediment targets calculated from the BSAF analysis for each lake and the applicable OC Pesticides and PCBs are described in the lake-specific chapters.

H.4.2 Equilibrium Model for OC Pesticides and PCBs in Lakes

The linkage analysis also employs a model of in-lake processes, described in Butcher (1997), Chapra (1991), and Chapra and Reckhow (1983). In general, the steady-state model presented here uses the notation and solutions of the full steady-state model presented in Chapra and Reckhow (1983), which accounts for partitioning, losses, burial, and recycling from the sediment. This model idealizes the lake as three zones, representing the water column, mixed or active sediment layer, and deep sediment and derives mass balances for each layer. The equilibrium model can be used to determine the rate of external loading that would be required to account for current observed sediment concentrations under steady-state conditions. It can also be used to estimate concentrations in water and sediment when these are below analytical detection limits.

Chapra's steady-state solution for contaminant concentration in the mixed sediment layer, $c_{t,m}$(µg/m^3) is

$$c_{t,m} = \frac{F_{dw}}{F_{dp}} R_{df}\, c_{t,w},$$

where $c_{t,w}$(µg/m^3) is the steady-state concentration in the water column, F_{dw} is the dissolved fraction in the water column, F_{dp} is the ratio of sediment porewater pollutant concentration to the total concentration of contaminant in the sediment, and R_{df} is the diffusive feedback ratio, i.e., the ratio of contaminant concentration in porewater to that dissolved in the water column. These are defined as follows:

$$F_{dw} = \frac{1}{1 + K_{d,w}\, s_{t,w}}$$

$$F_{dp} = \frac{1}{\phi + (1-\phi)\rho_p\, K_{d,s}}$$

$$R_{df} = \frac{\phi\left(D_s / z'_b\right) + K_{d,w}\, s_{t,w}\, v_w \left(A_w / A_m\right)}{\phi\left(D_s / z'_b\right) + s_{t,w}\, v_w \left(A_w / A_m\right)\left[K_{d,s} + \phi / \left[(1-\phi)\rho_p\right]\right] + \left(k_m\, z_m / F_{dp}\right) - \phi D_s \lambda_2}$$

In these equations,

$K_{d,w}$ is the partition coefficient to solids in the water column (m^3/g),

$K_{d,s}$ is the partition coefficient to solids in the sediment (m^3/g),

$s_{t,w}$ is the solids concentration in the water column (g/m^3),

φ is the sediment porosity (unitless),

ρ_p is the density of solids (g/m^3),

D_s is the diffusion rate for the contaminant in porwater (m^2/yr),

z'_b is a thickness (m) defining the gradient between the mixed sediment layer and the overlying water – nominally the average of the mixed layer depth and the overlying laminar layer,

v_w is the settling velocity of solids (m/yr),

A_w is the water surface area (m^2),

A_m is the sediment surface area (m^2),

k_m is the first order decay rate for the contaminant in sediment (yr^{-1}), assumed equal for the mixed and deep sediment layers,

z_m is the sediment mixed layer depth (m), and

$$\lambda_2 = \frac{v_b}{2\phi D_s F_{dp}}\left[1 - \sqrt{1 + \frac{4\phi F_{dp} D_s k_m}{{v_b}^2}}\right],$$

with v_b being the resuspension velocity (m/yr), defined as

$$v_b = \frac{v_w A_w s_{t,w}}{A_m (1-\phi)\rho_p}.$$

The steady-state solution for the fully-mixed water column concentration is given by

$$c_{t,w} = \frac{W_c}{Q + k_w V_{t,w} + v_a A_w} ,$$

where

W_c is the mass loading rate of the contaminant (µg/yr),

Q is the outflow (m^3/yr),

k_w is the first-order decay coefficient in the water column (yr^{-1}),

$V_{t,w}$ is the volume of the water column (m^3), and

$$v_a = F_{pw} v_w + \phi \frac{D_s}{z'_b} \frac{A_m}{A_w} F_{dw} \left(1 - R_{df}\right) - v_r \frac{A_m}{A_w} \frac{F_{dw}}{F_{dp}} R_{df} .$$

Here, F_{pw} is the fraction of pollutant mass attached to particulate matter in the water column,

$$F_{pw} = \frac{K_{d,w} s_{t,w}}{1 + K_{d,w} s_{t,w}} ,$$

and v_r is the resuspension velocity (m/yr).

Chapra's steady-state toxicant formulation does not explicitly account for volatilization losses; however, these are readily included in the general water column decay coefficient (k_w, yr^{-1}) by inclusion of a term v_v/H, where v_v is a volatilization velocity (m/yr) and H is the average lake depth (m). Volatilization velocity may be estimated by the two-film method of Mackay (1981) as

$$\frac{1}{v_v} = \frac{1}{K_l} + \frac{1}{K_g H_e} ,$$

where K_l is the liquid side mass transfer coefficient (m/yr), K_g is the gas side mass transfer coefficient, and H_e is the dimensionless Henry's Law constant, a measure of volatility. Methods to approximate values of the transfer coefficients from wind speed (W, m/s) and molecular weight (M) are also given in Chapra and Reckhow (1983). In units of m/yr, these are:

$$K_l = 204.4 \frac{W^{3/2}}{\sqrt{M}} \text{ and } K_g = 43800 \frac{W}{\sqrt{M}} .$$

The rate loss of a toxicant in the waterbody depends on physical and chemical characteristics, such as volatility, degradability, and tendency to sorb to particulate matter. The chemical-specific parameters used for the simulations were selected from Brunner et al. (1990), Hansen et al. (1999), Leatherbarrow et al. (2006), Li et al. (1990), and Mackay et al. (1992). Henry's law coefficients are weighted for presence of individual congeners and a separate PCB coefficient was determined for each waterbody. These values are displayed in Table H-3.

Table H-3. Chemical-Specific Parameters for Simulation

Parameters for Model Input	Total Chlordane	Total DDTs	Dieldrin	Total PCBs
Molecular weight	409.6[e]	355[e]	381[e]	326[e]
Dimensionless Henry's Law Constant (H_e) at 20°C	1.18E-03 [c]	1.21E-04[c]	2.14E-04[c]	6.70E-03[a]
Partition coefficient (K_{OC}; l/kg)	38,000[d]	240,000[d]	12,000[d]	676,000[b]
Degradation rate in sediment (yr^{-1})	0.30[c]	0.08[c]	0.25[c]	2.99[e]
Degradation rate in water (yr^{-1})	0.73[c]	0.73[c]	0.84[c]	2.07[e]

Sources: (a) Brunner et al. (1990); (b) Hansen et al. (1999); (c) Leatherbarrow et al. (2006); (d) Li et al. (1990); (e) Mackay et al. (1992).

Loss rates are also dependent on the physical characteristics of the individual lakes; specifically surface area, volume, drainage area, annual runoff, and organic carbon fraction in the sediments. The lake-specific parameters were gathered mainly from SCAG 2005 land use data. The organic carbon fraction of the lake sediment was calculated for each lake using data collected by USEPA and the Regional Board during sampling events conducted in 2008 and 2009. For other lake characteristics (e.g., sediment solids density, settling velocity, resuspension velocity, active sediment thickness, and sediment porosity), the model uses typical or assumed values appropriate for these lakes, as reported by Chapra and Reckhow (1983). The presumed values are shown in Table H-4.

Table H-4. Assumed Parameters for Simulation

Parameters for Model Input	Assumed Value
Sediment solids density (g/cm^3)	1.38
Settling velocity (m/yr)	100
Resuspension velocity (m/yr)	0.007
Active sediment thickness (cm)	5.0
Sediment porosity (unitless)	0.8
Diffusion rate in sediment porewater (m^2/yr)	0.01

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H.5 TMDL Development

A TMDL is defined by the loading capacity. The loading capacity of a waterbody represents the maximum amount of pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). The OC Pesticides and PCBs TMDLs are calculated based on the maximum amount of organochlorine compound loading consistent with meeting the fish tissue goals.

H.5.1 Loading Capacity and Allocations

The loading capacity for each lake and the applicable OC Pesticides and PCBs are determined using the target sediment concentration. Estimates of the existing sediment load to each lake are discussed in Appendix D (Wet Weather Loading) and the individual lake chapters. The loading capacity is expressed as a concentration in micrograms per dry kilogram (µg /kg dry weight). The loading capacity can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

Because the loading capacity is presented as a concentration, the WLAs and LAs are also shown as a concentration for each jurisdiction and subwatershed. The watershed areas associated with permitted Municipal Separate Storm Sewer Systems (MS4s) are assigned wasteload allocations, which are further broken down by jurisdiction and subwatershed. In addition, general industrial and general construction stormwater permittees are also assigned wasteload allocations. Load allocations are assigned to areas not draining to an MS4. The specific allocations for each lake are described by jurisdiction and subwatershed in further detail in their respective chapters.

H.5.2 Margin of Safety

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

H.5.3 Daily Load Expression

Sediment contamination and resulting bioaccumulation is a long-term process and annual loading rates are the most appropriate measure for the TMDL. However, USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River TMDL. The TMDLs developed here each include a daily maximum load estimate consistent with the guidelines provided by USEPA (2007). Because the majority of external OC Pesticides and PCBs loads occur during wet weather events that deliver sediment to the lakes, the maximum allowable daily load is calculated from the 99th percentile flow multiplied by the sediment concentration target of the OC Pesticides and PCBs and the sediment event mean concentration (annual average sediment load divided by annual average flow from the watershed).

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