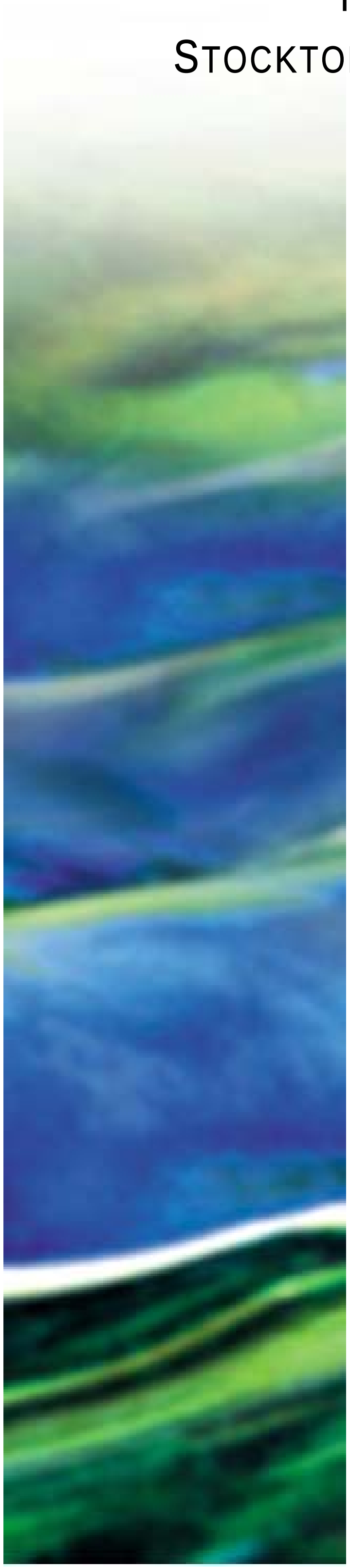
Evaluation of the Potential Effects of Nitrate Plus Nitrite Discharged from the Stockton Regional Wastewater Control Facility on the San Joaquin River in Support of Dilution Credit for NPDES Permitting



Prepared for:

Regional Water Quality Control Board
Central Valley Region

On Behalf of:

City of Stockton

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Solutions for Progress

July 2013

EVALUATION OF THE POTENTIAL EFFECTS OF NITRATE PLUS NITRITE DISCHARGED FROM THE STOCKTON REGIONAL WASTEWATER CONTROL FACILITY ON THE SAN JOAQUIN RIVER IN SUPPORT OF DILUTION CREDIT FOR NPDES PERMITTING

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EXECUTIVE SUMMARY

BACKGROUND

The City of Stockton (City) operates the Regional Wastewater Control Facility (RWCF), from which final effluent is discharged into the San Joaquin River, within the Sacramento–San Joaquin Delta (Delta). The RWCF effluent contains nutrients, including nitrate. The Central Valley Regional Water Quality Control Board (Central Valley Water Board) *Water Quality Control Plan for the Sacramento River and San Joaquin River Basins* (Basin Plan) applies California Department of Public Health (DPH) Maximum Contaminant Levels (MCLs) as water quality objectives for waters designated for Municipal and Domestic Supply (MUN). The MCL for nitrate plus nitrite is 10 milligrams per liter (as nitrogen) (mg/L-N). Beyond maintaining MUN water supply standards, Central Valley Water Board permitting staff have been concerned with the effects of elevated nitrate and nitrite discharges on biologically sensitive aquatic resources or critical habitats, development of objectionable bottom deposits, and other nuisance conditions that can be caused by elevated nutrient levels in riverine systems.

In its current National Pollutant Discharge Elimination System (NPDES) permit, the City has been granted dilution credit and an associated mixing zone for nitrate plus nitrite. As part of its upcoming NPDES permit renewal, the City is seeking dilution credit as a means of obtaining permitted effluent limitations greater than the MCL of 10 mg/L-N, in order to ensure future permit compliance. As part of this request, the City is requesting new boundaries for the RWCF mixing zone for nitrate plus nitrite; the mixing zone would extend 1.4 miles upstream and 1.7 miles downstream of the RWCF outfall (Robertson-Bryan, Inc. 2013). In granting dilution credits and associated mixing zones for both priority and non-priority pollutants, including nitrate plus nitrite, the Central Valley Water Board applies requirements specified in the State Water Resources Control Board (State Water Board) *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (commonly referred to as the Statewide Implementation Plan or SIP).

The purpose of this study was to determine whether current RWCF discharge levels of nitrate plus nitrite cause any of the 11 mixing zone requirements specified in the SIP to be exceeded. If so, the mixing zone requested may not be appropriate to grant. Conversely, if the requested dilution credit, and associated mixing zone, result in receiving water conditions that are consistent with all 11 requirements of the SIP (within and outside the mixing zone), then granting the requested dilution credit and associated mixing zone would be appropriate and protective of beneficial uses.

METHODOLOGY

Delta hydrology was modeled using Delta Simulation Model II (DSM2) to determine the proportion of water at drinking water intake locations that is constituted by RWCF effluent. This study also used modeling to make a detailed evaluation of RWCF effluent mixing and river velocities in the study reach of the San Joaquin River to explain, in part, algae community composition and structure within the study area.

Field monitoring also was conducted. The field study area for monitoring was a reach of the San Joaquin River extending from approximately 6 miles upstream of the RWCF outfall to approximately 2.25 miles downstream of the RWCF outfall. This area encompasses: (1) the existing mixing zone of the RWCF effluent and (2) an approximately 2.5-mile reach, upstream of the tidal movement zone, that is almost always unaffected by the RWCF discharge. This upstream reach is referred to herein as the *reference reach/sites*. Submerged and emergent vegetation and basic water quality parameters were monitored bimonthly at 11 monitoring sites, for a total of six monitoring events between March 2012 and January 2013. The benthic macroinvertebrate (BMI) community, algae community, and nitrate plus nitrite data were collected at seven monitoring sites on four occasions between March 2012 and November 2012.

KEY FINDINGS

Drinking Water Intakes

- Modeling results estimate that RWCF effluent makes up less than 1% of the water, on a long-term average, throughout the majority of the Delta channels and at drinking water intakes. In the south Delta, RWCF effluent makes up less than 2% of the water fraction on a long-term average basis at existing intakes, and less than 7% on a maximum daily basis. On a maximum daily basis, the incremental contribution of nitrate from the RWCF to south Delta drinking water intakes could be as high as approximately 2 mg/L-N, whereas on a long-term average basis, the maximum incremental contribution would be approximately 0.50 mg/L-N.
- None of the drinking water withdrawal locations show nitrate concentrations near or above the 10 mg/L-N drinking water MCL.
- Since nitrate concentrations at the south Delta pumping plants (i.e., Banks and Jones pumping plants) are typically already above 0.5 mg/L-N, it is not expected that the RWCF's incremental contribution of nitrate would cause algal blooms in State Water Project (SWP) or Central Valley Project (CVP) facilities downstream of the intakes, or result in taste or odor concerns for downstream water users, when they otherwise would not occur.

Algae Communities

- The scientific literature indicates that nutrient levels in Delta waters are sufficiently high that they do not control the growth of algae (i.e., nutrients are not a limiting factor in the growth of algae communities. In the Delta, light availability is the primary limiting factor for algae growth, although grazing (particularly by filter feeders) and hydraulic residence time limitations (how much time water resides at a given location due to river velocities) also play a role.
- The RWCF discharge and mixing zone does not stimulate phytoplankton growth relative to reference sites, even though nutrient concentrations are higher within the mixing zone compared to the reference sites. The biomass and density of all major algal divisions were lower within the mixing zone than at the reference sites, and showed a decline between the reference sites and the mixing zone.
- The decrease in total algal biomass from the reference sites to the mixing zone is mostly related to the decrease in diatoms, which typically make up most of the biomass. The decrease in diatom abundance has been documented in this reach by other researchers, and similar decreases have been documented in other estuaries. In the study reach, the phenomenon has been attributed to hydrodynamics (i.e., decreased velocities, leading to increased settling of diatoms), the increased depth of the nonphotic zone, and an increase in zooplankton grazing pressure with increasing distance downstream. River velocity decreases substantially from the reference sites to the mixing zone, due to both river cross-section changes and increasing tidal influence.
- The RWCF discharge does not adversely influence phytoplankton species composition. The proportions of total algal density made up by each algae division is virtually the same in the mixing zone as in the reference sites. As a fraction of the total biomass, cyanobacteria make up a larger fraction in the mixing zone than at the reference sites, while diatoms make up a smaller fraction in the mixing zone than at the reference sites. However, because the biomass of all divisions decreases in the mixing zone due to decreased river velocities, increased depth of the non-photic zone, and increased zooplankton grazing, this phenomenon is entirely driven by a larger decrease in diatom biomass in the mixing zone than the concurrent decrease in cyanobacteria biomass.
- Statistical analyses indicated that variability in algal biomass and density between sampling events was driven primarily by physical factors, specifically river velocity and temperature. Within each sampling event, variability in algal biomass and density between the reference sites and the mixing zone were driven primarily by river velocity.
- The density of potentially harmful algal species observed in samples was generally greater in the reference reach than in the mixing zone. The total abundance of cyanobacteria with the potential to produce toxins or taste/odor compounds did not exceed the World Health Organization (WHO) threshold of 20,000 cells per milliliter

(cells/mL) in any of the samples. With regard to *Microcystis aeruginosa* specifically, based on a review of the literature, there is very little evidence for a relationship between nitrate levels and *Microcystis* abundance. Also, there is no evidence to indicate that a modest increase in the nitrogen to phosphorus (N:P) ratio (in the range of 10:1 to 40:1) would have any significant effect on the abundance of *Microcystis* in the Delta.

Benthic Macroinvertebrate Communities

- If adverse effects on algae communities (in abundance or composition) were occurring in the study reach, it would be expected that the BMI community would show effects indicative of these changes. Based on this study, there is no evidence to suggest that the RWCF discharge has caused adverse changes in the BMI community within the study reach. Furthermore, based on the scientific literature, the BMI assemblage in the study reach is not characteristic of a community in a river system that has been degraded by elevated nutrient loads or eutrophication.

Submerged and Emergent Vegetation

- During most months, submerged and emergent vegetation covered less than 1% of the surface area at each sampling location. Vegetation was confined to within a few feet of shore in the shallow river margins, or to small patches of water hyacinth and Brazilian waterweed that uprooted and floated downstream in the mid-channel at the water surface.
- No trends in the abundance or density of growing submerged or emergent vegetation were observed in the study area that would indicate a response to elevated nutrient levels from the RWCF discharge.
- At no time or location were conditions observed in which the density or abundance of any submerged or emergent vegetation reached levels that would restrict the passage of aquatic life or be considered a nuisance.

A detailed assessment of each of the 11 SIP mixing zone requirements is included in Section 5. As described there, the proposed mixing zone meets all 11 SIP mixing zone requirements. Consequently, findings from analyzing information from the scientific literature coupled with the data collected by field monitoring under this study support the granting of the City's requested dilution credit and associated mixing zone for nitrate plus nitrite (see Section 5) that would be appropriate and protective of receiving water beneficial uses.

1 INTRODUCTION

1.1 BACKGROUND

The City of Stockton (City) operates the Regional Wastewater Control Facility (RWCF) under its current National Pollutant Discharge Elimination System (NPDES) permit (NPDES No. CA0079138, Order No. R5-2008-0154), adopted by the Central Valley Regional Water Quality Control Board (Central Valley Water Board) on October 23, 2008. The RWCF provides primary and secondary treatment of influent at the main plant on the east side of the San Joaquin River. Primary treatment consists of screening, grit removal, and primary sedimentation. High-rate trickling filters and secondary clarifiers make up the secondary treatment processes. The secondary effluent is then pumped under the river to the tertiary treatment facility. Tertiary treatment includes flow through facultative ponds, engineered wetlands, two nitrifying biotowers, dissolved air flotation, mixed-media filters, and chlorination/dechlorination facilities. Final effluent is discharged into the San Joaquin River, within the Sacramento–San Joaquin Delta (Delta). The permitted average dry-weather flow discharge rate is 55 million gallons per day (mgd).

The Delta supports numerous beneficial uses. These are: Municipal and Domestic Supply (MUN); Agricultural Supply (AGR); Industrial Service Supply (IND); Industrial Process Supply (PRO); Groundwater Recharge (GWR); Navigation (NAV); Water Contact Recreation (REC-1); Non-Contact Water Recreation (REC-2); Commercial and Sport Fishing (COMM); Warm Freshwater Habitat (WARM); Cold Freshwater Habitat (COLD); Wildlife Habitat (WILD); Preservation of Biological Habitats of Special Significance (BIOL); Rare, Threatened, or Endangered Species (RARE); Migration of Aquatic Organisms (MIGR); Spawning, Reproduction, and/or Early Development (SPWN); Shellfish Harvesting (SHELL); and Estuarine Habitat (EST).

The RWCF effluent contains nutrients, including nitrate. Nutrients play a complex role in water quality and the health of aquatic ecosystems. Aquatic life depends on the availability of nutrients; however, elevated concentrations of nutrients can cause eutrophication, in which high algal and bacterial growth and subsequent microbial respiration deplete oxygen, producing anoxic waters and sediments. The beneficial uses most directly affected by nutrient concentrations include those relevant to aquatic organisms (COLD, WARM, EST), drinking water supplies (MUN), and recreational activities (REC-1, REC-2), all of which can be indirectly affected by the nuisance eutrophication effects of nutrients.

Historical concentrations of nitrate in the San Joaquin River have commonly exceeded 10 mg/L-N in the vicinity of the RWCF outfall since completion of nitrification facilities in 2008. However, nitrite (NO_2^-) has been above the detection limit of 0.1 mg/L-N in the receiving water in only 11 of 974 measurements, dating back to the beginning of 2010. Even when nitrite is detected, concentrations of nitrate are generally at least an order of magnitude greater than nitrite concentrations. Nitrite persists in the environment only under reducing conditions not characteristic of the San Joaquin River. Therefore, throughout this report, frequent reference is made to "nitrate"—this term is intended to be synonymous with "nitrate plus nitrite."

1.2 Regulatory Policy and Guidance

Nitrate plus nitrite is regulated by the Central Valley Water Board via application of the California Department of Public Health (DPH) maximum contaminant level (MCL) of 10 milligrams per liter (as nitrogen) (mg/L-N). Beyond maintaining MUN water supply standards, Central Valley Water Board permitting staff have been concerned with the effects of elevated nitrate and nitrite discharges on biologically sensitive aquatic resources or critical habitats, development of objectionable bottom deposits, and other nuisance conditions that can be caused by elevated nutrient levels in riverine systems.

In its current NPDES permit, the City has been granted dilution credit and an associated mixing zone for nitrate plus nitrite. As part of its upcoming NPDES permit renewal, the City is seeking dilution credit and an associated mixing zone as a means of obtaining permitted effluent limitations greater than the MCL of 10 mg/L-N, in order to ensure future permit compliance.

In granting dilution credits and associated mixing zones, the Central Valley Water Board applies requirements specified in the State Water Resources Control Board (State Water Board) *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (commonly referred to as the Statewide Implementation Plan or SIP). Although the SIP addresses implementation of water quality standards for priority toxic pollutants, the Central Valley Water Board implements the SIP mixing zone provisions for non-priority pollutants as well, including nitrate plus nitrite. The SIP mixing zone requirements tier directly from U.S. Environmental Protection Agency (USEPA) guidance for mixing zones, contained in the *Technical Support Document for Water Quality-Based Toxics Control* (EPA/505/2-90-001, March 1991) (commonly referred to as TSD).

The SIP requires that a mixing zone be as small as practicable and that a mixing zone shall not:

1. compromise the integrity of the entire water body;
2. cause acutely toxic conditions to aquatic life passing through the mixing zone;
3. restrict the passage of aquatic life;
4. adversely impact biologically sensitive or critical habitats, including, but not limited to, habitat of species listed under federal or state endangered species laws;
5. produce undesirable or nuisance aquatic life;

6. result in floating debris, oil, or scum;
7. produce objectionable color, odor, taste, or turbidity;
8. cause objectionable bottom deposits;
9. cause nuisance;
10. dominate the receiving water body or overlap a mixing zone from different outfalls; or
11. be allowed at or near any drinking water intake.

1.3 PURPOSE AND OBJECTIVES OF THE NITRATE PLUS NITRITE STUDY

The purpose of this study was to determine whether current RWCF discharge levels of nitrate plus nitrite meet the 11 mixing zone requirements of the SIP. The eutrophication effects of current nitrate plus nitrite discharges were assessed by comparing physical, chemical, and biological conditions within and outside areas influenced by discharges. This study was performed in support of the City's request for maintaining dilution credit for nitrate plus nitrite upon renewal of its NPDES permit.

The specific objectives of the study were as follows:

1. Characterize and compare the nitrate plus nitrite concentrations in the RWCF discharge mixing zone and the upstream control reach, which is unaffected by the RWCF discharge. The control reach and its sampling sites are referred to herein as the *reference reach/sites.*
2. Characterize and compare the benthic macroinvertebrate (BMI) communities in the RWCF discharge mixing zone and the upstream reference reach.
3. Characterize and compare the algae, submerged vegetation, and emergent vegetation in the RWCF discharge mixing zone and the upstream reference reach.
4. Characterize and compare the chlorophyll *a* production in the RWCF discharge mixing zone and the upstream reference reach.
5. Determine whether measurable differences in the composition, structure, or abundance of BMI communities, algae communities, or submerged or emergent vegetation, or differences in chlorophyll production, occur in the RWCF mixing zone relative to the reference reach and, if so, determine whether the mixing zone contains nuisance conditions that can be attributed to the discharge of nitrate plus nitrite from the RWCF.
6. Determine the effect of RWCF discharge on drinking water intake nitrate levels when regulated to an effluent limitation of 10 mg/L-N, versus the seasonal effluent limitations of 26 mg/L-N for April through September and 30 mg/L-N for October through March, as proposed in the City's April 2013 Report of Waste Discharge (RWD).

7. Characterize, through desktop evaluations using available data, the eutrophication-related impacts of nitrate discharges from the RWCF on far-field areas, including State Water Project (SWP) reservoirs and canals south of the Delta.

1.4 Size and Location of Proposed Mixing Zone

In April 2013, the City submitted its RWD for the RWCF to the Central Valley Water Board (Robertson-Bryan, Inc. 2013). In the RWD, the City requested dilution credit as a means of obtaining NPDES-permit effluent limitations greater than DPH's MCL of 10 mg/L-N. As part of the dilution credit request, the City requested a mixing zone for nitrate plus nitrite. The requested zone would extend 1.4 miles upstream and 1.7 miles downstream of the RWCF outfall. This zone is smaller than the mixing zone granted in the current NPDES permit. The basis for the proposed distance is threefold: (1) complete lateral mixing is estimated to have occurred within this reach; (2) the distance corresponds to locations for which DSM2 modeling results of effluent dilution can be obtained; and (3) at these distances, the RWCF effluent has been diluted sufficiently to achieve minimum necessary dilution ratios (2.1:1 for April 1–September 30, and 2.5:1 for October 1–March 31) > 98% of the time upstream and > 95% of the time downstream. More detailed information on the technical support for the City's requests is available in the City's RWD for the RWCF (Robertson-Bryan, Inc. 2013).

2 ENVIRONMENTAL SETTING AND LITERATURE REVIEW

2.1 Environmental Setting

The San Joaquin River is approximately 330 miles long, and its watershed drains the southern part of the Central Valley. Predominant land uses in the watershed are agriculture, undeveloped land, and urban areas. The climate is characterized by cool, wet winters and warm, dry summers. Most precipitation falls between November and April, with little or no precipitation falling between May and October. Major reservoirs store winter runoff for release and use year-round. The hydrology of the river and its major tributaries is highly managed through dams, diversions, and artificial conveyances.

The river enters the south Delta at Vernalis, where tides begin to affect the flow (**Figure 1**). The Delta is a complex system of channels, sloughs, marshes, canals, and islands at the confluence of the Sacramento and San Joaquin Rivers. In addition to supporting local agriculture and recreation, the Delta is home to hundreds of aquatic and terrestrial species, some of which are threatened or endangered, as identified by the California and federal Endangered Species Acts. The Delta is also vital to California's water supply system, supplying water to more than 25 million people in the San Francisco Bay area, the Central Valley, and southern California.

The Delta is at the head of the San Francisco estuary, which extends down through Suisun and San Pablo bays to San Francisco Bay. The estuary exhibits marine dominance in central and southern San Francisco Bay, freshwater dominance in the Delta, and the greatest salinity variation in Suisun Bay. The northern part of Suisun Bay is bordered by Suisun Marsh, the largest contiguous wetland along the Pacific Coast of the United States. The estuary as a whole has been heavily modified through upper watershed hydraulic gold mining, channelization, introduced fish and copepod species, in-basin and out-of-basin diversions/exports, and construction of dams in the upper watershed. More than 95% of historical wetlands have been removed from the estuary through large-scale reclamation activities (Sommer et al. 2007). The flow regime is heavily managed to meet flow and water quality requirements mandated by state law, recent court orders, and federal, state, and local contracts.

Suisun Marsh is on the State Water Board's Clean Water Act section 303(d) list of impaired water bodies for nutrients. A total maximum daily load (TMDL) is currently being developed. However, there is considerable uncertainty regarding nutrients and what role they play in water quality impairment in Suisun Marsh (SFBRWQCB 2012). Local sources, including discharge from local duck club water management operations, may play a role. Also, as explained in more detail below, nutrients in the form of ammonia may be affecting algae bloom development in Suisun Bay, and water is exchanged between Suisun Bay and Suisun Marsh through tidal action. Therefore, the ongoing TMDL development for nutrients in Suisun Marsh is more related to ammonia levels in the Sacramento River, and to localized sources, than to nitrate sources from the Delta.

The recent decline in pelagic fishes in the Delta is referred to as the Pelagic Organism Decline (POD), and it refers generally to the decline, since approximately 2000, in indices representing the abundance of delta smelt, longfin smelt, striped bass, and threadfin shad. Multiple stressors may be leading to POD, including top-down effects (e.g., water diversion, predation), bottom-up effects (e.g., food availability and quality), and the effects of changes in physical and chemical fish habitat (e.g., water quality, contaminants, disease, toxic effects of toxic algal blooms) (Sommer et al. 2007).

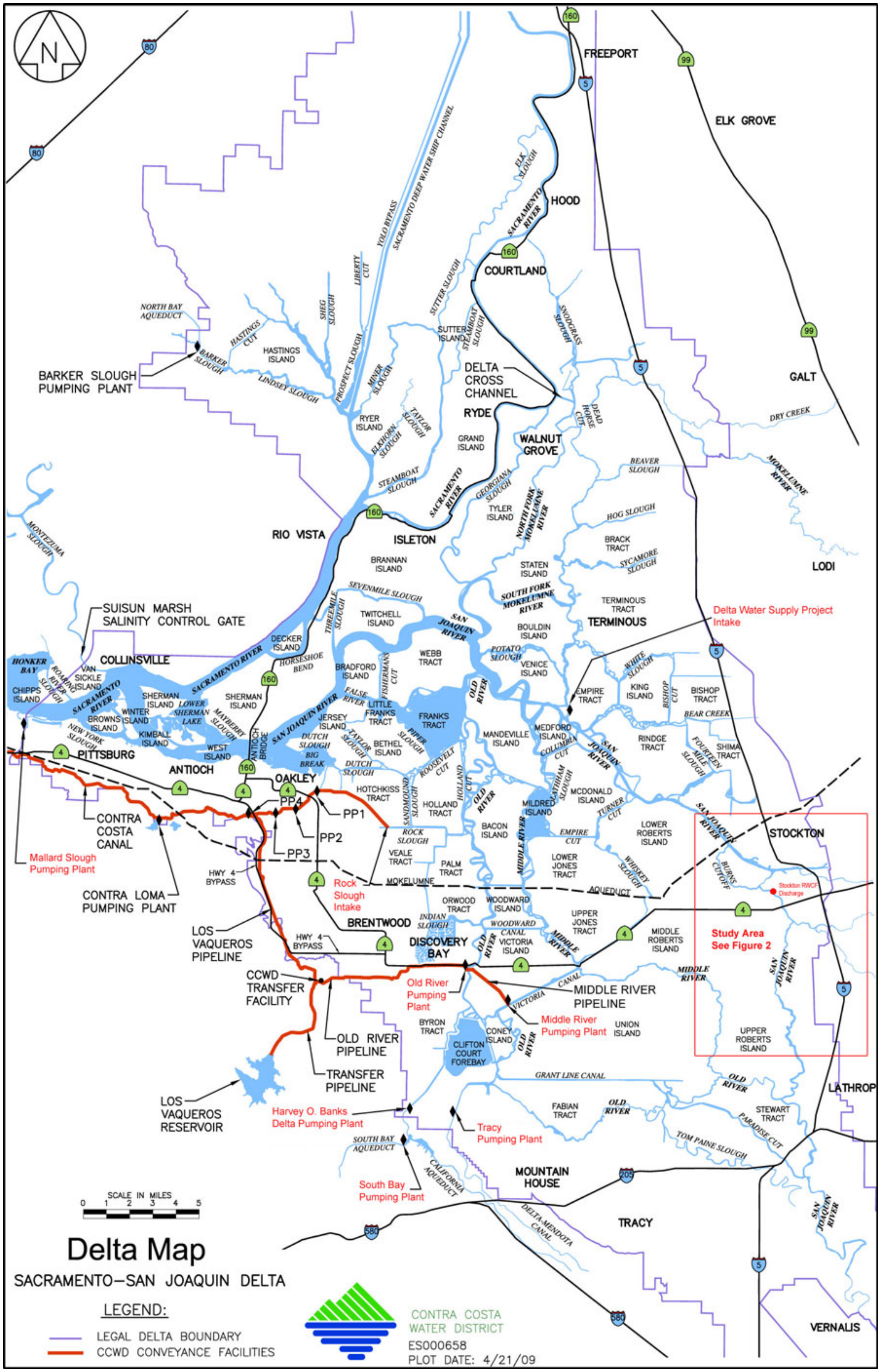


Figure 1. Field study area and drinking water intakes (in red) potentially influenced by the RWCF discharge.

2.2 NUTRIENTS AND PHYTOPLANKTON QUANTITY AND SPECIES COMPOSITION

As contributors to POD, both bottom-up effects and the stressors on physical and chemical fish habitat are related to primary productivity and phytoplankton; therefore, the role of nutrients in the Delta food web is important to the understanding POD. Further understanding of the role of nutrients would also inform potential management decisions to improve conditions.

Sources of nutrients in the Delta include the major tributaries (the Sacramento, San Joaquin, Mokelumne, Cosumnes, and Calaveras Rivers, as well as the Yolo Bypass), municipal discharges, and agricultural and urban runoff. Although nitrogen concentrations are generally higher in the San Joaquin River as it enters the Delta at Vernalis than in the Sacramento River as it enters the Delta at Freeport, nitrogen loads are higher in the Sacramento River due to the greater volume of flow. The loading of the San Joaquin River to the Delta may be responsible for 15–25% of the total annual nitrogen loading to the Delta, depending on water year type (Tetra Tech 2006).

Nutrient supply in the Delta is thought to be high enough that it does not control the growth of algae (i.e., nutrients are not limiting algae communities). Half-saturation constants (i.e., the nutrient concentration at which the growth rate is half of the maximum growth rate) for nutrient-limited phytoplankton growth are approximately 0.01 mg/L dissolved inorganic nitrogen and 0.003 mg/L soluble reactive phosphorus (Chapra 1997). Nutrient limitation of phytoplankton growth typically occurs only when nutrient concentrations fall below approximately 0.07 mg/L nitrogen and 0.03 mg/L phosphorus (Fisher et al. 1995). Historically in the Delta, there have been very few times when nutrients have fallen below either of these thresholds; therefore, nutrient limitation is considered extremely rare in the Delta (Jassby et al. 2002; Jassby 2005).

The ratio of nitrogen to phosphorus (N:P) in ambient water is often used to indicate which nutrient has the potential to be depleted or reach growth-rate-limiting concentrations first. Ambient N:P ratios only suggest potential N or P limitation because concentrations of both N and P may be so high that neither limits growth (Welch and Jacoby 2004), which appears to be the case in the Delta. Historically, it has been thought that, when this ratio is greater than 16:1 (by atoms), phosphorus is more limiting than nitrogen, and when it is less than 16:1, nitrogen is more limiting than phosphorus. However, this threshold is not rigid for all algal species and water bodies, and under nutrient-replete conditions may vary from 15:1 to 30:1 (N:P), depending on species composition (Geider and La Roche 2002). Thus, ambient N:P ratios (by atoms) less than 15 or greater than 30 suggest a potential for N or P limitation, respectively.

In the Delta, the availability of light is thought to be the primary limiting factor for algae growth, although grazing (particularly by filter feeders) and residence time limitations also play a role (Jassby et al. 2002). A study by Jassby et al. (2002) showed that primary productivity and chlorophyll *a* in the Delta declined significantly from 1975 to 1995, leading some to believe that food limitation may be playing a significant role in POD. Factors affecting the variability in the data set included the *Corbula amurensis* clam invasion, a long-term decrease in total suspended sediment (TSS) supply (which increased clarity), interannual variability in river flow (higher flows reduce residence time and contribute to higher TSS, reducing clarity and limiting growth), and a winter decline of unknown cause. However, data from 1995 to 2006 exhibited either a neutral or a positive trend in primary productivity and chlorophyll *a*, indicating that food limitation may not play as large a role in POD as previously thought (Jassby 2008). Phytoplankton biomass at Vernalis is highly correlated to discharge, and peak annual chlorophyll *a* values are determined by discharge rates in early summer. In fact, it appears as if low peak annual chlorophyll *a* values from 1977 through the early 1990s were entirely due to early summer discharge rates (Jassby 2005).

Regarding Delta algal species composition and food quality, the composition of the phytoplankton community over the last 40 years has generally shifted from diatoms toward green algae, cyanobacteria, and miscellaneous flagellate species. The abundance of diatoms decreased in the early 1980s, whereas dinoflagellates, cryptophytes, and chlorophytes were generally dominant from the late 1980s to mid-1990s. Cyanobacteria, including *Microcystis aeruginosa*, increased from the late 1990s to the mid-2000s. The changes in phytoplankton composition, and especially *Microcystis aeruginosa* blooms, have been implicated as possible factors in POD (Ballard et al. 2009). However, the reasons for the shifts in algal species composition are not clear.

The Delta's zooplankton community has changed over time as well. Rotifers, cladocerans, and copepods have experienced significant declines with little recovery. Two calanoid copepods, *Eurytemora affinis* and *Acartia* spp., once dominated the zooplankton community of the Low-Salinity Zone, but their abundance declined significantly in the late 1980s and early 1990s (Kimmerer 2004; Lehman 2004; Glibert et al. 2011). The decline of both has been attributed to increased grazing by the invasive clam *Corbula amurensis*, which became established in the Delta in the mid-1980s (Kimmerer 2004). *Pseudodiaptomus forbesi*, a calanoid copepod, has now become moderately abundant (Kimmerer 2004).

Researchers have recently investigated the role of elevated ammonia concentrations in limiting algae blooms in Suisun Bay and northern San Francisco Bay. The research has indicated that ammonia, while stimulating diatom growth at very low concentrations, also can inhibit uptake of nitrate in diatoms as concentrations increase above about 4 micromoles per liter (µmol/L) (0.056 mg/L-N) (Dugdale et al. 2007). This inhibition is of concern in Suisun Bay, where algal blooms may be prevented when conditions otherwise would be favorable (Wilkerson et al. 2006). Ammonia has thus been hypothesized to have contributed to the shift from a diatom-based community to one of smaller zooplankton in Suisun Bay.

Glibert (2010) analyzed more than 30 years of Delta water quality data, concluding that aquatic organism population shifts were associated with changes in the quality and quantity of nutrients, most specifically ammonia, discharged from the Sacramento Regional Wastewater Treatment Plant. Subsequently, others have criticized this work by demonstrating that the statistical techniques used were not appropriate and that the conclusions were therefore flawed (Cloern et al. 2011). Glibert and others agreed that the statistical conclusions of the 2010 review paper should be disregarded (Lancelot et al. 2012), but argued that some of their conclusions were nonetheless valid. It is not clear, absent statistical confirmation, that the relationships identified in this work are accurate and meaningful.

In the San Joaquin River, Lehman (2007) reported a shift from diatoms and green algae upstream of the RWCF to flagellates with increasing distance downstream. This shift was attributed mostly to hydrodynamics because the river in this reach shifts from a riverine to lake-like habitat in the Deep Water Ship Channel (DWSC). Turbulent mixing keeps heavy diatom cells suspended in the water column in shallow water habitats, but when velocity drops and mixing decreases, heavy diatom cells are lost to sedimentation (Reynolds 1994). Because diatoms are fairly large, the loss of diatoms resulted in an overall loss of phytoplankton biomass. This seaward decrease in diatom density had been reported in other estuaries as well, including the Westerschelde estuary, Elbe estuary, Schelde River estuary, and Parana River (Rijstenbil et al. 1993; Kies 1997; Muylaert et al. 2000; Izaguirre et al. 2001). At the time the Lehman 2007 data were collected, the RWCF was not nitrifying its effluent, so the nitrogen in the discharge comprised mostly ammonia, not nitrate. Following on Lehman's work, a three-year field study by Litton et al. (2008) attributed algal loss between Vernalis and the DWSC to light limitation due to the increased depth of the nonphotic zone and to an exponential increase in grazing pressure below the head of Old River. Net river flows less than 1,800 cubic feet per second (cfs) provided sufficient residence time for these mechanisms to contribute substantially to algal loss, which represented 20–80% of total algal biomass, depending on flow (Litton et al. 2008).

Generalizations regarding the trophic status of the San Joaquin River between Vernalis and the DWSC are complicated by the influence of flow on phytoplankton loss during the productive spring-fall period. **Table 1** shows the general boundary conditions in terms of algal metrics between oligotrophic-mesotrophic and between mesotrophic-eutrophic waterways (USEPA 2001). Oligotrophic refers to a waterway lacking in nutrients and primary productivity; mesotrophic, to a waterway containing moderate nutrients and primary productivity; and eutrophic, to a waterway containing abundant nutrients and primary productivity. Under high flows and above the head of Old River (where grazing pressure is low), chlorophyll *a* and algal abundance are typical of eutrophic waterways (Brunnell et al. 2008; Litton et al. 2008). However, chlorophyll *a* concentrations and algal abundance decrease in this section of the river during low flows and may, at times, be indicative of less productive, oligotrophic or mesotrophic waterways.

Table 1. Chlorophyll *a* concentrations that represent the boundary conditions for oligotrophic, mesotrophic, and eutrophic waterways.

Boundary	Chlorophyll *a* (µg/L)[a]
Oligotrophic-Mesotrophic	10
Mesotrophic-Eutrophic	30

[a] EPA 2001; µg/L = micrograms per liter.

2.3 Microcystis Aeruginosa

Microcystis aeruginosa (*Microcystis*) is a harmful cyanobacterial algal bloom species. In addition to producing surface scums that interfere with recreation and cause aesthetic problems, it also causes taste and odor in drinking water and produces toxic microcystins that are associated with liver cancer in humans and wildlife. *Microcystis* blooms can cause toxicity to phytoplankton, zooplankton, and fish, and also can affect feeding success or food quality for zooplankton and fish. Blooms of *Microcystis* require high levels of nitrate and phosphorus to develop, but also require high water temperature and long residence time, since the species is fairly slow growing (Lehman et al. 2008). In addition, low vertical mixing allows *Microcystis* colonies to float to the surface of the water column, where they outcompete other species for light.

In a study conducted in 2004 (Lehman et al. 2008), the San Joaquin River exhibited the highest *Microcystis* concentrations in the Delta, although levels throughout the Delta were rarely above the WHO threshold of 20,000 cells/mL. In the study, *Microcystis* occurred within a narrow range of environmental conditions, when water temperature was greater than 20°C, TSS was between 100 and 500 mg/L, specific conductance was between 100 and 300 microSiemens per centimeter µS/cm, ammonia was between 0.01 and 0.03 mg/L, and streamflow in the San Joaquin River was between 1,000 and 1,250 cfs. Nutrient concentration was not significantly related to variation in the *Microcystis* bloom sampled around the Delta. Although the high nutrient (nitrogen and phosphorus) concentrations found throughout the Delta were cited as prerequisites for the bloom, the persistence of the bloom was not related to nutrients because nutrient concentrations are much higher than limiting values throughout the Delta (Lehman et al. 2008). In a later study, *Microcystis* concentrations between sites in the Delta were positively correlated with nitrate-N, soluble phosphorus, and total nitrogen (total-N) (Lehman et al. 2010), and were also negatively correlated with chloride, TSS, and organic carbon. However, *Microcystis* levels were generally fairly low, and made up a substantial amount of the biomass at only one site, in Old River.

High residence time, low zooplankton grazing pressure, ample ammonia, limiting nitrate, and a low N:P ratio (< 15) have been associated with *Microcystis* bloom formation (Jacoby et al. 2000). At nutrient concentrations much lower than in the San Joaquin River, very high N:P ratios (upwards of 40) have also been shown to be conducive to blooms (Glibert et al. 2011). In any case, based on the information available, there is no evidence to indicate that a modest increase in the N:P ratio (in the range of 10:1 to 40:1) would have any significant effect on the abundance of *Microcystis* in the Delta.

2.4 NUTRIENT STOICHIOMETRY

As described above, several literature articles suggest that reducing ammonia concentrations in the Sacramento River and Suisun Bay would lead to positive effects on the ecosystem. Glibert et al.'s discussion of nutrient stoichiometry in the Delta (2011) is one of the only articles to argue that reductions in total-N, not just ammonia, could lead to positive impacts on the ecosystem. The article is based on the hypothesis that total N:P ratios control all aspects of ecosystem structure. Using historical data from Suisun Bay, which integrate nutrient concentrations and loading from both the Sacramento and San Joaquin Rivers, the authors show that the N:P ratio increased through the same period that saw many problematic ecosystem variable shifts (i.e., increases in flagellates, cyanobacteria, piscivorous fish, and invasive vegetation and bivalves; and declines in the zooplankton *Eurytomea* sp., delta smelt, and diatoms). The study showed many correlations between the N:P ratio and negative ecosystem responses. When viewed through a purely stoichiometric perspective, these correlations are argued to be causal relationships. For example, the authors argue that invasions were "set up" by altered stoichiometric ratios, while acknowledging that the initial introduction of invasives may have been a "stochastic" event (e.g., introduction via ballast water exchange). However, invasive species often proliferate and dominate ecosystems for reasons totally unrelated to nutrients, such as lack of endemic population controls like predation and competition. Further, because the historical N:P ratio in Suisun Bay is dominated by the concentration of ammonia arriving via the Sacramento River, most if not all of the correlations described in the study hold equally true for both the N:P ratio and ammonia. Since ammonia has been linked, through laboratory experiments and extensive study of the Sacramento River and Suisun Bay, to potential effects on algae and therefore on the ecology of Suisun Bay, it could be argued that a correlation between ammonia and negative effects represents a more compelling relationship.

Although not less credible than any other hypothesis, the N:P ratio hypothesis described in the Glibert et al. study remains a single hypothesis among many concerning POD in the Delta. It may be more appropriate to describe the N:P ratio (and nutrient stoichiometry generally) as one potential stressor—in the category of physical and chemical changes in fish habitat—in the POD multiple stressor theory. However, it is not known whether the current N:P ratio in the Delta is truly a stressor or not.

2.5 DRINKING WATER INTAKES CONSIDERATIONS

When waters of the Delta are exported into relatively shallow conveyance canals, algae may no longer be light limited, and growth of epibenthic algae and submerged aquatic vegetation may occur, in addition to phytoplankton growth. Thus, it is possible that increases in nutrient levels in Delta export waters may increase vegetative growth in the canals. Enhanced algal photosynthesis and growth may also result from the decrease in light attenuation that occurs when submerged aquatic vegetation filters suspended sediments and turbidity from waters conveyed through the Delta's canals. Dense stands of filamentous algal mats can obstruct water conveyance facilities and clog filters, and are cited as a primary contributor to taste and odor problems in Delta-based domestic water supplies (State Water Project Contractors Authority 2007; Janik and Losee, as cited in Lee 2008).

Delta-specific information on the relationship between benthic algae and nitrate levels, or any other nutrient, is sparse. However, the available research suggests that there is no obvious correlation between benthic algal abundance in the Delta and nutrient concentrations or fluxes (Hutton, as cited in Lee 2008). Complications in relating benthic algal abundance to nutrient levels are common (Royer et al. 2008; Hutton, as cited in Lee 2008), which is important in that benthic and attached algae are potentially more likely to affect taste and odor than is planktonic biomass generally (Juttner and Watson 2007; Taylor et al. 2006). A meta-analysis of studies covering 300 sampling periods from temperate streams, as well as benthic chlorophyll *a* and nutrient concentration data from a subset of 620 United States National Stream Water-Quality Monitoring Networks, was used to determine the relationship between nutrient concentrations (or ratios) and the biomass of benthic stream algae (Dodds et al. 2002, 2006). The comprehensive analysis showed that when total-N concentrations were greater than about 0.5 mg/L-N, there was no correlation between nutrient levels and benthic algal biomass. Total-N levels greater than this threshold make waterways nitrogen-replete, and light attenuation, substrate suitability, and temperature are then more likely to correlate with benthic algal abundance (von Schiller et al. 2007).

Total-N concentrations at the Banks Pumping Plant and in the SWP conveyance canals are, on average, higher than the 0.5 mg/L-N threshold determined by Dodds et al. (2006). Monthly average total-N concentrations at the Banks Pumping Plant ranged from 0.49 to 1.64 mg/L-N from 1999 to 2011 (California Department of Water Resources Water Data Library 2012). The lowest total-N concentrations at the Banks Pumping Plant are observed in August (0.16 to 0.73 mg/L-N; average of 0.49 mg/L-N). These data correspond well to those reported in the 2006 Sanitary Survey (State Water Project Contractors Authority 2007), where the average total-N concentration in August from 2001 to 2005 was 0.45 mg/L-N. The South Bay Aqueduct, California Aqueduct, and Delta Mendota Canal also have minimum summer total-N concentrations of about 0.5 mg/L-N (State Water Project Contractors Authority 2007). Because these canals are already nitrogen rich, other factors, such as temperature and increased water clarity, are more likely to affect the production of problematic algal mats than small increases in nitrate concentrations.

3 METHODOLOGY

3.1 San Joaquin River Flow and Tidal Hydrodynamic Assessment

The purpose of this section is to discuss the physical hydrodynamic setting of the San Joaquin River, and specifically the mixing zone, to contextualize sampling and monitoring results. The San Joaquin River enters the Delta at Vernalis, 31 miles upstream of the RWCF outfall (Figure 1). A substantial portion of the river, typically between 50% and 90% of the flow, is diverted into Old River, approximately 12.25 miles upstream of the RWCF outfall. Agricultural diversions are also located between Vernalis and the RWCF outfall. The river flow fluctuates tidally in the vicinity of the RWCF outfall. River flow gauges are maintained by the U.S. Geological Survey (USGS) at Vernalis and on the San Joaquin River at Garwood Bridge, 0.5 mile upstream of the outfall.

Mixing of the effluent with the river (i.e., dilution) is primarily a function of net river flow near the discharge. The tide moves upstream during flood-tide, and downstream during ebb-tide. Some volume of water may move past the discharge several times, resulting in multiple "doses" of effluent. The number of times that a tidal volume of water passes the RWCF is lower during high river flows, and higher during low river flows. The fraction of river water that is RWCF effluent generally increases from zero at the upstream extent of the mixing zone to its maximum at the downstream extent.

The tidal pattern for each lunar day (24.8 hours) consists of two high tides of unequal magnitude and two low tides of unequal magnitude. The lowest low tides and the highest high tides occur during the new moon and full moon each month (spring tides), whereas the tides during the neap-tide period are smaller.

In 2005, Jones & Stokes conducted a study and wrote a report indicating that the maximum upstream tidal movement was 3.75 miles from the RWCF outfall, and the maximum downstream tidal movement was 2.25 miles from the RWCF outfall, into the DWSC. However, the actual size of the mixing zone and the fully mixed river concentration depend on the net river flow. **Figure 2** shows a map of the San Joaquin River and mixing zone as defined in the Jones & Stokes report. In the present study, estimated effluent percentage at the time of sampling was estimated using modeling (see the next section). Flow data from the USGS gauge at the Garwood Bridge (0.5 mile upstream of the discharge) were also reviewed to determine measured tidal hydrodynamics around the time of sampling.

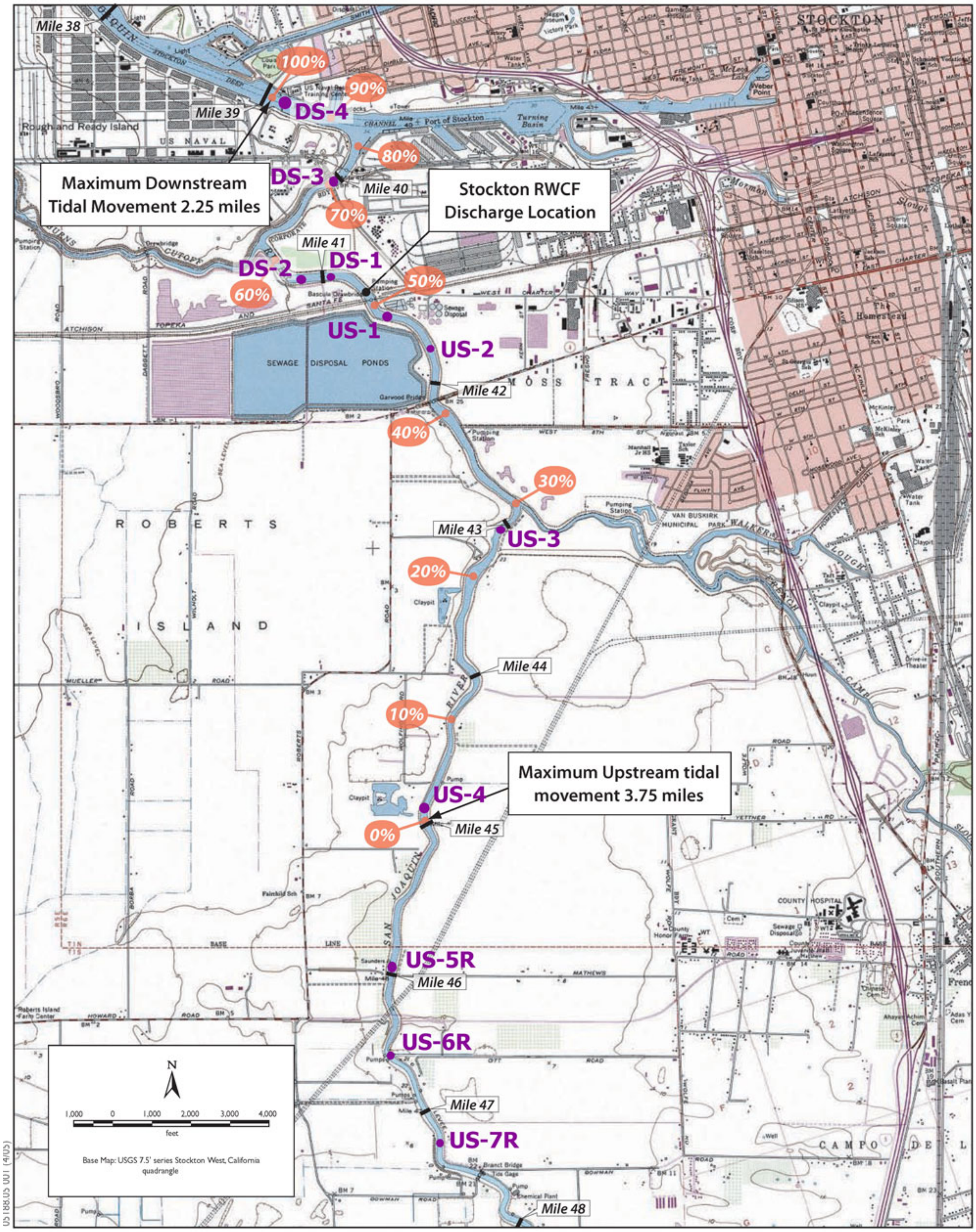


Figure 2. Locations of the sampling reaches and sampling sites used for monitoring BMI, plankton, and water quality. Percentages represent the portion of RWCF effluent that is mixed in the tidal mixing volume at low flows in the San Joaquin River from 0% of fully mixed river concentration at the upstream end to about 50% of the fully mixed river concentration at the RWCF outfall (Source: Jones & Stokes 2005).

3.2 MODELING OF NEAR AND FAR-FIELD CONDITIONS USING DSM2

The purpose of modeling near-field conditions was to provide an understanding of how much RWCF effluent (and, therefore, what incremental contribution of nitrate) was present at various near-field sites over the course of the sampling period. The purpose of modeling far-field conditions was to assess the incremental contribution of nitrate to drinking water intakes and Suisun Marsh.

The Delta Simulation Model II (DSM2), developed by the Delta Modeling Section of the California Department of Water Resources (DWR), is a one-dimensional computer model for simulating hydrodynamics, water quality, and particle transport in the Delta. A model grid representing the network of Delta channels was developed by DWR to cover major Delta channels, the Sacramento River to Sacramento, and the San Joaquin River to Vernalis. DSM2 was calibrated and validated in 1997 by DWR and in 2000 by a group of agencies, water users, and stakeholders. In 2009, DSM2 was calibrated and validated again to account for morphological changes, such as the flooded Liberty Island, and bathymetry, hydrodynamic, and water quality data collected after the 2000 calibration. DSM2 has been used frequently by DWR, other agencies, and stakeholders to simulate the potential impacts of Delta-related projects.

For this study, DSM2 was first used to simulate the transport and mixing of the RWCF effluent at its permitted capacity of 55 mgd in Delta channels for water years 1991–2012. Data output for locations of drinking water withdrawals was evaluated to identify the fraction of RWCF effluent at those locations. DSM2 was also run using actual RWCF discharge rates for 2012, to simulate the mixing of the effluent in the reach that was evaluated in detail in this study.

Table 2 shows the Sacramento and San Joaquin Valley Water Year Hydrologic Classifications for the modeled period.

Boundary river flow, stage data, and exports used as inputs to the DSM2 model were taken from the historical simulation template distributed with DSM2 version 8. Also used were more recent data supplied by the DWR Delta Modeling Support group, data downloaded from the California Data Exchange Center (CDEC), and data obtained from DWR's DAYFLOW data sets. In general, all data are approximately the same as would be obtained from the California Data Exchange Center (CDEC) measured data, except that data missing from the CDEC record were replaced with estimates for those periods. All of the data used for boundary river flow and stage data were checked against data sets from CDEC. **Table 3** summarizes the data from CDEC stations used for the DSM2 simulation.

Table 2. Hydrologic water year classifications for modeled years.

	Water Year Hydrologic Classification	
Water Year	Sacramento Valley	San Joaquin Valley
1991	Critical	Critical
1992	Critical	Critical
1993	Above Normal	Wet
1994	Critical	Critical
1995	Wet	Wet
1996	Wet	Wet
1997	Wet	Wet
1998	Wet	Wet
1999	Wet	Above Normal
2000	Above Normal	Above Normal
2001	Dry	Dry
2002	Dry	Dry
2003	Above Normal	Below Normal
2004	Below Normal	Dry
2005	Above Normal	Wet
2006	Wet	Wet
2007	Dry	Critical
2008	Critical	Critical
2009	Dry	Below Normal
2010	Below Normal	Above Normal
2011	Wet	Wet
2012	Above Normal	Dry
% Wet	32%	36%
% Above Normal	23%	14%
% Below Normal	9%	9%
% Dry	18%	18%
% Critical	18%	23%

Data concerning the operation of the Delta Cross Channel Gates, installation and removal of the temporary barriers in the south Delta, and operation of the Clifton Court Intakes were also provided by the historical simulation template distributed with DSM2 version 8. These were supplemented by more recent data supplied by the DWR Delta Modeling Support group, data from the gate operation log published by the Central Valley Operations Office of the U.S. Bureau of Reclamation, Mid-Pacific Region, and the schedule listed on DWR's website.

Table 3. Boundary conditions, exports, and gate inputs to DSM2, along with data sources used to supplement data supplied with DSM2.

DSM2 Input	CDEC Station ID	DSM2 Abbreviation
Sacramento River flow	FPT	RSAC155
San Joaquin River flow	VNS	RSAN112
Yolo Bypass flow	YBY	BYOLO040
Cosumnes River flow	MHB	RCSM075
Mokelumne River flow	CMN	RMKL070
Calaveras River flow	NHG	RCAL009
Stage at Martinez	MRZ	RSAC054
CVP export	TRP	CHDMC004
SWP export	HRO	CHSWP003
CCWD Old River export	IDB	ROLD034
CCWD Rock Slough export	INB	CHCCC006
North Bay Aqueduct export	BKS	SLBAR002
CCWD Victoria Canal export	CCW	CHVCT000
Grantline Canal Barrier	[BDO]	GL_CN
Middle River Barrier	[BDO]	MID_R
Old River Barrier	[BDO]	OLD_R
Head of Old River Barrier	[BDO]	ORHRB/ORHRB_FALL
Delta Cross Channel	[USBR MP CVO]	RSAC128
Clifton Court Intakes	[DWR DMS]	CHWST000

Modeling output was used to estimate nitrate concentrations. These estimates were compared to nitrate concentrations determined analytically using samples. The estimates were also used to develop a continuous time series of effluent fraction and estimated nitrate concentrations in the river for use in the analysis. To model concentrations of nitrate in the mixed condition, the analysis incorporated data on the relative proportions of San Joaquin River water, east-side tributaries water (primarily the Calaveras River), agricultural return, and RWCF effluent, along with estimates of the nitrate concentrations associated with these sources (**Table 4**).

Table 4. Source water concentrations of nitrate (mg/L-N) used in the modeling of nitrate concentrations in the San Joaquin River in the vicinity of the RWCF discharge.

Date	Effluent [a]	San Joaquin River [b]	East Side Tributaries [c]	Agricultural Return Drains [d]
3/28/2012	20	1.7	0.17	3
5/23/2012	14	0.79	0.17	1.7
7/31/2012	11	2.1	0.17	1.4
11/27/2012	21	2.5	0.17	0.5

[a] Values taken from nearest weekly effluent nitrate sample results.
[b] Values detected at US-7R (the most upstream reference site).
[c] Average of data from the Mokelumne and Cosumnes River USGS gauges. It is assumed that nitrate in the Calaveras River is similar to these other two east-side tributaries, based on similar watershed land uses and geology. Data are derived from 45 data points pooled from both rivers, from 1961 to 1993.
[d] Estimated qualitatively based on agricultural drain data contained in the DWR Water Data Library (1990-2001).

3.3 FIELD MONITORING AND DATA COLLECTION

The following sections outline the field monitoring and data collection efforts conducted as part of this study. Data quality assurance and quality control procedures are discussed in Appendix A.

3.3.1 FIELD STUDY AREA AND MONITORING SITES

The field study area was a reach of the San Joaquin River extending from approximately 6 miles upstream of the RWCF outfall to approximately 2.25 miles downstream of the outfall (Figure 2). This area encompasses: (1) the existing mixing zone of the RWCF effluent and (2) an approximately 2.5-mile reach, upstream of the tidal movement zone, that is mostly unaffected by the RWCF discharge. The initial jet mixing zone extends less than approximately 125 feet (ft) upstream and 125 ft downstream of the RWCF outfall; in this area, complete lateral mixing has not occurred (Jones & Stokes 2005). Because the effluent has not distributed laterally within the initial jet mixing zone, this study did not assess conditions within this zone (i.e., 125 ft upstream and downstream of the RWCF discharge), but examined conditions in the existing mixing zone upstream and downstream of the initial jet mixing zone, as described in **Table 5** and Figure 2. Locations of the sampling reaches and sampling sites used for monitoring BMI, plankton, and water quality. Percentages represent the portion of RWCF effluent that is mixed in the tidal mixing volume at low flows in the San Joaquin River from 0% of fully mixed river concentration at the upstream end to about 50% of the fully mixed river concentration at the RWCF outfall (Source: Jones & Stokes 2005).. As mentioned in Section 0, the new proposed mixing zone is smaller than the existing zone; it extends 1.4 miles upstream and 1.7 miles downstream of the outfall. Because this study was designed based on the existing mixing zone, the term "mixing zone" used throughout this report refers to the existing mixing zone. Since the proposed mixing zone is contained within the existing mixing zone, all conclusions made in this report regarding the existing mixing zone extend to the proposed mixing zone. Note that site names were changed from the original work plan and lab reports, as described in **Table 6**.

Table 5. Description of the field study mixing zone and reference sites.

Zone	Description
Existing Mixing Zone	Approximately 125 ft to 2.25 miles downstream, and approximately 125 ft to 3.75 miles upstream of the RWCF outfall, where complete lateral mixing of effluent has occurred; the limits represent the downstream and upstream extent of tidal movement of the RWCF effluent (Jones & Stokes 2005).
Reference Sites	Reference sites located upstream of the influence of the RWCF effluent (Jones & Stokes 2005). Reference sites/reach are 4.75 miles to 6.25 miles upstream of the RWCF outfall.

Table 6. Approximate river mile (RM) location and parameters monitored at each sampling site.

Work Plan/Lab Report Site Name	Final Report Site Name	Approximate Location (RM) [a]	Parameters Monitored				
			BMI [b]	Algae [b]	Nitrate Plus Nitrite [b]	Submerged and Emergent Vegetation	Temperature, DO, pH, EC, Depth, and Clarity
2B-2	DS-4	39.0				X	X
2B-1	DS-3	40.0	X	X	X	X	X
1B-2	DS-2	40.75				X	X
1B-1	DS-1	41.0	X	X	X	X	X
1A-1	US-1	41.5	X	X	X	X	X
1A-2	US-2	41.75				X	X
2A-2	US-3	43.0				X	X
2A-1	US-4	45.0	X	X	X	X	X
3A	US-5R	46.0	X	X	X	X	X
3B	US-6R	46.5	X	X	X	X	X
3C	US-7R	47.5	X	X	X	X	X

X: Sites in which the parameter was collected or monitored. Shaded sites are within the proposed mixing zone (approx. RM 39.5-42.6)
DO: Dissolved oxygen
EC: Electrical conductivity
[a] RWCF outfall is located at approximately RM 41.25.
[b] Parameter was not collected during every sampling event.

3.3.2 SAMPLING SCHEDULE

Field data collection occurred bimonthly over a one-year period. Submerged and emergent vegetation, temperature, dissolved oxygen (DO), electrical conductivity (EC), water depth, and water clarity were monitored bimonthly at all 11 monitoring sites, for a total of six monitoring events (**Table 7**). BMI community, algae community, and nitrate plus nitrite data were collected four times at sites DS-3, DS-1, US-1, US-4, US-5R, US-6R, and US-7R over the course of the study (Table 7). During January and September, only submerged and emergent vegetation, temperature, DO, EC, water depth, and water clarity were monitored at these six sites.

Table 7. Schedule for collection of algae, BMI, submerged and emergent vegetation, and water quality data from the San Joaquin River.

Sampling Date	Algae [a]	BMI [a]	Nitrate Plus Nitrite [a]	Submerged and Emergent Vegetation	Temperature, DO, pH, EC, Depth, and Clarity
3/28/2012	X	X	X	X	X
5/23/2012	X	X	X	X	X
7/31/2012	X	X	X	X	X
9/26/2012				X	X
11/27/2012	X	X	X	X	X
1/28/2013				X	X

X: dates the parameter was collected or monitored.
[a] Parameters monitored at sites DS-3, DS-1, US-1, US-4, US-5R, US-6R, and US-7R only.

3.3.3 WATER QUALITY

Water temperature, DO, pH, and EC were measured at each sampling site using a handheld meter (YSI or HydroLab) and recorded on waterproof data sheets. Water depth was measured using a boat-mounted depth meter. Water clarity was measured using a standard Secchi disk. Water samples were collected to conduct analyses of nitrate plus nitrite, total-N, total phosphorus (total-P), and dissolved ortho-phosphate; these samples were collected by dipping a bottle supplied by the analytical laboratory into the river and allowing it to fill. Latex or nitrile gloves were worn by the sampling personnel to minimize the potential for contamination of the sample. The sample bottles were stored and preserved according to the laboratory's guidelines and delivered under chain of custody to the laboratory within the laboratory-specified hold times following sampling.

Following all field sampling events, all water quality data were entered into a database for analysis. In addition, San Joaquin River data collected in the last three years by the City, under its required monitoring program, were obtained and entered into the database for further analysis and comparison with study data. These data and other parameters were used (and compared as necessary to values in the scientific literature) to determine whether the RWCF discharge correlates with indications of eutrophication in the San Joaquin River mixing zone. Appendix G describes the statistical methodology used to assess changes in water quality parameters between the reference and mixing zone sites.

3.3.4 BENTHIC MACROINVERTEBRATE COMMUNITY

BMI sampling was conducted at three locations along a transect established perpendicular to the channel and flow. The sampling was performed using a hand-operated dredge (petite Ponar ® dredge) deployed from a boat. The petite Ponar sampler is a stainless steel dredge-type sampler that has a scoop volume of approximately 2,400 mL and a surface sampling area of 6 x 6 inches. This dredge—which is designed specifically to be used in substrate conditions ranging from soft sediments to firm, hard bottoms composed of sand, gravel, consolidated marl, or soft clay—is used widely for sampling sediments of Delta waterways. Samples were collected at the following three points along each transect:

1. one-quarter of the transect width,
2. the midpoint (i.e., half of the width) of the transect, and
3. three-quarters of the transect width.

The three samples collected at each transect were combined to form one composite sample representing each transect. The composite sample material was placed into a sample container and preserved with 95% ethanol solution.

The contents of each sample bottle were poured into a No. 35 (0.5-millimeter-mesh size) standard testing sieve and gently rinsed with fresh water to remove fine sediments. All large debris (e.g., wood, leaves, rocks/gravel) was removed from the sieve and discarded. The remaining sample contents were transferred to a sorting tray, from which 500 (±5%) organisms were randomly selected. The subsampled organisms were identified to the standard taxonomic level provided in the *CAMLnet List of Californian Macroinvertebrate Taxa and Standard Taxonomic Effort* (Harrington 2003), using a standard taxonomic key (e.g., Merritt and Cummins 1996a; Pennak 1989).

A taxonomic list of BMIs identified in each of the samples was created using Microsoft® Excel. The metrics listed in **Table 8** were calculated for each composite sample. To determine the effect of the effluent discharge on the BMI community of the San Joaquin River, spatial and temporal trends in BMI community structure and function were quantitatively analyzed through the following means:

- long-term changes in BMI community metrics at each transect were examined,

- statistical comparisons of BMI metrics at each transect within the mixing zone upstream and downstream of the RWCF outfall were conducted,
- statistical comparisons of each affected transect to those of the upstream reference sites were conducted, and
- pair-wise similarity indices (e.g., Sϕrenson's) of samples on both temporal (i.e., intratransect) and spatial (i.e., intertransect) scales were calculated and analyzed.

Of particular interest were the functional feeding groups (FFGs). Specifically notable were the portion of FFGs at each sampling site that are classified as filterers, which filter food (including phytoplankton) from the water column, and scrapers, which scrape food (including periphyton) from substrates (Table 8).

Table 8. Biological metrics for BMI communities and the expected response to water quality degradation.

Biological Metrics	Description	Response to Degradation
Richness Measures		
Taxonomic	Total number of distinct taxa.	Decrease
EPT	Number of taxa in the orders Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly).	Decrease
Ephemeroptera	Number of mayfly taxa (genus or species).	Decrease
Plecoptera	Number of stonefly taxa (genus or species).	Decrease
Trichoptera	Number of caddisfly taxa (genus or species).	Decrease
Coleoptera	Number of beetle taxa (genus or species).	Decrease
Predator	Number of taxa that prey on living organisms.	Decrease
Taxonomic Composition Measures		
EPT Index (%)	Percent composition of mayfly, stonefly, and caddisfly larvae.	Decrease
Sensitive EPT Index (%)	Percent composition of mayfly, stonefly, and caddisfly larvae with Tolerance Values of 0 through 3.	Decrease
Shannon Diversity Index	General measure of sample diversity that incorporates richness and evenness (Shannon and Weaver 1963).	Decrease
Dominant Taxa (%)	Percent composition of the single most abundant taxon.	Increase
Non-Insect Taxa (%)	Percentage of taxa that are not in the Class Insecta.	Variable
Tolerance/Intolerance Measures		
Tolerance Value	Value between 0 and 10, weighted for abundance of individuals designated as pollution tolerant (higher values) and intolerant (lower values).	Increase
Intolerant Organisms (%)	Percent of organisms in sample that are highly intolerant to impairment, as indicated by a tolerance value of 0, 1, or 2 .	Decrease
Intolerant Taxa (%)	Percentage of taxa that are highly intolerant to water and/or habitat impairment, as indicated by Tolerance Values of 0, 1, or 2.	Decrease
Tolerant Organisms (%)	Percentage of organisms in sample that are highly tolerant to	Increase

Biological Metrics	Description	Response to Degradation
	impairment, as indicated by a tolerance value of 8, 9, or 10,	
Tolerant Taxa (%)	Percentage of taxa that are highly tolerant to water and/or habitat impairment, as indicated by Tolerance Values of 8, 9, or 10.	Increase
Functional Feeding Groups		
Collector-Gatherers (%)	Percentage of macroinvertebrates that collect or gather fine particulate matter	Increase
Filterers (%)	Percentage of macroinvertebrates that filter fine particulate matter	Increase
Scrapers (%)	Percentage of macroinvertebrates that graze upon periphyton	Variable
Predators (%)	Percentage of macroinvertebrates that feed on other organisms	Variable
Shredders (%)	Percentage of macroinvertebrates that shred coarse particulate matter	Decrease
Other (%)	Percentage of macroinvertebrates that occupy an FFG not described above	Variable
Other		
Abundance	Estimated number of BMIs in a sample based on the proportion of BMIs subsampled; characterized by the number of organisms per square foot and per square meter	Increase

3.3.5 SUBMERGED AND EMERGENT VEGETATION

Submerged aquatic and emergent vegetation was assessed using visual observations to make qualitative and semiquantitative characterizations. At each vegetation monitoring site, the area within 100 ft upstream and downstream of the site was assessed visually for the presence of submerged and emergent aquatic vegetation. Field crew members wore polarized glasses to maximize their ability to see below the water surface. The area was photographed looking upstream, downstream, and along river left and river right banks, covering the area extending 100 ft upstream and 100 ft downstream of the monitoring site. To make a semiquantitative assessment of vegetation, crew members estimated the portion of the river channel occupied by submerged or emergent vegetation within 100 ft upstream and downstream of each sampling site.

To assess whether submerged and emergent vegetation was indicative of eutrophication caused by the RWCF discharge, the mixing zone sites were compared to the reference sites using site-specific data and qualitative evaluation of the types and areas of vegetative cover. This information was used to determine whether trends in vegetative coverage were present and, if they were, to assess whether vegetative cover was higher in the mixing zone. These summary characterizations were used to determine, based on best professional judgment and available scientific literature, if the area of vegetative coverage was substantial enough to restrict the movement of aquatic life, adversely affect habitat for special-status species, produce undesirable or nuisance species, or otherwise cause a nuisance.

3.3.6 ALGAE COMMUNITIES

Algae was collected at each sampling site using standard grab samples. Field staff collected samples by submerging a capped 1-liter amber bottle to a depth of 0.5 meter, removing the lid, allowing the bottle to fill to approximately 3–5 centimeters (cm) from the bottle mouth (to allow headspace for preservative), and replacing the cap while still submerged. Clean latex or nitrile gloves were worn by field staff collecting the samples at each site to prevent cross-contamination of samples. Upon collection, samples were preserved in the field as follows.

1. 1.0 mL of 25% aqueous general grade glutaraldehyde was added to each 100 mL of algal sample to be preserved. The final concentration of glutaraldehyde in the sample was 0.25–0.50%.
2. Lids were securely tightened onto the sample bottle, and bottles were shaken vigorously five times.
3. Samples were uncapped and the lids were smelled. If the glutaraldehyde was evident in the sample, bottles were tightly closed and the sample was considered preserved. If the glutaraldehyde could not be detected in the sample, additional glutaraldehyde was added (in maximum increments of 0.25 mL of glutaraldehyde per 100 mL of sample) to the sample bottle.

Upon sample preservation, the sample bottles were protected against breakage (by wrapping in bubble wrap and/or Styrofoam), stored on ice in a dark cooler, and covered with aluminum foil. Upon completion of each sampling event, algal samples were shipped under chain of custody via overnight delivery to the analytical lab.

Algae samples were analyzed by identifying phytoplankton to genus, enumerating, and calculating biovolume. Identification, enumeration, and calculation of biovolume were conducted under microscope by an expert in phycology using the appropriate standard analytical method. The method applied was based on whether the sample composition was: (1) dominated by soft algae greater than 10–20 microns (µm) in greatest axial length dimension (GALD), (2) dominated by soft algae less than 10–20 µm in GALD or by fragile, difficult to identify taxa, or (3) was dominated by diatoms. The magnification used depended on the size of the dominant taxa and the size and number of particulates. The goal was to count at multiple magnifications in order to correctly enumerate and identify taxa present that may vary by several orders of magnitude in size. If the sample was dominated by cells below 10–20 µm, or if the cells were fragile and difficult to identify, the majority of counting was completed at 400x–1,000x. Quantifying biovolume included measuring the GALD and additional measurements including length, width, and depth of different aspects of the colony or cell, which were approximated to a geometric figure and or figures) and making the appropriate calculations.

Chlorophyll *a* was not analyzed in samples collected as part of this study. Chlorophyll *a* data were obtained from the City, which collected data as part of its monitoring and reporting program, and from gauge data obtained from CDEC.

The statistical methodology used to assess changes in the algae community between the reference and mixing zone sites is described in Appendix G.

4 RESULTS AND DISCUSSION

4.1 SAN JOAQUIN RIVER FLOW AND TIDAL HYDRODYNAMIC ASSESSMENT

Figure 3 shows the measured range and daily average flow in the San Joaquin River at the USGS gauge at Garwood Bridge for 2012, along with the tidally filtered daily flow and DSM2 modeled daily average flow. Daily average flow was between 0 and 1,000 cfs for January through March, between 1,000 and 3,500 cfs in April and May, between 0 and 1,000 cfs in June, approximately 0 for July and August, between 0 and 1,000 cfs in September and October, and approximately 0 in November, before increasing due to high runoff in mid- to late December. Periods of approximately zero daily average flow coincide with periods during which nearly all of the flow in the San Joaquin River is diverted into Old River 12.25 miles upstream of the RWCF outfall. Reverse flows due to tidal action occurred in every month.

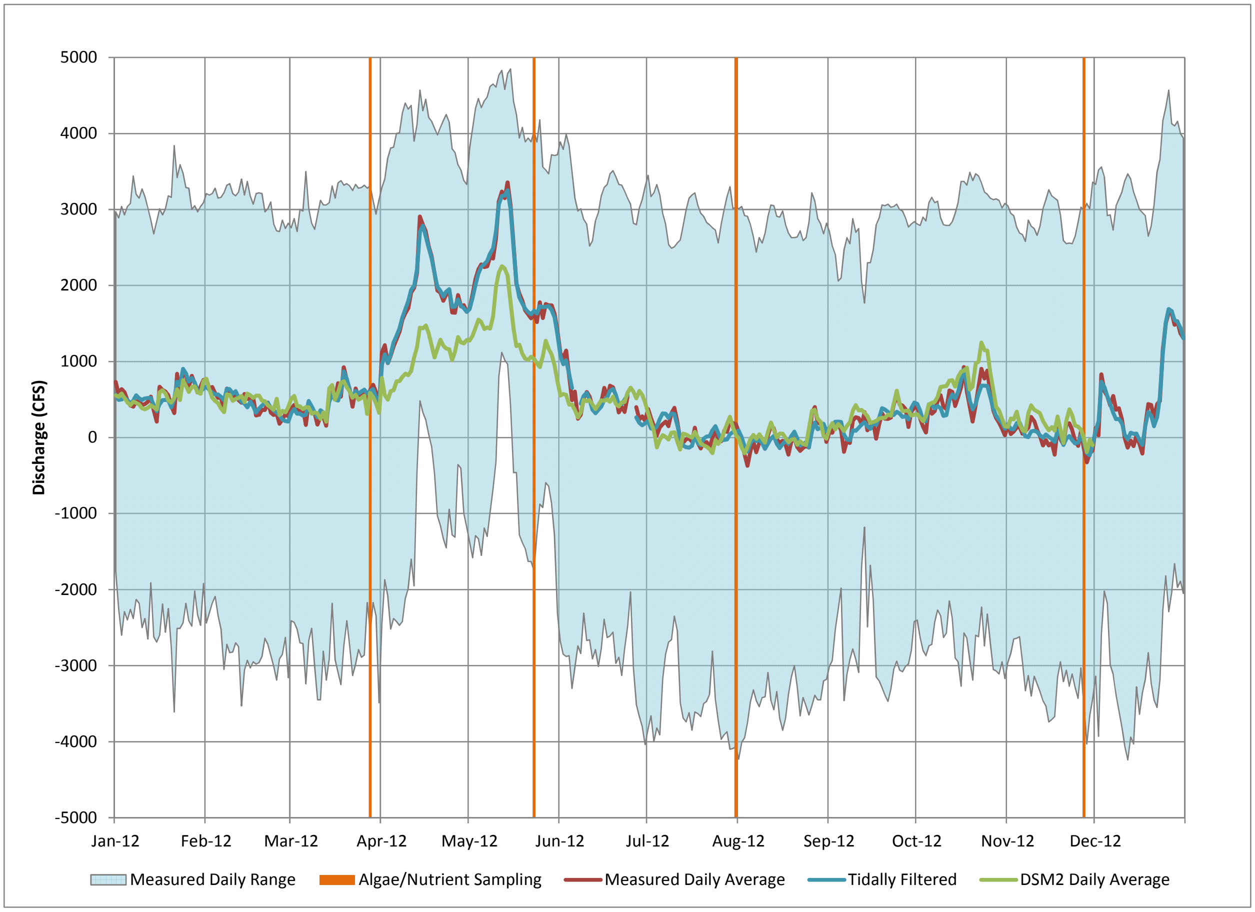


Figure 3. Measured and modeled flow in the San Joaquin River at the USGS Garwood Bridge gauge for 2012.

4.2 Modeling of Near and Far-Field Conditions Using DSM2

4.2.1 Near-Field Conditions

This section discusses the results of DSM2 simulations for 2012 using actual RWCF discharge rates.

Figure 4 shows the monthly average modeled river velocity at all sample sites in the San Joaquin River for March, May, July, and November (the months in which algae and nutrient samples were taken). It is evident from the figure that river velocity decreases substantially from the reference sites US-7R, US-6R, and US-5R downstream to the furthest downstream site, DS-4. DS-4 is located in the DWSC, where the cross-sectional area of flow is nearly five times the cross-sectional area of flow of the river upstream (Jones & Stokes 2005); hence, the velocity drops dramatically at that site. However, it is clear that the velocity decreases even upstream of that point, due to both cross-sectional changes and increasing tidal influence.

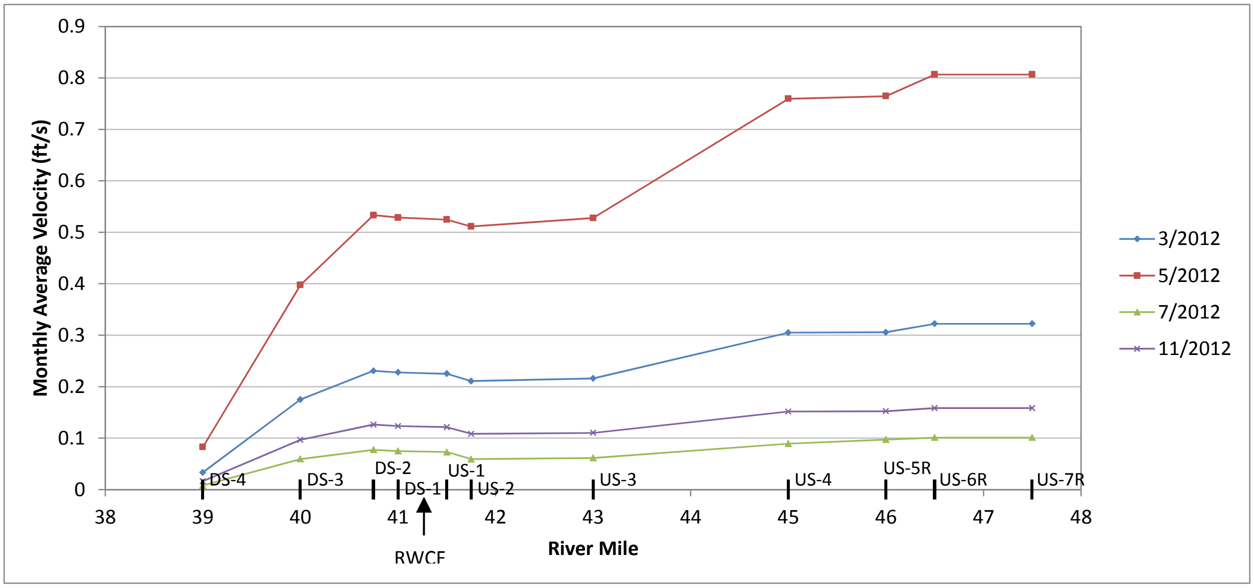


Figure 4. Monthly average modeled river velocity in the San Joaquin River at all study sample sites for March, May, July, and November (months of algae/nutrient sampling).

In Appendix B, Figure B-5 shows the source water fractions for January through November 2012 for all 11 sample sites. The figure shows that, for times when there was net downstream flow in the San Joaquin River in the vicinity of the RWCF (i.e., January through June, September, and October), the RWCF effluent fraction was generally low (0–10%) both upstream and downstream of the outfall. However, during July and August particularly, and somewhat in November, there was very little net downstream flow; therefore, the percent of the water made up of RWCF effluent increased substantially at all downstream sites with multiple tidal dosing. A similar increase was seen at upstream sites, with the effect decreasing with increased distance upstream. In July and August, agricultural return water showed a similar dramatic increase, although its effect was more or less constant across sample sites.

Figures B1 through B4 in Appendix B show the source water fractions on a 15-minute basis for March 28, May 23, July 31, and November 27, respectively (i.e., the dates of algae and nutrient sampling). On March 28 and May 23, there was little variability, and effluent was evident only at sites DS-4 upstream through US-3. In July, the large increase in effluent and agricultural return water is shown, and in November, just an increase in effluent. For the July and November sampling events, sites DS-4 upstream through US-2 did not show much variability throughout the day, while sites US-3 and US-4 showed variability with the tidal flow.

For July and November, the model result also indicate the potential influence of the effluent at reference sites US-5R, US-6R, and US-7R. These sites were specifically chosen to be outside the mixing zone defined in the 2005 Jones & Stokes report. However, it does not appear as if such a prolonged period of essentially zero flow in the vicinity of the RWCF had occurred in the data set examined by Jones & Stokes. Thus, although the reference sites were not influenced by

RWCF effluent for the majority of the study, it is possible, based on the modeling results, that there was influence during the July sampling event.

4.2.2 FAR-FIELD CONDITIONS

Table 9 summarizes the modeled long-term average and maximum daily RWCF effluent concentrations at locations of major intakes in the Delta. The table also shows the incremental amount of nitrate that would be contributed from the RWCF to each drinking water intake under conditions where the RWCF is regulated to: (1) a 10 mg/L-N nitrate effluent limitation and (2) the nitrate effluent limitations of 26 mg/L-N for April through September and 30 mg/L-N for October through March, as proposed in the City's RWD for NPDES permit renewal. It should be noted that the model was run using the maximum permitted average dry weather flow, so results are not indicative of historical conditions.

The Contra Costa Water District Victoria Canal Intake showed the greatest average (1.77%) and maximum daily (6.64%) effluent concentrations. Banks Pumping Plant (represented by Clifton Court Forebay) and Jones Pumping Plant (represented by Delta Mendota Canal) had the next highest average and maximum daily effluent concentrations. Figures in Appendix C show the modeled concentration of RWCF effluent at each of the major intake locations on a monthly average basis for each water year type and on a long-term monthly average basis, as well as time series of the concentrations for the modeled period. For Victoria Canal, Clifton Court Forebay, and Delta Mendota Canal, the highest concentrations tend to occur in critical water years.

On a maximum daily basis, the modeled incremental contribution at Victoria Canal was as high as approximately 2 mg/L-N, while at the Banks and Jones Pumping Plants, it was as high as approximately 1 mg/L-N. On a long-term average basis, the incremental contribution at Victoria Canal was 0.50 mg/L-N, while at the Banks and Jones Pumping Plants, it was 0.29 and 0.20 mg/L-N, respectively. None of the locations showed nitrate concentrations near or above the 10 mg/L-N drinking water MCL; thus, the incremental contribution of nitrate under either effluent limitation scenario would not cause or contribute to exceedance of the MCL. Furthermore, given the information discussed in the literature review section, since nitrate concentrations at the Banks and Jones pumping plants are generally well above 0.5 mg/L-N, it is unlikely that incremental contributions of nitrate under either effluent limitation scenario would cause algal blooms in SWP or CVP facilities downstream of the intakes, or result in undesirable tastes and odors for downstream water users, when they otherwise would not occur.

Table 9. RWCF effluent concentrations (%) at Delta water intakes, and incremental nitrate contributions under 10 mg/L-N and 26/30 mg/L-N effluent limitation scenarios.

Location	DSM2 Location Code	RWCF Effluent Concentration (%)		Incremental Nitrate Contribution at 10 mg/L-N Effluent Concentration		Incremental Nitrate Contribution at 26/30 mg/L-N Effluent Concentration	
		Long-Term Average	Maximum Daily	Long-Term Average	Maximum Daily	Long-Term Average	Maximum Daily
Barker Slough / North Bay Aqueduct	SLBAR002	0.00	0.00	0.00	0.00	0.00	0.00
Sacramento River at Mallard Island (MAL)	RSAC075	0.04	0.40	0.00	0.04	0.01	0.10
San Joaquin River at Antioch	RSAN007	0.08	0.69	0.01	0.07	0.02	0.18
San Joaquin River at Venice Island	RSAN043	0.20	2.49	0.02	0.25	0.06	0.65
CCPP Rock Slough Intake	SLRCK005	0.22	1.94	0.02	0.19	0.06	0.51
Victoria Canal (CCWD AIP)	CHVCT000	1.77	6.64	0.18	0.66	0.50	1.99
Clifton Court Forebay (Banks Pumping Plant)	CLIFTON_ COURT	1.02	3.40	0.10	0.34	0.29	1.02
Delta Mendota Canal (Jones Pumping Plant)	CHDMC004	0.73	3.36	0.07	0.34	0.20	0.91

CCPP: Contra Costa Pumping Plant
CCWD AIP: Contra Costa Water District Alternative Intake Project

4.3 Field Monitoring and Data Collection

4.3.1 Water Quality

Secchi Depth, Turbidity, DO, Water Temperature, and EC

Physical and chemical water quality parameters measured as part of this study are presented in **Figure 5**. Water temperatures throughout the mixing zone and reference reach were uniform for each sampling event. Water temperatures were the highest in July and lowest in January. Because of instrumentation failures, DO measurements were not collected in May and July, but measurements from the other sampling events yielded DO concentrations typically between 8 and 10 mg/L. EC was variable among sampling events, ranging from 400 to 1,000 µS/cm. During the months of March, July, and September, EC decreased in the direction of the RWCF outfall in the upstream reach, and increased in the downstream section with increasing distance downstream.

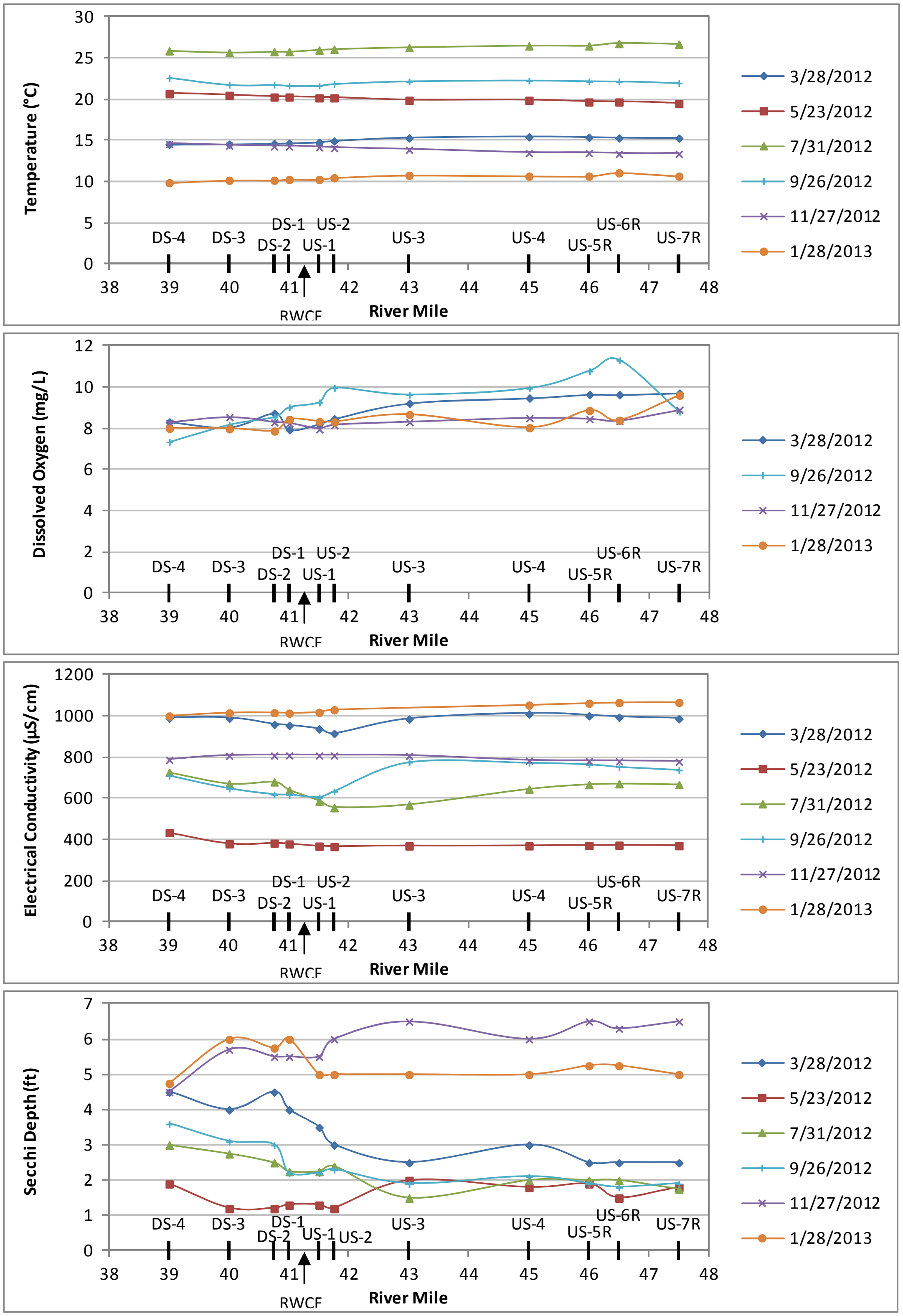


Figure 5. Temperature, dissolved oxygen, electrical conductivity, and Secchi depth, measured upstream and downstream of the RWCF outfall in the San Joaquin River from March 2012 through January 2013.

Secchi depth, a measure of water clarity, showed month-to-month and within-reach variability. November and January yielded Secchi depths much greater than other months, likely related both to low primary productivity because of low water temperature, as well as to the low suspended sediment load that is typically associated with low river flow. Likewise, high river flow and high suspended sediment resulted in the low Secchi depths measured uniformly throughout the river in May. Low water velocity near and within the DWSC (Figure 4), which allows for settling of suspended particles, also led to greater Secchi depth downstream of the RWCF (compared to the upstream sections) in March, July, and September.

Turbidity measurements are made by RWCF staff for the receiving water monitoring program. Turbidity data from 2010 through 2012 show that upstream of the DWSC, river water is more turbid (Figure D-5 in Appendix D).

Nutrients

Nitrate represented the majority of total-N measured during the study, which accounts for the similarity in trends and correlation between total-N and nitrate (**Figure 6**; Table G-1 in Appendix G). At the reference sites, nitrate and total-N concentrations were greatest in November, followed by July, March, and May. This trend was inversely related to river discharge, as flow in the San Joaquin River was greatest in May, followed by March, July, and November (Figure 3). The relationship between flow and nitrate is also evident in the San Joaquin River at Vernalis, upstream of the reference sites. At Vernalis, nitrate concentrations are generally greater than 1 mg/L-N under baseflow conditions, and nitrate concentrations are relatively lower when flows are elevated due to increased runoff or reservoir releases (**Figure 7**). Even though high river flows dilute nitrate concentrations within both the reference and mixing zone reaches, overall, nitrate and total-N concentrations were significantly greater in the mixing zone compared to the reference reach (**Table 10**). However, at no time were nitrate concentrations in excess of the public health MCL of 10 mg/L-N. In comparison to nitrate, total Kjeldahl nitrogen (TKN) concentrations were not significantly different between the mixing zone and reference reach (Table 10).

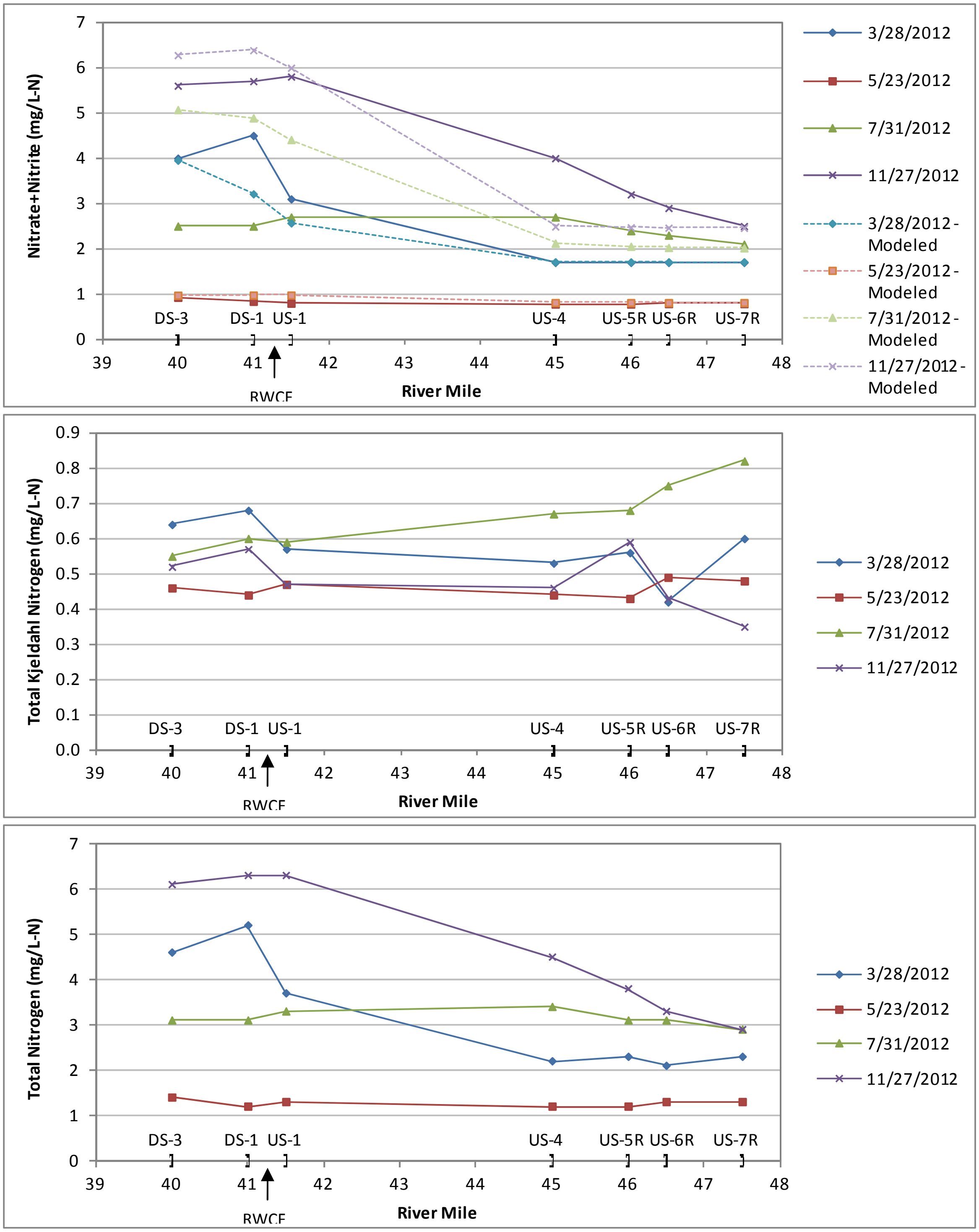


Figure 6. Nitrate plus nitrite, TKN, and total-N concentrations measured in water samples taken upstream and downstream of the RWCF outfall in the San Joaquin River from March 2012 through November 2012.

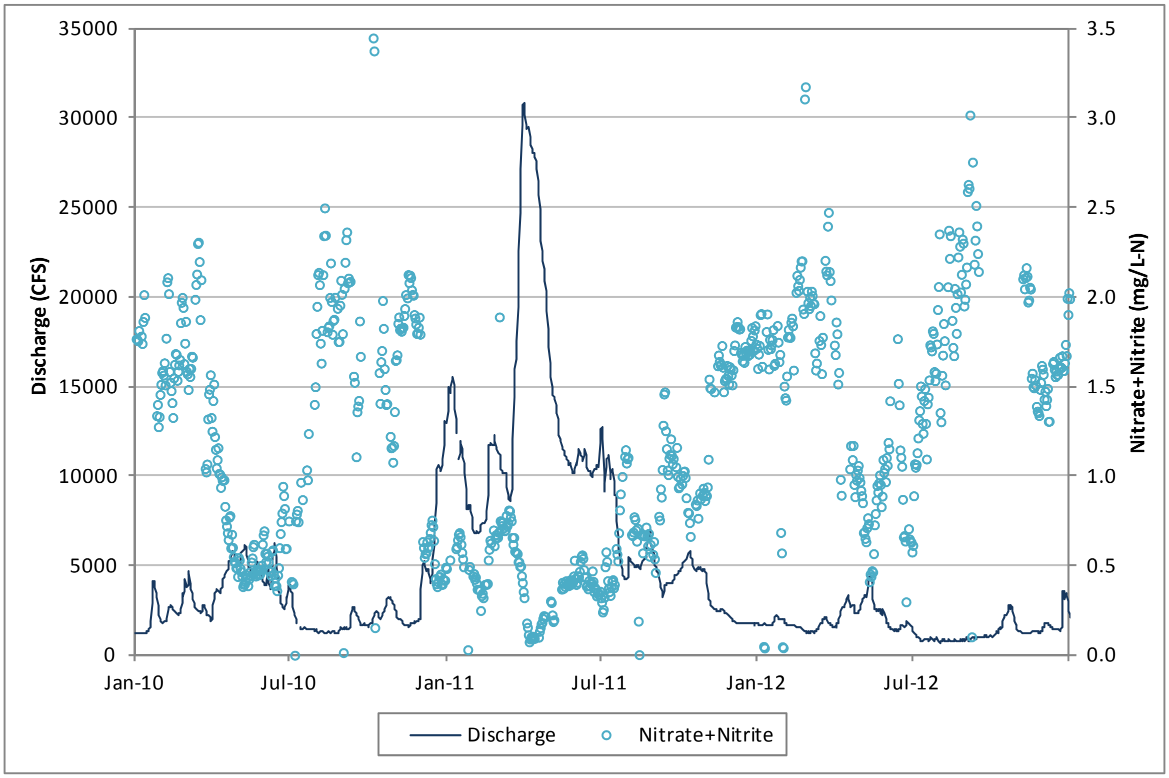


Figure 7. Nitrate plus nitrite concentrations in the San Joaquin River and river discharge at Vernalis from January 2010 through December 2012 (data from CDEC).

Table 10. Least-squares mean values of nitrate plus nitrite, TKN, total-N, dissolved phosphate, and total-P concentrations, and N:P, measured in water samples taken from the San Joaquin River in the mixing zone and reference reaches during the study.

	Means [a]		P-values [b]	
	Reference Reach	Mixing Zone Reach	Site US-4 Included	Site US-4 Excluded
Nitrate+Nitrite (mg/L-N)	1.90	3.25	<0.0001	0.0002
TKN (mg/L-N)	0.55	0.55	0.9232	0.9247
Total-N (mg/L-N)	2.47	3.80	0.0002	0.0005
Dissolved Phosphate (mg/L-P)	0.14	0.27	<0.0001	<0.0001
Total-P (mg/L-P)	0.24	0.37	<0.0001	<0.0001
N:P (atoms N/atoms P)	23.3	23.6	0.7146	0.8269

[a] Least-square means were calculated excluding site US-4. See Appendix G for more details.

[b] P-values were generated from a general linear model analysis, and comparison was made between the results obtained when site US-4 was included or excluded from the analysis.

The influence of flow as a primary driver of the variability in nitrate concentrations between the reference reach and the mixing zone was investigated with hydrodynamic modeling. DSM2 modeling confirmed that most of the variability in nitrate concentrations in the reference reach and mixing zone was attributable to river flow/dilution and vicinity to the RWCF outfall ("modeled" series in Figure 6). Measured nitrate concentrations in March, May, and November agreed well with the modeling assessment. During sampling in May, high river flows were sufficient to dilute nitrate and total-N from the RWCF discharge such that relatively low, uniform concentrations were measured and modeled among all sampling sites. Less dilution was available in March and November, which led to higher nitrate concentrations near the RWCF outfall. Concentrations measured in the vicinity of the RWCF outfall in July were markedly lower than those projected by the model. It is unknown whether in-river nitrate processing (uptake by algae, or denitrification) was responsible for the low nitrate concentrations measured in the vicinity of the RWCF outfall in July. Nonetheless, samples taken independently of the study on July 31, 2012, as part of City's routine receiving water monitoring yielded nitrate concentrations similar to those of samples collected for this study in the vicinity of the discharge (2.1 mg/L-N; Table D-2 in Appendix D).

Total-P and dissolved phosphate concentrations measured during the study showed similar trends to nitrate and total-N (**Figure 8**), and were significantly greater in the mixing zone than in the reference reach (Table 10). In contrast to the nitrate trend, dissolved phosphate and total-P concentrations were much greater near the RWCF outfall than at the reference sites in July.

The total-N and total-P concentrations were used to calculate the ratio of N:P (by atoms) for each sampling event, the results of which are shown in **Figure 9**. During the study, N:P (by atoms) measured in the San Joaquin River ranged from approximately 13:1 to 40:1 in the zone affected by the RWCF discharge, and from approximately 16:1 to 28:1 at the reference sites. N:P ratios in the zone affected by the RWCF discharge suggest that, were nutrients limiting in the system, nitrogen limitation would occur in July and phosphorus limitation would occur in March; however, at no time during the study were total-N or total-P concentrations below the critical thresholds that would actually be expected to limit phytoplankton growth.

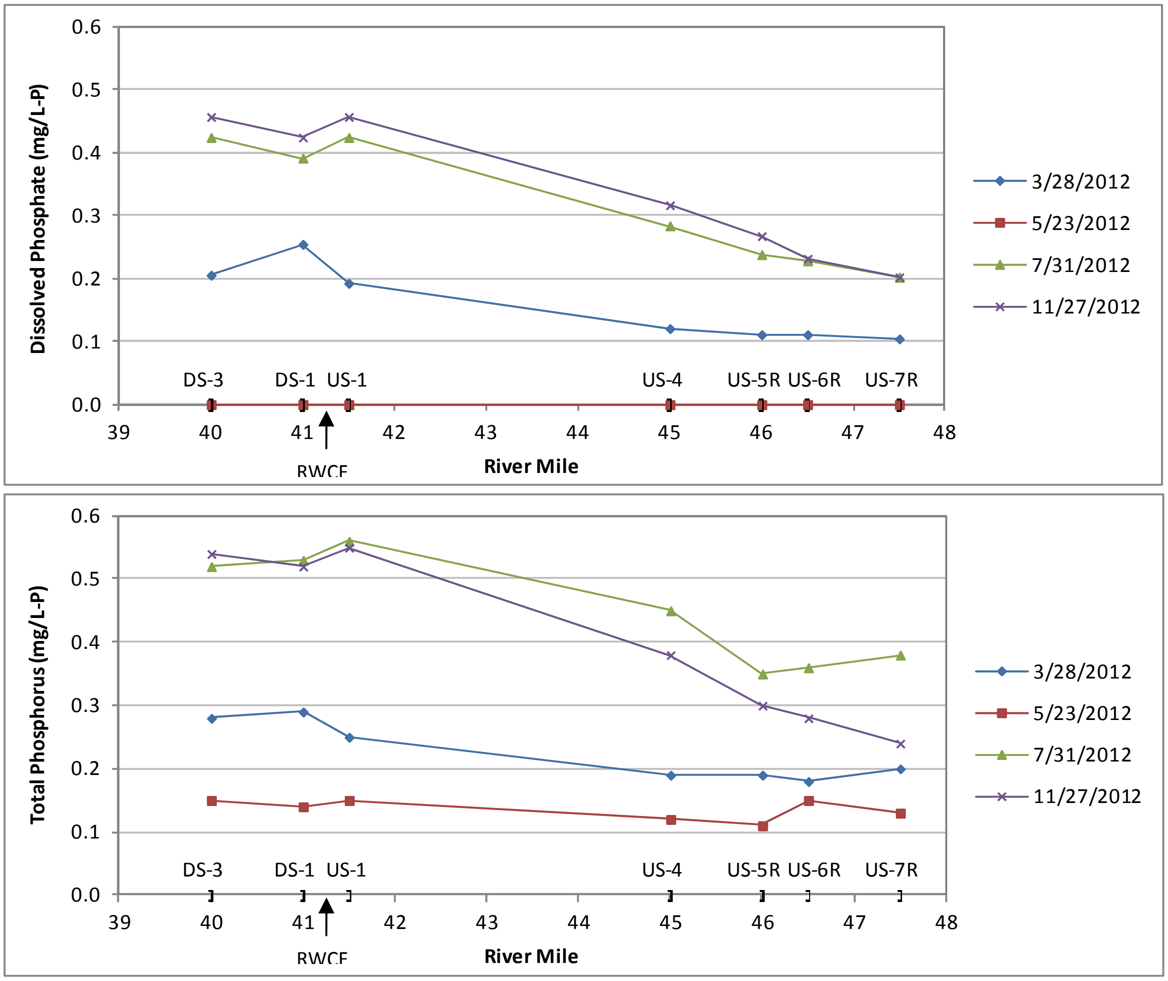


Figure 8. Dissolved phosphate and total-P concentrations measured in water samples taken upstream and downstream of the RWCF outfall into the San Joaquin River from March 2012 through November 2012.

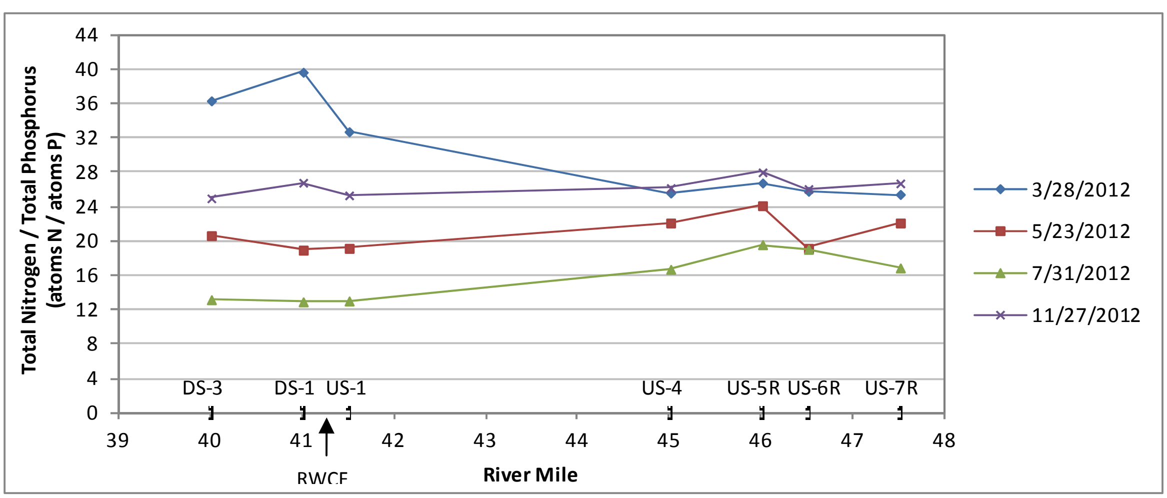


Figure 9. N:P measured in water samples taken upstream and downstream of the RWCF outfall into the San Joaquin River from March 2012 through November 2012.

4.3.2 BENTHIC MACROINVERTEBRATE COMMUNITY

In streams that are nutrient-limited, increases in nutrient levels and primary production may have the secondary effect of increasing the abundance of organisms at higher trophic levels, such as BMI, plankton, and fish. However, due to the complexity of riverine food webs, the relative abundance of organisms at each trophic level may be controlled by top-down as well as bottom-up mechanisms (Shurin et al. 2002). Consequently, determining the effects of nutrient enrichment and increased autotrophic production on BMI community diversity and relative abundance is difficult.

Numerous researchers have demonstrated that BMI communities are limited by food availability in rivers (e.g., Hill et al. 1995; Rosemond et al. 1993; Lamberti 1996). In large rivers, the secondary effects of nutrient enrichment on BMI density or abundance may vary depending on numerous factors, including the depth of sunlight penetration into the water column, water depth, temperature, turbidity, and substrate composition. If conditions are suitable for supporting a diverse BMI community, nutrient enrichment may result in increased BMI abundance. For example, one group of researchers observed a five-fold increase in BMI abundance downstream of a wastewater treatment plant outfall in the St. Lawrence River, Montreal (de Bruyn et al. 2003). However, as discussed in greater detail below, increases in BMI abundance may not be observed in large, deep rivers (such as the lower San Joaquin River) in response to nutrient enrichment if the other abiotic factors discussed above limit the growth of attached periphyton and other forms of benthic algae.

BMI Taxonomic Richness

Metrics for BMI taxonomic richness provide an indication of the number of BMI taxa present at a sampling location. In general, these metric scores decrease in response to degradation of water quality or habitat, particularly for taxa that are intolerant of degraded water quality. However, because these metric scores may respond to numerous water quality and other factors associated with available habitat quality and diversity, they are not useful for identifying a specific cause of the degradation (e.g., elevated nutrient levels or eutrophication).

BMI taxonomic richness was low at all sites during all four sampling events, with a combined total of only 23 different BMI taxa collected from all sites over all sampling events. The number of taxa collected at each site on each sampling day ranged from 4 to 11 (Appendix E, Tables E-1 through E-4). The total number of BMI taxa collected at each site over all four sampling events ranged from 6 (DS-1) to 15 (US-6R).

No specimens belonging to the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), or Trichoptera (caddisflies) were collected at any sites during any of the sampling events. Consequently, metric values for EPT (i.e., the combined number of specimens in these three orders) were zero at all sites during all sampling events (Appendix E, Tables E-1 through E-4).

These three taxonomic orders are represented by species that are generally intolerant of degraded conditions and are typically found in highest abundance in relatively undisturbed water bodies. They are typically absent from water bodies that are degraded by elevated levels of toxic materials or temperatures, or that have degraded habitat conditions. They are uncommon in large, deep rivers, such as the lower San Joaquin River, where the benthos receives little sunlight and the substrate is dominated by fine sediments. Also, no specimens in the order Coleoptera (aquatic beetles) were collected at any sites during any sampling events; therefore, all values for this metric were zero.

The number of predator taxa at each site/event was low, ranging from zero to three (Appendix E, Tables E-1 through E-4). Reference sites (US-5R, US-6R, and US-7R) generally had more predator taxa than downstream sites. No predators were collected from site US-1 during any of the four sampling events, and only one predator taxon (Prostoma, a ribbon worm that occurred in low abundance at most sites) was collected at site DS-1. Predators were collected at reference site US-6R during each of the four sampling events. However, each of the other six BMI collection sites yielded no predators during at least one sampling event. Predators were collected at site DS-3 only during the March sampling event. The low number or lack of predator taxa at all sites suggests that the BMI community is not a typical "top-down" trophic structure, in which the top predators control the abundance of lower trophic levels (Power 1992). The lower numbers of predator taxa occurring in the downstream reaches is likely due to an increased benthic sediment load, which is less suitable for predators and more suitable for supporting burrowing organisms that obtain their food through collecting and filtering. Also, because predators require a prey food base of BMI taxa at lower trophic levels, the low number of predator taxa is likely attributable to a low availability of prey taxa. Moreover, the small and variable numbers of predators occurring at all sites provide no conclusive spatial or temporal trends indicating that the RWCF discharge is causing a shift in the BMI community toward a bottom-up trophic structure or otherwise altering the trophic structure.

BMI Composition

Like the BMI richness metrics discussed above, BMI composition metrics provided little information regarding the effects of nutrient enrichment or eutrophication on the BMI assemblage in the study reach.

BMI Diversity Index Scores

Because of the small number of taxa present at each site (4 to 11), Shannon Diversity Index scores, an indicator of both taxonomic richness and degree of evenness in relative abundance of each taxa, were low at all sites, ranging from 0.4 to 1.4, primarily due to the relatively small number of taxa at most sites and the relatively large variation in abundance. These values were generally similar among all sites sampled within a sampling event, but variable among sampling events. Shannon Diversity Index scores were lowest during the May sampling event, where values ranged from 0.4 (US-7R and US-1) to 0.7 (DS-1), and were highest in July, ranging from 1.0 (US-1) to 1.4 (DS-3). These values provide no indication that the RWCF discharge is having an effect on BMI diversity.

Dominant Taxa

Dominant taxa scores were generally high at all sites, ranging from 37% to 90%, with an average of 63% of the BMI community at each site dominated by one taxon. At all sites and sampling events, the dominant taxon was the amphipod Corophiidae or the annelid (worm) Oligochaeta. This metric score was lowest at site DS-1 (i.e., immediately downstream of the RWCF outfall) in three of the four months, which provides evidence that the RWCF discharge is not adversely altering the composition of the BMI community. Values for this metric were generally high and variable among all sites. Therefore, there is no indication that the RWCF discharge is adversely affecting the BMI community by increasing the proportion of the community dominated by a single taxon.

The majority of BMI taxa present at all sites were not in the class Insecta. Consequently, metric scores for non-insect taxa were also high at all sites, ranging from 70% to 100%, with an average of 91% of the taxonomic composition at each site comprising non-insects (Appendix E, Tables E-1 through E-4). The non-insect taxa that dominated the BMI community composition at each site primarily included arthropods, annelids, mollusks, and turbellaria. This metric provides no evidence that the RWCF discharge or associated nutrient levels are having any measurable adverse effect on the number of non-insect taxa in the BMI community. In water bodies that are dominated by taxa in the class Insecta, a site that is dominated by non-insect taxa may be an indicator of degraded conditions, depending on other factors (e.g., tolerance of the non-insect taxa). However, because the BMI community at all sites were dominated by non-insect taxa, this metric suggests that the predominance of non-insect taxa is due to similar conditions at all sampling locations.

In a study of point-source pollution effects on BMI communities in an agricultural-dominated watershed in Germany, Chambers et al. (2006) observed a change in BMI communities from one that is dominated by chironomids (midges) and amphipods (scuds) to one that is dominated by oligochaetes (aquatic worms) and gastropods (snails) in response to decreases in inorganic nitrogen. However, this shift was not observed in the BMI community sampled in the seven lower San Joaquin River sampling locations. As discussed above, levels of inorganic nitrogen were generally lower upstream of the RWCF outfall. Chironomid abundance was low overall at all sampling locations. However, chironomid abundance was higher overall (n=54 combined total) at the three reference sites, where inorganic nitrogen was lowest. By comparison, a combined total of eight chironomids were collected in the mixing zone sampling locations,where inorganic nitrogen was the highest. Amphipods and oligochaetes were present in relatively high abundance at all sites during all events, and combined to account for the majority of BMIs present at all sampling locations. However, abundance of these taxa varied among sites and sampling events, and showed no observable response to inorganic nitrogen levels. A combined total of eight gastropods was collected at all sampling locations over all four events and likewise showed no observable response to inorganic nitrogen levels.

BMI Community Tolerance

The metric scores for BMI community tolerance indicate that the BMI community in the study reach is dominated by taxa that are moderately to highly tolerant of degraded conditions, and devoid of taxa that are intolerant of such conditions. Of the 23 combined total taxa collected during the surveys, the tolerance value for each taxon ranged from 4 to 10 and averaged 6.4 (values are on a scale of 1 to 10, with higher values indicating higher tolerance to degraded conditions). The tolerance value metrics at each site ranged from 4.2 to 7.8. Overall, tolerance values were lowest and most consistent during the May sampling event, when values ranged from 4.2 at US-6R to 4.6 at DS-1 (Appendix E, Table E-2). Tolerance values were variable among sampling locations and do not provide any spatial or temporal trends indicating that the RWCF discharge is causing a shift in the BMI community toward higher tolerances. Although these relatively high tolerance values and a lack of intolerant taxa indicate that water quality conditions at all sites are moderately degraded, they provide no information for determining the source of the degradation.

As discussed above, no organisms classified as intolerant were collected at any of the seven sampling sites during any of the four sampling events. Therefore, all metric scores for intolerant organisms and intolerant taxa were 0.0%. Metric scores for tolerant organisms were lowest and most consistent during the May sampling event, ranging from 0.7% (US-6R) to 2.6 (US-5R) (Appendix E, Table E-2), but were more variable among sites during the remaining three sampling events. Overall, a comparison of temporal and spatial scores for this metric between reference and mixing zone sites showed no consistent trends that would indicate that the RWCF discharge is affecting BMI community tolerance. Likewise, tolerant taxa metric scores were highly variable, ranging from 17% to 50%, with no clear spatial or temporal trends indicating that the RWCF discharge is affecting BMI community tolerance.

Functional Feeding Groups

Of the BMI metrics calculated, changes in the composition of FFGs are the best indicators of the effects of increased algae growth or eutrophication on BMI communities. Scrapers (including grazers), which primarily consume periphyton, are typically present in highest abundance when nutrient levels are elevated and algae growth is increased (Miltner and Rankin 1998). Therefore, scraper abundance is a good indicator of elevated nutrient levels. However, FFGs classified as scrapers typically were not present at most sampling locations during each sampling event. In the few sites from which scrapers were collected, they were present in low abundance, ranging from 0% to 1.2% of the BMI assemblage (**Figure 10**; Appendix E, Tables E-1 through E-4). The absence or relatively low abundance of scrapers in the BMI community at each site is likely due to several factors. First, the river substrate at all seven sampling locations was composed of fine, unstable sediments, which are generally unsuitable for supporting the growth of attached periphyton, which scrapers and grazers consume—no periphyton was observed in the petite Ponar grabs. Second, based on the Secchi disk readings, the photic (i.e., sunlight-receiving) zone reached depths ranging from 1.2 to 6.5 ft, with an average of 3.4 ft. The water depth at all BMI sampling location was substantially greater than the Secchi disk readings; therefore, sunlight is not penetrating the water column by a sufficient amount to facilitate the growth and proliferation of benthic periphyton, regardless of nutrient levels. Consequently, the benthic conditions at all sampling locations were largely unsuitable for supporting scrapers or grazers.

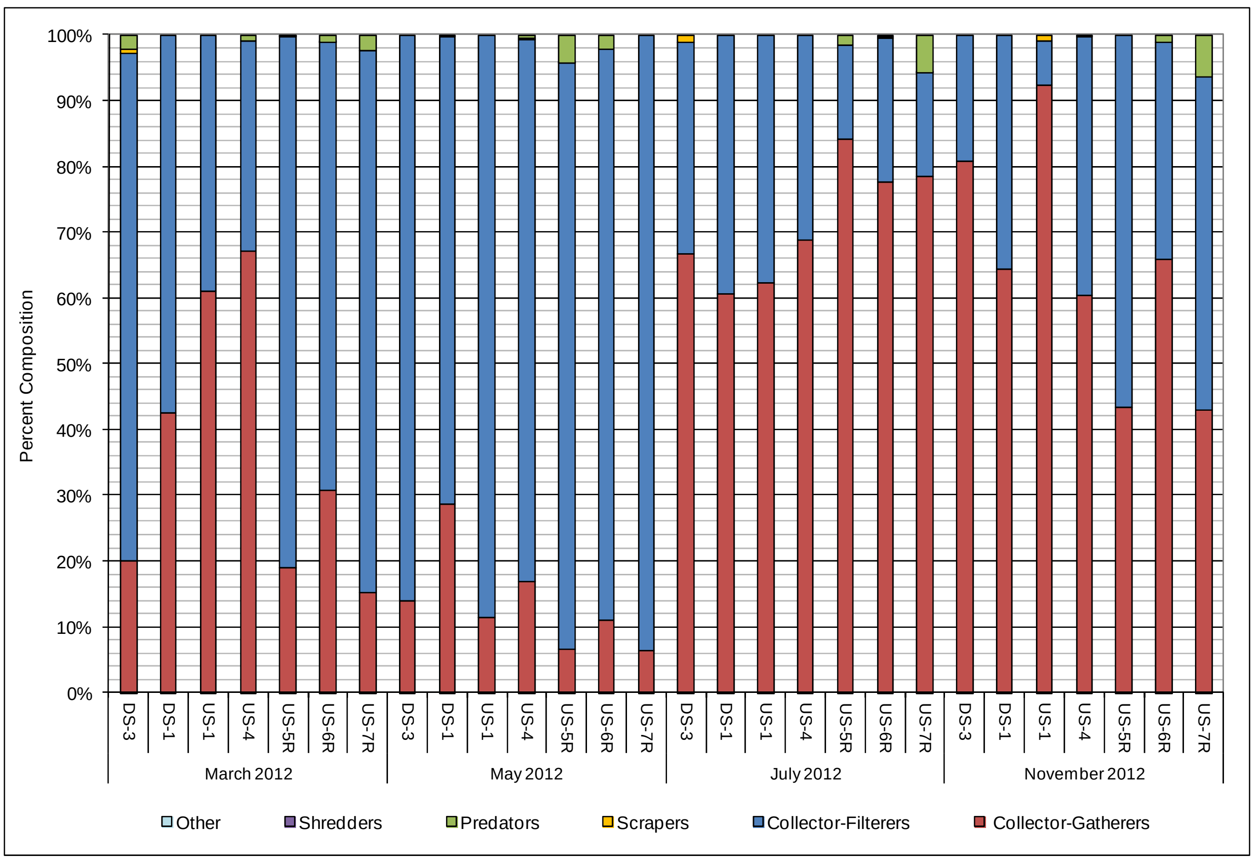


Figure 10. BMI functional feeding group composition at seven lower San Joaquin River sampling locations in March, May, July, and November 2012.

Collector-filterers and collector-gatherers, which consume fine particulate organic matter, dominated the FFG composition of the BMI community at all sampling locations. These FFGs, which either filter their food from the water column or gather it from the sediments, combined to account for 90% or more of the BMI assemblage at all sampling locations during all sampling events (Figure 10; Appendix E, Tables E-1 through E-4).

As discussed above, predators typically were not present, or were present in low abundance, ranging from 0% to 6.4% of the BMI assemblage at each site/event. The highest proportion of predators typically occurred at the three upstream reference sites. As stated, the low proportion of predators is likely due to a low availability of BMI prey taxa at lower trophic levels at all sites, but particularly in the downstream sites where the increased load of fine sediments and lack of stable substrates in the river bed favor burrowing organisms over predators, resulting in a bottom-up trophic structure, rather than a top-down trophic structure in which predators control BMI abundance. For reference, Merritt and Cummins (1996b) cite a predator to prey ratio of 0.15 for top-down control in BMI communities. The observed ratio at all sites in the study reach was lower than this value (ranged from 0.0 to 0.07), indicating that the BMI community has a bottom-up trophic structure in which the low availability of prey is responsible for the low relative abundance of predators.

Shredders, which are found in pristine headwater streams, where they consume coarse particulate matter, are not typically found in valley floor rivers of the Central Valley, and were not collected at any sampling locations during any sampling events. Also, no FFGs classified as "other" were collected during this study.

Overall, the relative proportions of FFGs at all sites upstream and downstream of the RWCF outfall indicate that the trophic structure of the BMI community is heavily dominated by collector-filterers and collector-gatherers, with other FFGs (scrapers, shredders, and predators) absent or present in very low abundance. Consequently, the trophic structure and integrity of the BMI community at all reference and study locations is considered to be altered to some degree; in other words, it is not representative of an unaltered BMI community, which would have a more balanced representation of FFGs. Although the FFG composition indicates that the BMI community is likely altered in response to degraded conditions at all sampling locations, there is no evidence to suggest that the RWCF discharge has caused or observably exacerbated the imbalance in FFG composition.

Comparison of BMI Assemblage by Site

The BMI community compositions at all sites were compared within each sampling event using Sϕrenson's (1948) Index of Similarity. This index allows for pairwise comparisons of overlap in taxonomic composition (i.e., presence/absence) among all sites, and is calculated as:

$$QS = 2C / (A + B),$$

where QS is the index of similarity, A is the number of individual taxa present in one site, B is the number of individual taxa present in the comparison site, and C is the number of species common to both sites. Values range from 0 to 1, where values of 0 indicate no species overlap and values of 1 indicate that the BMI taxonomic composition is identical at both sites (i.e., 100% overlap in taxonomic composition).

For the study sites, Sϕrenson's QS values ranged from 0.50 to 1.00 for all pairwise comparisons of sites (**Table 11**), indicating that a relatively high degree (50–100%) of taxonomic overlap was observed among all sites within each of the four sampling events. The QS values for pairwise comparisons of the three upstream reference locations (US-5R, US-6R, and US-7R) ranged from 0.50 to 0.83, with an average of 0.725. This value indicates that, on average, the BMI community taxonomic composition overlapped by 72.5% among the three sampling locations in the reach upstream of the influence of the RWCF discharge. By comparison, QS values for all pairwise comparisons of the four sampling locations within the influence of the RWCF discharge (DS-3, DS-1, US-1, and US-4) to the three reference locations ranged from 0.55 to 1.00, with an average of 0.713. This indicates that the taxonomic composition of the BMI community in the four sites affected by the RWCF discharge overlapped, on average, by 71.3% with the compositions of the reference sites—an overlap value that is nearly identical to the 72.5% degree of taxonomic overlap observed among the three reference sites. Based on the relatively high degrees of similarity observed between the reference sites and sites within the influence of the RWCF discharge, the RWCF discharge does not appear to cause a measurable change in the taxonomic composition of the BMI community in the lower San Joaquin River.

Overall, the Sϕrenson's QS values indicated a relatively high degree of taxonomic similarity among the sites and showed no observable spatial or temporal trends indicating that the RWCF discharge is adversely affecting the taxonomic composition of the BMI community at any of the sampling locations. The range of QS values comparing the sites within the mixing zone reach, and between the mixing zone reach and the reference reach, were similar to the variability observed among the three sites in the reference reach.

4.3.3 SUBMERGED AND EMERGENT VEGETATION

Submerged and emergent vegetation observed at all sampling locations consisted almost exclusively of bulrushes (*Scirpus* spp.), Brazilian waterweed (*Egeria densa*), and water hyacinth (*Eichhornia crassipes*) (**Figure 11**; Appendix F). Bulrushes are endemic to California and occur commonly in wetlands and near-shore areas of waterways in the Central Valley. Water hyacinth and Brazilian waterweed are invasive nuisance species that have become widespread throughout the San Joaquin River and Delta. Studies of the effects of nutrient enrichment from wastewater effluent discharges have documented increases in plant biomass (Chambers and Prepas 1994; Gucker et al. 2006). However, the availability of sunlight and nutrients are two primary limiting factors affecting the growth of aquatic vegetation. All three of the aquatic macrophytes observed in this study are rooted and therefore obtain nutrients primarily from sediments. As such, elevated nutrients in the water column must be absorbed into the sediments before they are available for aiding the growth of these species.

Table 11. Sϕrenson's index of similarity (QS) values for pairwise comparisons of BMI taxonomic composition at all San Joaquin River sites sampled in March, May, July, and November 2012.

March 28, 2012						
	DS-1	US-1	US-4	US-5R	US-6R	US-7R
DS-3	0.77	0.62	0.75	0.80	0.63	0.67
DS-1		0.80	0.77	0.83	0.63	0.67
US-1			0.77	0.83	0.63	0.67
US-4				0.93	0.84	0.89
US-5R					0.78	0.82
US-6R						0.76
May 23, 2012						
	DS-1	US-1	US-4	US-5R	US-6R	US-7R
DS-3	0.80	0.89	0.71	0.83	0.71	1.00
DS-1		0.89	0.71	0.83	0.57	0.80
US-1			0.62	0.73	0.62	0.89
US-4				0.88	0.67	0.71
US-5R					0.75	0.83
US-6R						0.71
July 31, 2012						
	DS-1	US-1	US-4	US-5R	US-6R	US-7R
DS-3	0.80	0.80	0.73	0.71	0.71	0.71
DS-1		1.00	0.89	0.67	0.67	0.67
US-1			0.89	0.67	0.67	0.67
US-4				0.62	0.62	0.62
US-5R					0.75	0.75
US-6R						0.75
November 27, 2012						
	DS-1	US-1	US-4	US-5R	US-6R	US-7R
DS-3	1.00	0.60	0.80	0.89	0.55	0.89
DS-1		0.60	0.80	0.89	0.55	0.89
US-1			0.50	0.55	0.62	0.55
US-4				0.91	0.62	0.73
US-5R					0.50	0.80
US-6R						0.50

Figure 11. Three submerged and emergent vegetation species observed at seven sampling locations in the lower San Joaquin River: (A) bulrushes (*Scirpus* spp.), (B) water hyacinth (*Eichhornia crassipes*), and (C) Brazilian waterweed (*Egeria densa*).

Photos of conditions at each sampling location, taken during seasonally representative sampling events, are provided in Appendix F. Bulrushes occurred frequently in small to medium-sized patches near the river margins at most sampling locations, and were observed during all sampling events. Water hyacinth and Brazilian waterweed were observed primarily in July, September, November, and January, in small to medium-sized patches along the river margins; also, uprooted patches of these plants were observed floating at the river surface mid-channel. The roots of free-floating patches of vegetation observed at the river surface extended no more than approximately 1 ft from the river surface (i.e., to less than 10% of the river depth). In no cases did the submerged or emergent vegetation cover more than approximately 5% of the entire channel cross section at any sampling site, or at any location between the sampling sites.

Vegetation observed at all sampling locations in March and May 2012 consisted almost entirely of small patches of bulrushes along the river margins that typically covered less than 1% of the total channel surface area (**Table 12**). In July 2012, small patches of Brazilian waterweed were observed in the near-shore margins and free-floating at the river surface mid-channel. The total estimated coverage of rooted and floating vegetation was less than 1% at all sites in July 2012. In September 2012, small to medium-sized patches of water hyacinth were observed drifting mid-channel at all sites, and small patches of rooted water hyacinth were observed in the near-shore margins at sites DS-2, US-1, US-2, and US-4.

Table 12. Estimated percentage of near-shore and mid-channel surface area occupied by submerged and emergent vegetation within 100 ft upstream and 100 ft downstream of each sampling location.

Sampling Location	Estimated Area of Coverage (%)					
	March 28, 2012	May 23, 2012	July 31, 2012	September 26, 2012	November 27, 2012	January 28, 2013
DS-4	<1	<1	<1	5	<1	<1
DS-3	0	<1	<1	5	<1	0
DS-2	<1	<1	<1	2	<1	<1
DS-1	<1	<1	<1	2	<1	<1
US-1	<1	<1	<1	2	<1	<1
US-2	<1	<1	<1	2	<1	<1
US-3	<1	<1	<1	<1	<1	<1
US-4	<1	<1	<1	<1	<1	0
US-5R	<1	<1	<1	<1	<1	0
US-6R	<1	<1	<1	<1	<1	<1
US-7R	<1	<1	<1	<1	<1	0

The estimated area of vegetative coverage was highest in September, steadily increasing from less than 1% in the upstream reference sites to approximately 5% at the furthest downstream sampling locations. The increase in vegetative coverage was primarily due to increased concentrations of free-floating patches of water hyacinth drifting downstream. In addition, patches of water hyacinth were observed between sampling locations at numerous locations throughout the study area, and in the dead-end portion of the DWSC of the San Joaquin River, upstream of the confluence of the lower San Joaquin River (i.e., outside the study area). Furthermore, a dense colony of Brazilian waterweed was observed at and around the boat launch at the upstream terminus of the DWSC (i.e., upstream of the study reach).

In November 2012, the estimated vegetative coverage decreased to less than 1% of the total surface area. Vegetative coverage in November consisted of small free-floating patches of water hyacinth at all 11 sampling locations, with bulrushes and Brazilian waterweed occurring in the near-shore margins at most sites. In January 2013, vegetative coverage ranged from 0% to less than 1% at all 11 sampling locations. Vegetation included small patches of bulrushes at the river margins and infrequent, mid-channel, small patches of free-floating water hyacinth.

The percent coverage of submerged and emergent vegetation was low at all locations during all sampling events. During most months, the coverage was less than 1% of the surface area at each sampling location and was confined to within a few feet from shore in the shallow river margins, or to small patches of water hyacinth and Brazilian waterweed that uprooted and floated downstream in the mid-channel at the water surface. Aquatic vegetation was distributed relatively evenly among all sampling locations during all months, with the exception of September. In September, patches of free-floating water hyacinth that uprooted from colonies within and upstream of the study reach were observed at all sites, and concentrations of these patches increased from the upstream reference sites to the most downstream site (DS-4) in the

DWSC. However, in the study area, no observable trends in abundance or density of water hyacinth colonies growing in the river margins were observed that would indicate a response to elevated nutrient levels from the RWCF discharge. At no time or location were conditions observed in which the density or abundance of any submerged or emergent vegetation reached levels that would restrict the passage of aquatic life or would be considered a nuisance.

4.3.4 ALGAE COMMUNITIES

Chlorophyll *a* concentrations, measured in 15-minute intervals by sondes in the San Joaquin River at Vernalis (RM 72.25) and Garwood Bridge (RM 42), plus data from the RWCF receiving water monitoring sites, were compiled to assess the seasonal algal dynamics of the lower San Joaquin River in the vicinity of the RWCF outfall. Garwood Bridge is located less than a mile upstream of the RWCF discharge. Data on concentrations of chlorophyll *a* (sondes maintained by DWR) and river flow were obtained for the years 2010, 2011, and 2012 (**Figure 12**; Figures D-1 and D-2 in Appendix D). When river discharge was low, typical of most summer months, chlorophyll *a* concentrations at Vernalis were generally above 15 micrograms per liter (µg/L), while concentrations downstream at Garwood Bridge generally were less than 5 µg/L. When river discharge was high (2010), chlorophyll *a* concentrations at Vernalis and Garwood Bridge were low and were similar. Thus, flow is a primary driver of algal biomass (as measured by chlorophyll *a*) at Garwood Bridge. Except for a few short periods, the chlorophyll *a* concentrations at Garwood Bridge from 2010 through 2012 were less than the mesotrophic-eutrophic threshold for rivers (30 µg/L; USEPA 2001). Because chlorophyll *a* concentrations at Garwood Bridge often ranged from 0 to 20 µg/L during the summer months, when river temperatures are highest, the algal productivity in the vicinity of the RWCF outfall can be classified as oligotrophic to mesotrophic.

The RWCF receiving water monitoring data from 2010 through 2012 also show that chlorophyll *a* concentrations in the vicinity of the RWCF outfall (at site RSW-002/002A) were less than those measured at the upstream monitoring site outside the influence of the RWCF discharge (RSW-001; Figure D-4 in Appendix D). Comparison of the trends in nitrate and chlorophyll *a* further suggest that the loss of chlorophyll *a* between RSW-001 and RSW-002A occurred independent of changes in nitrate concentrations (Figure D-3 in Appendix D).

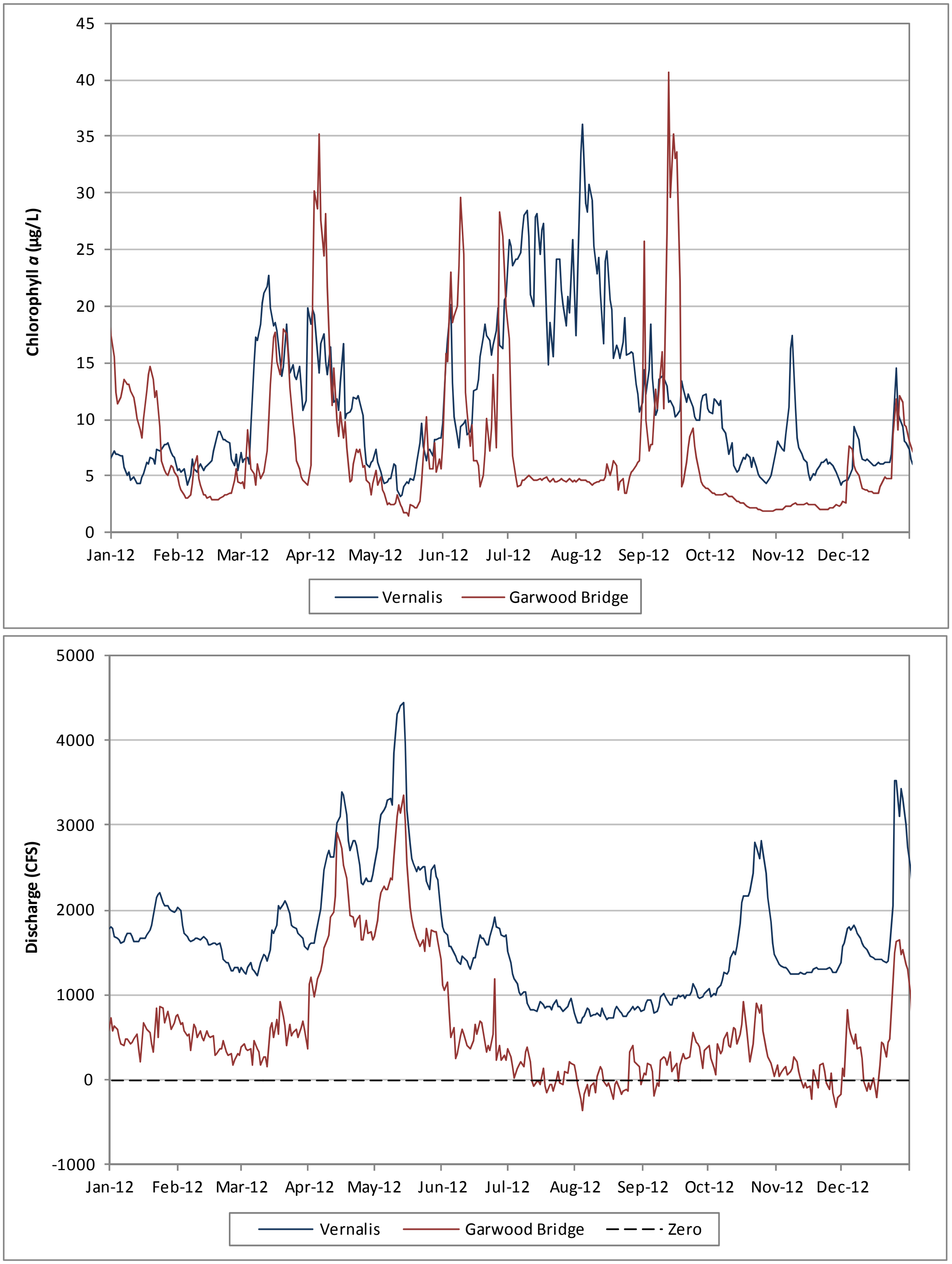


Figure 12. Chlorophyll *a* (top) and discharge (bottom) measured for the San Joaquin River at Vernalis and Garwood Bridge for January 2012 through December 2012. Data were compiled from CDEC.

Because chlorophyll *a* concentrations represent only a single measure of total algal abundance, further algal metrics were measured to determine the influence of nitrate from the RWCF discharge on algae community dynamics. Standard metrics used in the analysis of phytoplankton communities are density (cells/milliliter [mL]) and a morphometric measurement, biovolume, from which biomass (mg/L) can be estimated (USEPA 2010). *Density* refers to the abundance or number of cells in the sample and is independent of algal morphology and size. Density provides information on the composition of the entire community, the dominance of each species, and changes in abundance and composition over time. *Biomass* is an assessment of the magnitude of the algae community in terms of mass, and is determined by the density and morphology (geometry/size) of the enumerated algae. As a metric, algal biomass is valuable because it is directly related to the amount of material and energy available at the base of the aquatic food web.

The total algal biomass and density measured in the San Joaquin River during the study are presented in **Table 13**. November yielded the lowest amount of algal biomass and abundance. In contrast, May was the most productive month in terms of biomass, yet the density of algae in May and July were somewhat similar. Total algal density and biomass were greater in the upstream reference sites in March and July, although May and November also exhibited this trend to a lesser extent. Combining all sampling events, total algal biomass and density were significantly greater at the reference sites compared to the reach influenced by the RWCF discharge (**Table 14** and **Table 15**).

Table 13. Total algal biomass and density in water samples taken during 2012 from the San Joaquin River in the vicinity of the RWCF discharge and in the upstream reference reach.

Metric	Date	Sample Site (River Mile)						
		DS-3 (40)	DS-1 (41)	US-1 (41.5)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
Biomass (mg/L)	3/28/2012	0.79	1.29	0.42	3.32	2.09	4.42	3.03
	5/23/2012	4.59	5.90	5.79	8.25	9.80	8.75	9.72
	7/31/2012	0.59	0.45	0.34	1.82	4.13	3.32	11.10
	11/27/2012	0.15	0.15	0.15	0.11	0.13	0.15	0.21
Density (cells/mL)	3/28/2012	13,195	5,606	4,918	14,381	16,598	24,864	21,618
	5/23/2012	14,911	16,914	18,072	17,099	24,184	19,285	59,196
	7/31/2012	24,131	21,662	13,368	11,726	45,082	25,546	32,649
	11/27/2012	5,966	5,856	7,451	4,576	7,115	8,279	11,319

The trends in algal metrics were dominated by a few algal phyla (**Figure 13** through **Figure 16**; Appendix D). Species of green algae (Chlorophyta), chrysophytes (Chrysophyta), cryptophytes (Cryptophyta), cyanobacteria (Cyanophyta), euglenophytes (Euglenophyta), pyrrhophytes (Pyrrhophyta), and diatoms (Bacillariophyta) were identified during algal enumeration of San

Joaquin River samples taken in March, May, July, and November 2012. Analysis of means showed that algal biomass and density of the dominant algal phyla (diatoms, green algae, and cryptophytes) decreased significantly between the reference reach and the mixing zone, with the exception of cyanobacteria (Table 14 and Table 15), consistent with findings for chlorophyll *a*, discussed above.

Table 14. Least-squares mean values of algal taxa and total algal biomass measured in water samples taken during 2012 from the San Joaquin River in the mixing zone and reference reaches.

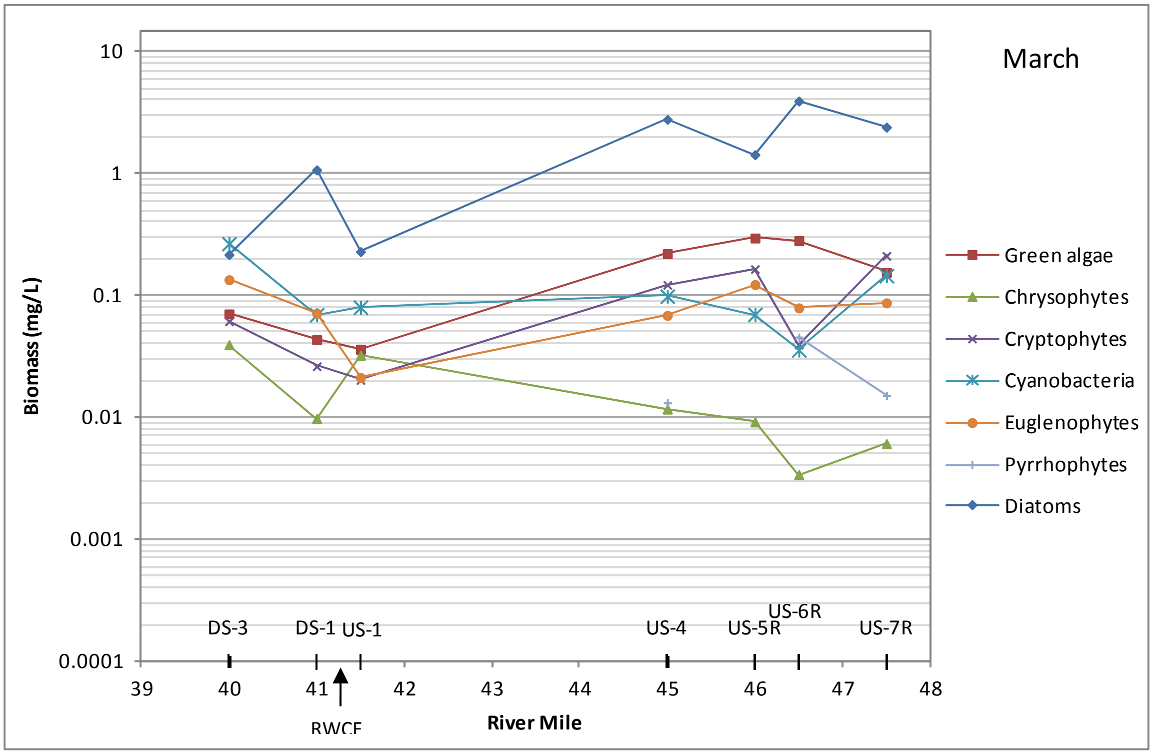
	Means [a]		P-values [b]	
	Reference Reach	Mixing Zone Reach	Site US-4 Included	Site US-4 Excluded
Diatom biomass (mg/L)	4.32	1.44	0.0012	0.0013
Green algae biomass (mg/L)	0.12	0.06	0.0230	0.0137
Cyanobacteria biomass (mg/L)	0.16	0.13	0.6030	0.4428
Cryptophyte biomass (mg/L)	0.10	0.06	0.0384	0.0272
Total algal biomass (mg/L)	4.74	1.72	0.0009	0.0009

[a]Least-squares means were calculated excluding site US-4. See Appendix G for more details.
[b]P-values were generated from a general linear model analysis, and comparison was made between the results obtained when site US-4 was included or excluded from the analysis. See Appendix G for more details.

Table 15. Least-squares mean values of algal taxa and total algal density measured in water samples taken during 2012 from the San Joaquin River in the mixing zone and reference reaches.

	Means [a]		P-values [b]	
	Reference Reach	Mixing Zone Reach	Site US-4 Included	Site US-4 Excluded
Diatom density (cells/mL)	7,387	2,934	0.0002	<0.0001
Green algae density (cells/mL)	2,174	767	0.0004	0.0001
Cyanobacteria density (cells/mL)	14,068	8,416	0.2608	0.0753
Cryptophyte density (cells/mL)	943	475	0.0218	0.0159
Total algal density (cells/mL)	24,645	12,671	0.0241	0.0037

[a] Least-squares means were calculated excluding site US-4. See Appendix G for more details.
[b]P-values were generated from a general linear model analysis, and comparison was made between the results obtained when site US-4 was included or excluded from the analysis. See Appendix G for more details.



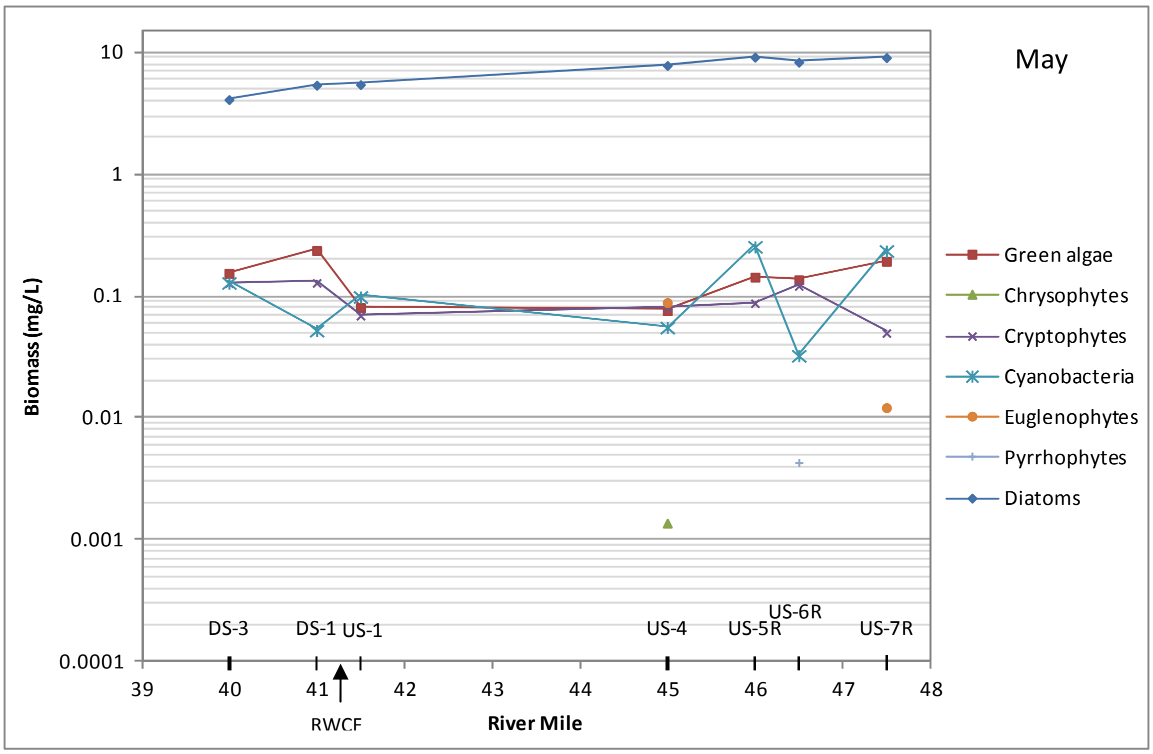


Figure 13. Algal biomass by phyla (division) in water samples taken upstream and downstream of the RWCF discharge in the San Joaquin River during March and May 2012.

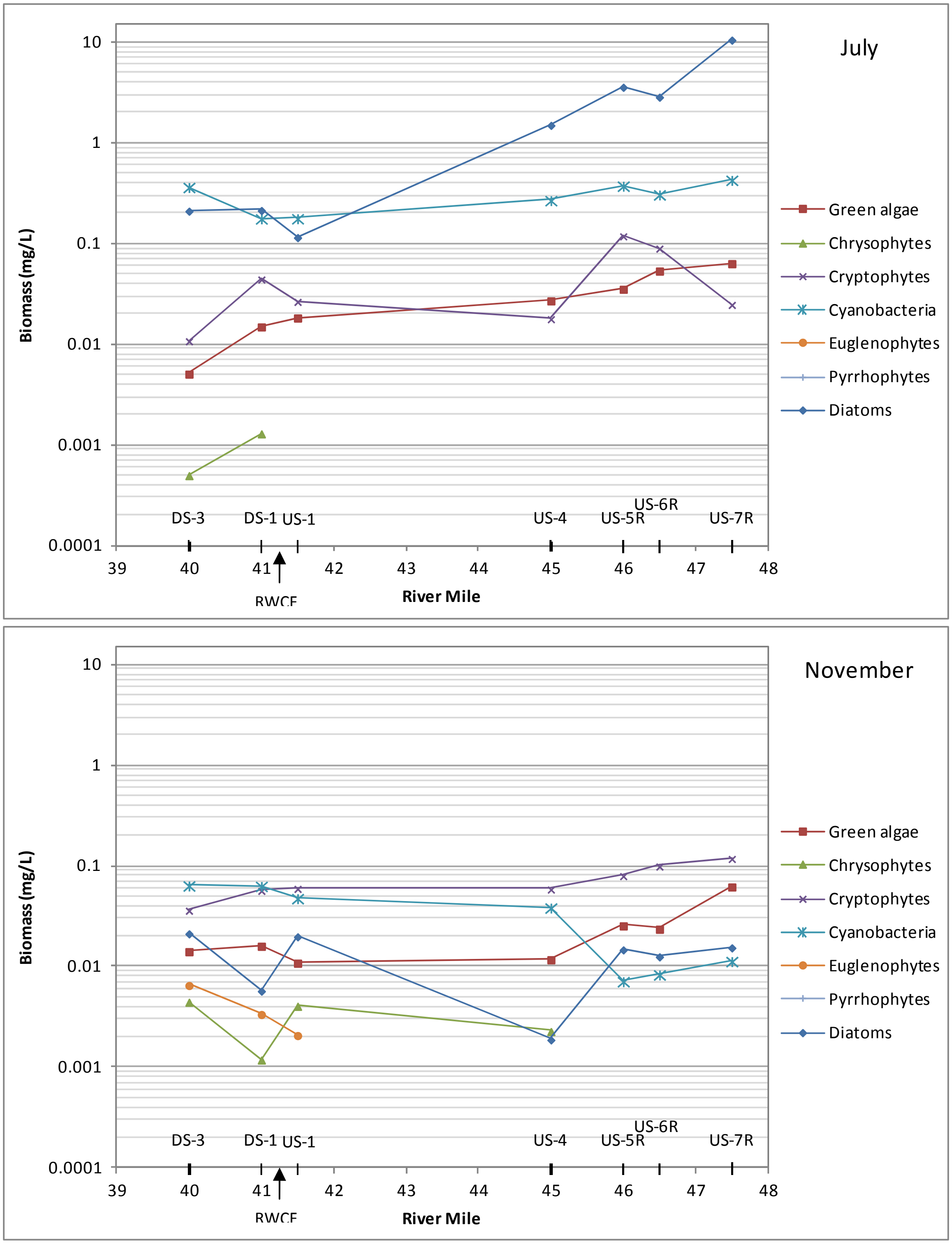


Figure 14. Algal biomass by phyla (division) in water samples taken upstream and downstream of the RWCF discharge in the San Joaquin River during July and November 2012.

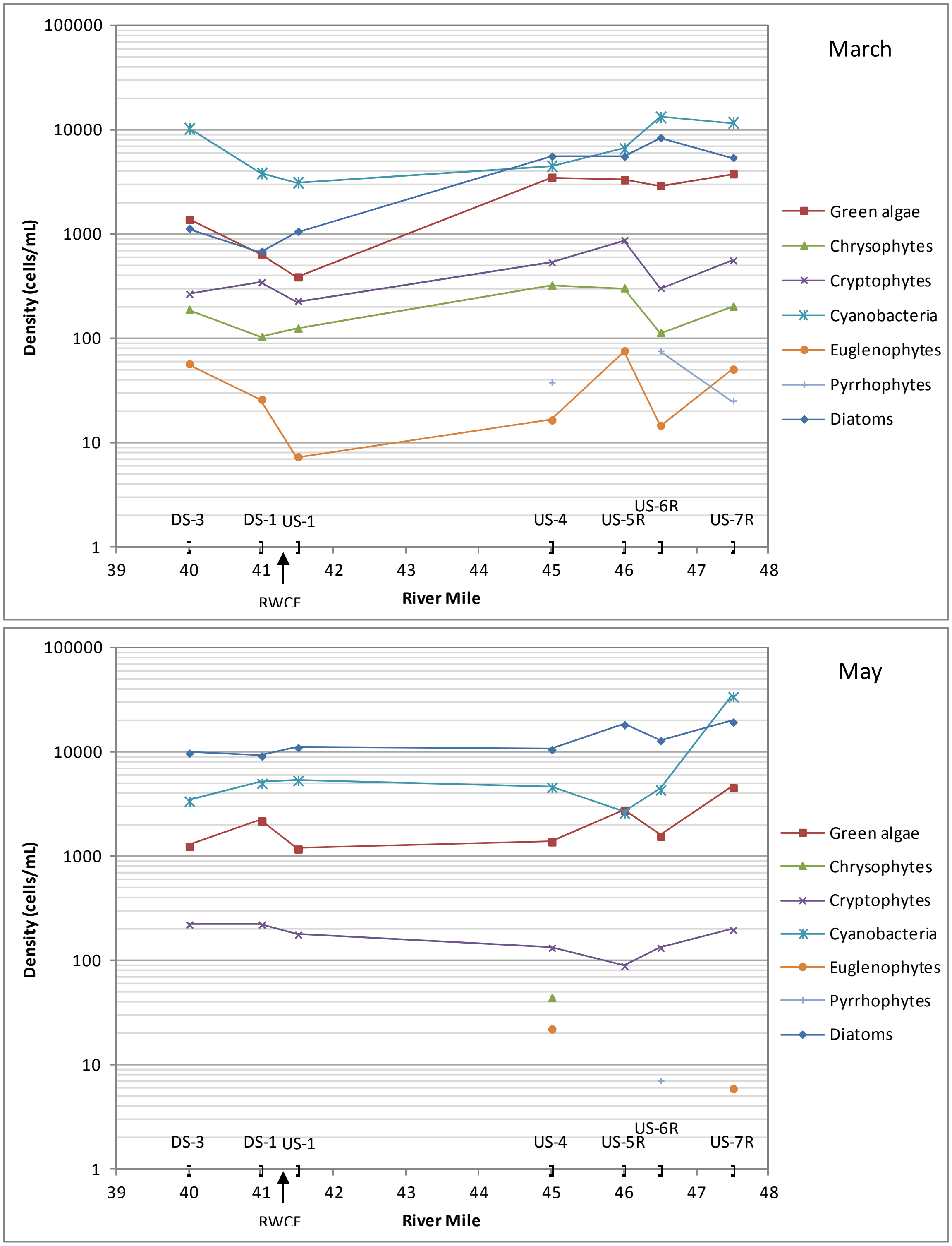


Figure 15. Algal density by phyla (division) in water samples taken upstream and downstream of the RWCF discharge in the San Joaquin River during March and May 2012.

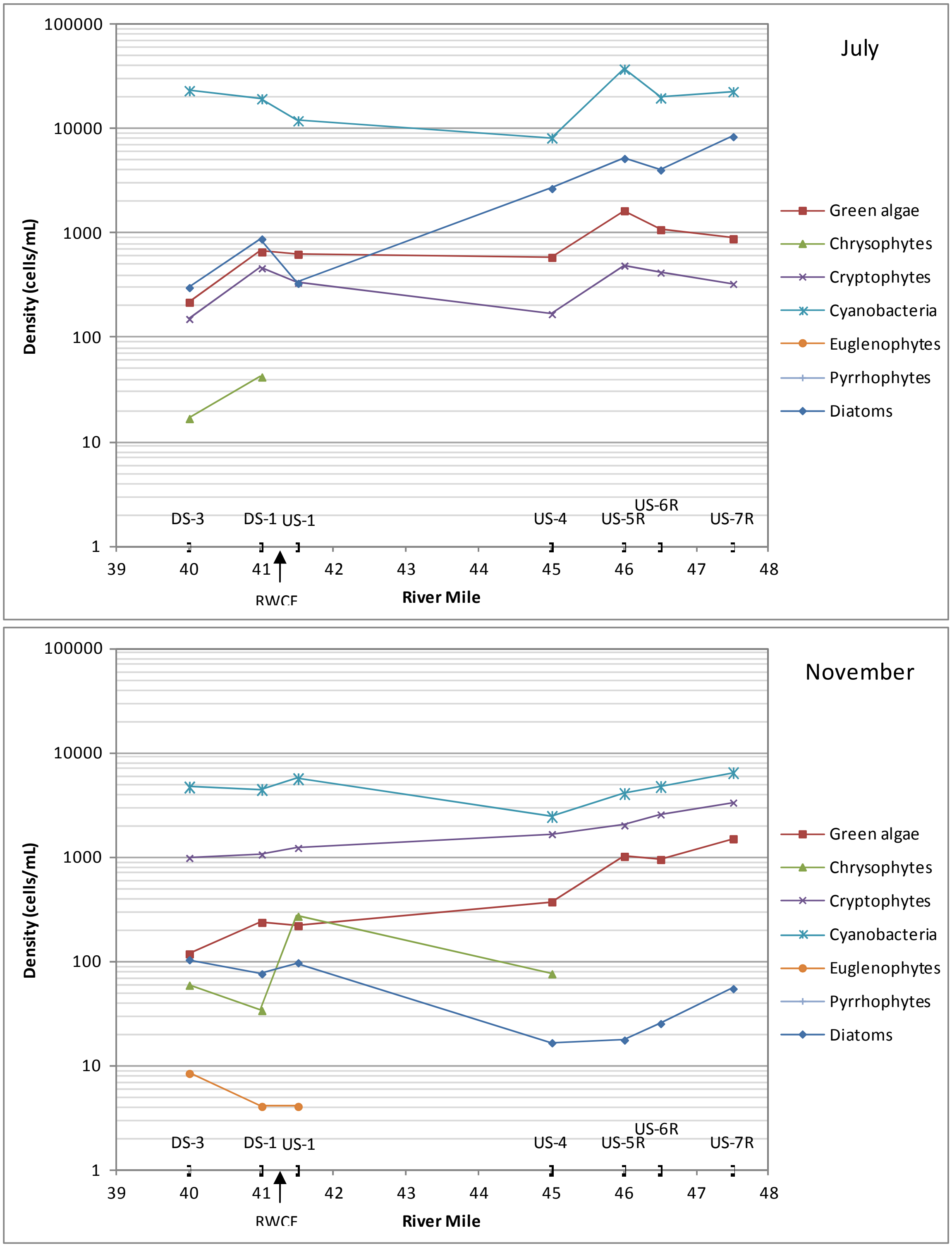


Figure 16. Algal density by phyla (division) in water samples taken upstream and downstream of the RWCF discharge in the San Joaquin River during July and November 2012.

In November, cryptophytes and green algae dominated the algal biomass of the reference sites, but their density and biomass decreased with distance downstream. Cyanobacteria density was greatest across all sampling sites in November, and their biomass increased downstream of the reference sites. During March, May, and July, species of cyanobacteria, diatoms, and (to a lesser extent) green algae were the most abundant algal species; however, diatoms tended to dominate total algal biomass in these months. As such, changes in diatom biomass were responsible for the loss of algal biomass, moving from the reference sites downstream toward the DWSC. While much less abundant than the diatoms, green algae density and biomass also showed a decreasing trend from upstream to downstream of the RWCF discharge. Cyanobacteria biomass remained fairly constant among the reference, upstream, and downstream reaches during March, May, and July; however, the density of cyanobacteria was generally highest in the reference reach, decreased upstream of the RWCF outfall, and then increased downstream of the outfall. Because of the substantial loss of diatoms from upstream to downstream in the months of March and July, the contribution of cyanobacteria to total algal density and biomass increased with decreasing river mile during these months (**Figure 17** and **Figure 18**). Due to their high density and biomass, the loss of diatoms between the reference sites and the sites influenced by the discharge was the primary and most significant change that occurred in the algae community during the study.

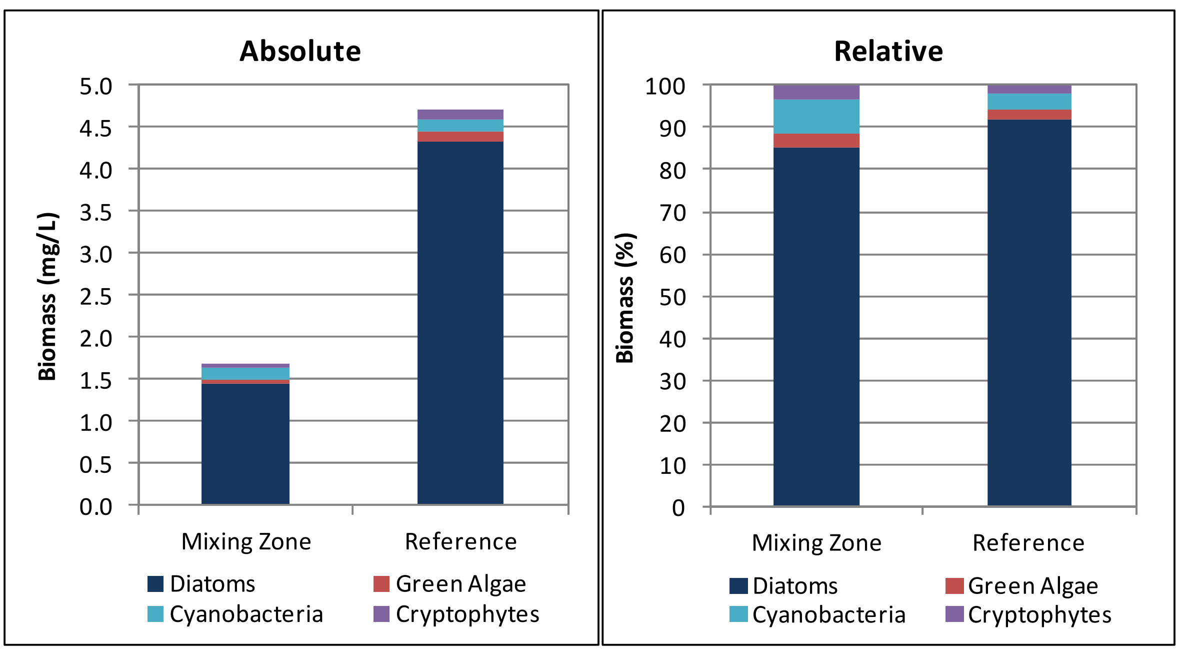


Figure 17. Absolute and relative average algal biomass of the dominant algal phyla observed at the reference sites and the mixing zone sites. Data are also shown in Table 14.

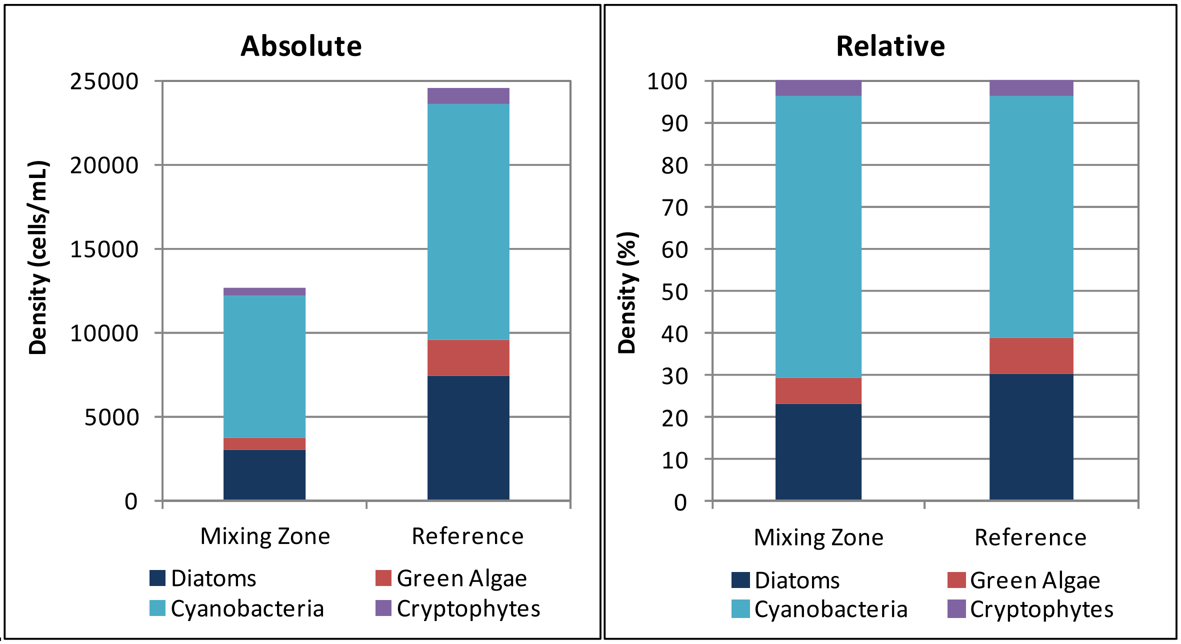


Figure 18. Absolute and relative average algal density of the dominant algal phyla observed at the reference sites and the mixing zone sites. Data are also shown in Table 15.

A statistical analysis was conducted to determine the relative influence of the environmental parameters measured or estimated at each sampling site (nitrate, dissolved phosphate, Secchi depth, temperature, percent effluent, and river velocity) on algal biomass and density (details of the analysis are in Appendix G). River velocity and percent effluent were estimated by DSM2 modeling for each sampling site and sampling event (within 15 minutes of when each sample was taken). Statistical models assessed the combined influence of the environmental parameters on the algae community. In general, the explanatory power (R^2) of a model increases as additional factors are added, and, as expected, the most complex models evaluated had the highest R^2 values (Tables G-4 and G-7 in Appendix G). The loss in explanatory power as variables are dropped provides some insight into the relative importance of the environmental parameters. However, because many of the environmental parameters showed some degree of correlation, the amount of variance explained is not completely additive. Nonetheless, a highly parameterized model that included nutrients (nitrate and dissolved phosphate) and physical factors (Secchi depth, velocity, and temperature) was compared both with a model consisting only of nutrients and with another model consisting only of physical factors. The comparison shows that physical factors contributed the bulk of explanatory capacity for nearly all algal taxa biomass and density measures (Table G-8 in Appendix G). Notably, physical factors accounted for nearly all of the variance in cyanobacteria and diatom biomass and density.

A complementary assessment, partial correlation analysis, also was performed. This analysis showed that variability in biomass across all phyla, but especially for cyanobacteria, was most affected by temperature (and Secchi depth for green algae; see Table G-9 in Appendix G).

However, just as much variability in total algal biomass and diatom biomass was explained by water velocity. Cyanobacteria, diatom, and total algal biomass and density showed little relationship to nitrate and dissolved phosphate concentrations.

Taken as a whole, the trends in water quality constituents and the statistical analyses show that algal biomass and density between sampling events is driven by river velocity and temperature. However, during each sampling event, algal biomass and density in the reference reach and mixing zone was primarily driven by river velocity.

The loss of algal biomass, particularly diatoms, in the San Joaquin River between Vernalis and the DWSC is well documented in the scientific literature (see Section 2.2). Potential loss mechanisms include light limitation, grazing, and settling, all of which are related by a common variable—river discharge. Light limitation controls algal death due to greater respiration than photosynthesis rates. Algal biomass loss due to light limitation in the San Joaquin River is higher downstream of RM 53 (head of Old River), where average channel depth (and thus travel time spent in the nonphotic zone) increases from 5 to approximately 20 ft at the entrance to the DWSC (Litton et al. 2008; Welch and Jacoby 2004). Settling of nonbuoyant algal species during periods of low discharge (e.g., summer low flows) or during slack tide results in algal sedimentation losses, as well as in increased losses attributable to light limitation (Litton et al. 2008; Lehman 2007). When daily river flows were less than 1,800 cfs, Litton et al. (2008), during their three-year study, consistently observed an exponential increase in zooplankton abundance near RM 45, coincident with the region of maximum algal loss. From their field observations and modeling effort, Litton et al. (2008) concluded that net river flows above ~1,800 cfs did not provide sufficient residence time to cause light limitation or to allow development of a flourishing zooplankton community upstream of the DWSC.

Settling, light limitation, and zooplankton grazing are the best explanations for the selective loss of diatoms documented by this study, given that the statistical analysis shows that water velocity was a major factor explaining the variability of total algal biomass and diatom biomass. Net daily river flows during the March, July, and November sampling events were low and provided sufficient residence time for the loss mechanisms to affect the algae community (-146, 176, and 622 cfs, respectively). However, the net daily flow during the May sampling event was substantially higher (1,640 cfs), possibly too high to permit a thriving zooplankton population or to result in light limitation and settling.

Light limitation and zooplankton grazing, in particular, likely influenced the cryptophytes and green algae community, as well. However, these potential loss mechanisms do not necessarily influence cyanobacteria to the same degree as they affect diatoms. Cyanobacteria have the ability to control their buoyancy to avoid light limitation and sedimentation; also, they are nutritionally poor and often difficult for zooplankton to ingest due to their filamentous and colonial structures. Cyanobacteria losses in the study reach were evident from the algal density data, but their losses were not as substantial as the losses of diatoms, whose high nutritional value and unicellular colonality lend to their preferential selection by grazing zooplankton (Welch and Jacoby 2004).

Cyanobacteria with the potential to produce toxins or taste/odor compounds (hereafter, TTO) were identified by comparing taxa enumerated in water samples taken from the San Joaquin River to the comprehensive list of TTO-producing cyanobacteria published by WHO (1999). The identified cyanobacteria included *Anabaena* spp., *Aphanizomenon* spp., *Cylindrospermopsis* spp., *Microcystis* spp., *Oscillatoria* spp., *Planktothrix* spp., *Pseudanabaena* spp., and *Synechocystis* spp. Because phytoplankton was identified to the genus level, it was not possible to fully determine if the enumerated individuals were of strains known to produce TTOs. The abundances of these algae are presented in **Figure 19** for each sampling event in 2012.

The density of potentially harmful algal species was generally greater in samples from the reference reaches than in those from the mixing zone, but the difference was not statistically significant ($\alpha = 0.05$) (**Table 16**). The density of TTO-producing species in March, July, and November was high in the reference sites, decreased to a minimum in the upstream mixing zone, and increased in the vicinity and downstream of the RWCF outfall (Figure 19). In contrast, TTO-producing species density was fairly constant in May, except at US-7R. The trends in March, May, and November were typically driven by *Synechocystis* spp., *Oscillatoria* spp., and *Planktothrix* spp. (the latter two species are of the Oscillatoria family). The density minimum, which typically occurred between US-1 and US-4, was a result of the loss of *Synechocystis* spp., which, unlike the other TTO-producing species, is unicellular, and thus more susceptible to zooplankton grazing. The abundance and diversity of potentially harmful cyanobacteria was greatest in July, and it was in this month that *Microcystis* spp. was present. The greatest *Microcystis* spp. abundance occurred at the reference sites.

For protection from health outcomes not due to cyanotoxin toxicity, but rather to the irritating or allergenic effects of other cyanobacterial compounds, WHO has recommended a threshold of 20,000 cells/mL for water dominated by *Microcystis* spp. and *Planktothrix* spp. (WHO 2003, p. 149). As shown in Figure 19, the total abundance of TTO-producing cyanobacteria did not exceed the 20,000 cells/mL threshold in any of the study samples.

Table 16. Least-squares mean values of the total density of cyanobacteria with the potential to produce toxins or taste/odor compounds.

	Means [a]		P-values [b]	
	Reference Reach	Mixing Zone Reach	Site US-4 Included	Site US-4 Excluded
Density (cells/mL)	8,893	6,068	0.2049	0.0514

[a] Least-squares means were calculated excluding site US-4. See Appendix G for more details.
[b] P-values were generated from a general linear model analysis, and comparison was made between the results obtained when site US-4 was included or excluded from the analysis. See Appendix G for more details.

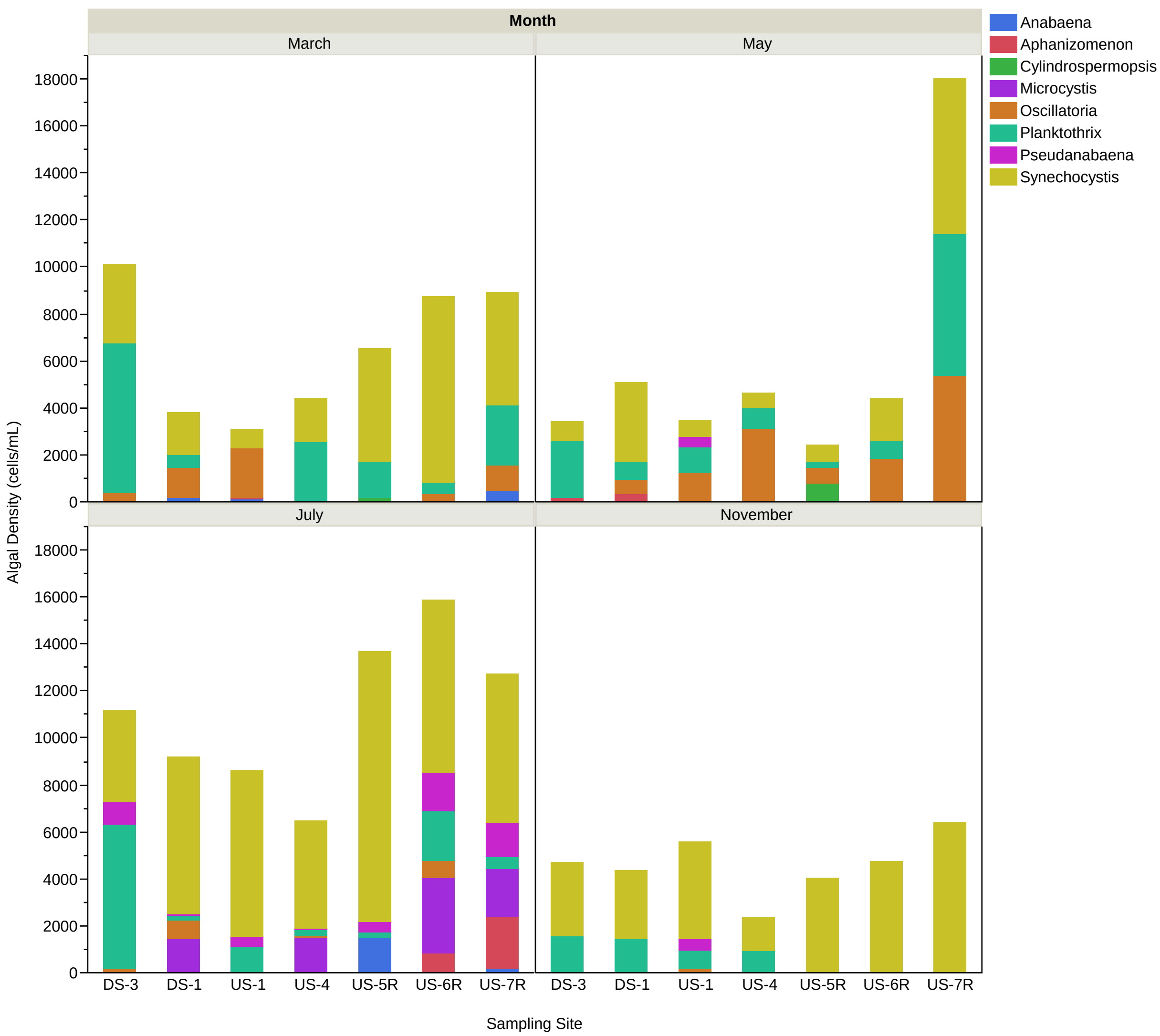


Figure 19. Density of cyanobacteria with the potential to produce toxins or taste/odor compounds. Water samples were taken from the San Joaquin River in the mixing zone and reference reaches on 3/28/2012, 5/23/2012, 7/31/2012, and 11/27/2012.

5 EVALUATION OF MIXING ZONE

As part of its request for dilution credit for nitrate plus nitrite in the RWD for NPDES permit renewal, the City requested dilution credit as summarized in Table 17 below. The associated mixing zone for nitrate plus nitrite would extend 1.4 miles upstream and 1.7 miles downstream from the RWCF outfall (Robertson-Bryan, Inc. 2013). This distance is smaller than the mixing zone granted in the current NPDES permit.

Table 17. Dilution Credit and Associated Effluent Limits Requested in the Report of Waste Discharge (Robertson-Bryan, Inc. 2013).

Season	Estimated Performance-based Limitation (mg/L-N)	Maximum Monthly Average Background Receiving Water Concentration (mg/L-N)	Minimum 30-day Average Dilution Ratio [a]	Dilution Ratio Frequency
1 April–30 September	26	2.2	2.1	≥ 94%
1 October–31 March	30	1.9	2.5	≥ 99%

[a] The minimum dilution ratio is the minimum ratio of receiving water to effluent needed to achieve 10 mg/L-N nitrate concentration in the receiving water at a 55 mgd discharge rate and the specified maximum background receiving water concentration and performance-based effluent concentration.

The SIP lists 11 conditional requirements for mixing zones (see Section 1.2). These conditions are discussed below with regard to the proposed nitrate mixing zone in the San Joaquin River. The conditions are identified by their SIP number in parentheses.

The mixing zone shall not (1) compromise the integrity of the entire water body.

The RWCF discharges into the San Joaquin River, within the Sacramento–San Joaquin Delta. The San Joaquin River is more than 330 miles long, and the Delta is approximately 1,100 square miles and includes approximately 700 miles of channels. The requested mixing zone extends 1.4 miles upstream and 1.7 miles downstream of the discharge, for a total length of 3.1 miles. Thus, the mixing zone represents a small fraction of a percent of the entire water body. Nevertheless, this study has evaluated the potential effects of the mixing zone in the near-field (lower San Joaquin River) and far-field (Delta) water bodies; the results of this evaluation are discussed below.

Near-Field Effects (Lower San Joaquin River)

One possible adverse effect of elevated nitrate concentrations is eutrophication, which can include excessive algae growth and growth of undesirable algal species. However, the RWCF discharge and mixing zone do not stimulate phytoplankton growth relative to reference sites (Table 13, Table 14, Figure 13 through Figure 16), even though nutrient concentrations are greater in the mixing zone compared to the reference sites (Figure 6, Table 10). In fact, the biomass and density of all major algal divisions (green algae, cryptophytes, cyanobacteria, and diatoms) decreased from the reference sites to the mixing zone. The decrease in total algal biomass from the reference sites to the mixing zone is mostly related to the decrease in diatoms, which generally make up the majority of the biomass. Diatom abundance exhibits a statistically significant decrease between the reference sites and mixing zone (Table 14). Because nitrate and diatom abundance both vary over this stretch of the river, a correlation between nitrate and diatom biomass exists (Table G-1 in Appendix G). However, the decrease in diatom abundance has been documented in this reach by other researchers, and similar decreases have been documented in other estuaries. In this reach, the phenomenon has been attributed to hydrodynamics (i.e., decreased velocities leading to increased settling), increased depth of the nonphotic zone, and an increase in zooplankton grazing pressure with increasing distance downstream (Lehman 2007; Litton et al. 2008; see also Section 2.2). The latter two factors may also be responsible for the decrease in other algal divisions from the reference sites to the mixing zone.

The RWCF discharge does not adversely influence phytoplankton species composition. As a fraction of the total algal density, all algal divisions are approximately the same in the reference sites as in the mixing zone (even though the total algal density decreases in the mixing zone; Figure 18). Cyanobacteria make up a larger fraction of the total biomass in the mixing zone than at the reference sites, while diatoms make up a smaller fraction in the mixing zone than at the reference sites (Figure 17). However, because the biomass of all divisions decreases in the mixing zone, this phenomenon is entirely driven by a larger decrease in diatom biomass in the mixing zone (perhaps due to settling) than in cyanobacteria biomass.

Were adverse effects on algae communities (abundance or composition) occurring between the reference sites and the mixing zone, it would be expected that the BMI community would show effects indicative of these changes. As described more fully in Section 4.3.2, the relative proportions of BMI FFGs at all sites upstream and downstream of the RWCF outfall indicated that the trophic structure of the BMI community is heavily dominated by collector-filterers and collector-gatherers, with other FFGs (e.g., scrapers, shredders, and predators) absent or present in very low abundance. Consequently, the trophic structure and integrity of the BMI community at all the reference and mixing zone sites are considered to be altered from a more balanced representation of FFGs. Although the FFG composition indicates that the BMI community has likely altered in response to degraded conditions at all sampling locations, there is no evidence to suggest that the RWCF discharge has caused or observably exacerbated the imbalance in FFG composition. Furthermore, there is no indication that the BMI assemblage is representative of a community in a river system that has been degraded by elevated nutrient loads or eutrophication.

Far-Field Effects (the "Entire Water Body" of the Delta)

Modeling results estimate that RWCF effluent makes up less than 1%, on a long-term average, of water throughout the majority of the Delta. In the south Delta, where the San Joaquin River represents a greater proportion of the flows in the channels, effluent makes up less than 2% of water on a long-term average basis (Table 9).

As explained in detail in Section 4, there is little or no evidence that current nitrate levels are causing or contributing to beneficial use impairment, or that they are compromising the integrity of the Delta. Specific findings of importance include the following:

- There is statistical and historical evidence that algae in the Delta are light limited and not nutrient limited, and thus that modest changes in nitrate levels will have no impact on productivity or chlorophyll *a* levels (Jassby et al. 2002, Jassby 2008).
- *Microcystis* levels cannot be correlated with current levels of nitrate, and have been linked more closely with temperature and residence time (Lehman et al. 2008), as well as with extreme N:P ratios (Glibert et al. 2011). Based on the information available, there is no evidence to indicate that a modest increase in nitrate or shift in the N:P ratio would have any significant effect on *Microcystis* in the Delta.
- Contributions of effluent and effluent-sourced nitrate to Suisun Bay, and by extension to Suisun Marsh, are so small as to be immeasurable. The ongoing TMDL for nutrients in Suisun Marsh is more related to ammonia levels in the Sacramento River and to localized sources than to nitrate from upstream in the Delta.
- The hypothesis in Glibert (2011) is that N:P ratios are currently too high, and that lowering this ratio would lead to ecosystem recovery. However, there is very little evidence that this is the case. Not only are there many other known stressors contributing to the ecosystem changes seen over the last 10 to 30 years that would not be affected by a change in the N:P ratio, but it is not even certain that one of the stressors is the N:P ratio at all (and, by extension, that nitrate is at all responsible or involved). A more likely stressor may be ammonia levels (which are almost totally the driver of the N:P ratio trends examined in the Glibert analysis).

The mixing zone shall not (2) cause acutely toxic conditions to aquatic life passing through the mixing zone.

The most sensitive endpoints (LC50s[1]) for acute nitrate toxicity to aquatic organisms are nearly 10 times greater than the highest nitrate concentrations measured in this study. The most

[1] LC50 = The lethal concentration of a substance that kills 50% of test organisms in a given period.

sensitive organisms appear to be aquatic invertebrates, with the lowest LC50 assigned to the amphipod *Echinogammarus echinosetosus*, which has a 120-hour LC50 of 56.2 mg/L-N (Camargo et al. 2005). The most sensitive fish endpoint was for Siberian sturgeon (*Acipenser baeri*), which has a 96-hour LC50 of 397 mg/L-N (Hamlin 2006). Because there are no regulatory objectives or recommended water quality criteria for the protection of aquatic life from the toxic effects of nitrate, and because nitrate is not toxic to aquatic life at the levels that occur in the effluent or in the receiving waters, this study concludes that the mixing zone would not cause acutely toxic conditions to aquatic life living in or passing through the mixing zone.

The mixing zone shall not (3) restrict the passage of aquatic life.

The only feasible manner in which the mixing zone could restrict the passage of aquatic life would be if excessive submerged and emergent vegetation growth was present, physically restricting the cross section of the channel. However, as described in Section 4.3.3, any influence from the RWCF discharge on submerged and emergent vegetation is minimal, and the vegetation that is present does not restrict the passage of aquatic life (Table 12).

The mixing zone shall not (4) adversely impact biologically sensitive or critical habitats, including, but not limited to, habitat of species listed under federal or state endangered species laws.

The fish species in the Delta listed as threatened or endangered under the state and federal Endangered Species Acts are steelhead, winter and spring-run Chinook salmon, longfin smelt, delta smelt, and green sturgeon. Critical habitat for all of these species exists in the Delta. In the mixing zone itself, critical habitat has been designated for only steelhead and delta smelt. As mentioned above, nitrate never reaches toxic levels in the mixing zone or at any other location upstream or downstream of the discharge. Critical habitats are not adversely affected by nitrate levels because, as summarized for SIP requirement 1 above, effects of the mixing zone on primary and secondary production abundance and composition have not been observed and are not expected to occur.

The mixing zone shall not (5) produce undesirable or nuisance aquatic life.

Undesirable and nuisance aquatic life that could be of concern in the vicinity of the study area include undesirable algal species, such as *Microcystis*, and other nuisance vegetative growth. Nuisance growth of submerged and emergent vegetation is discussed in regard to SIP requirement 3, above.

As described in Section 4.3.4, nuisance algal species that produce microcystins were present only in July, and those producing taste and odor compounds were present throughout the year (Figure 19). Although detected in July water samples, *Microcystis* abundance in the mixing zone was not significantly greater than at the reference sites (Table 14). Levels of detected nuisance species did not reach thresholds at which the species would cause or contribute to nuisance.

As described in Section 2.5, when waters of the Delta are exported into relatively shallow conveyance canals, algae may no longer be light limited. However, considering ambient nitrate concentrations in the Delta, the RWCF's incremental nitrate contribution at drinking water intakes at these conveyances (Table 9) is not sufficient to result in any substantial difference in algal biovolume or aquatic plant density (see Section 2.5).

The mixing zone for nitrate shall not (6) result in floating debris, oil, or scum;

The only significant floating debris witnessed during the field investigations was isolated floating water hyacinth, observed during the September 26, 2012, sampling event. However, these plants were documented to occur throughout the reference sites and mixing zone. There were no indications that the presence of floating water hyacinth resulted from the RWCF discharge or mixing zone.

The mixing zone for nitrate shall not (7) produce objectionable color, odor, taste, or turbidity.

The applicability of this requirement to the requested mixing zone relates to algal species that produce taste and odor compounds, in both the near-field and the far-field. Regarding the near-field, as described in Section 4.3.4, algal species that produce taste and odor compounds were present throughout the year (Figure 19); however, these species did not reach levels that would cause or contribute to taste and odor problems. Regarding the far-field, as discussed previously, the fraction of effluent at drinking water intakes is very low; when considering ambient nitrate concentrations in the Delta, the RWCF's incremental nitrate contribution at drinking water intakes (Table 9) is not sufficient to result in any substantial difference in algal biovolume (see Section 2.5). Further, it is currently not possible to relate the growth of attached or benthic algae (which may be important in the development of taste and odor compounds) to nutrients in Delta waterways or conveyance canals.

The mixing zone shall not (8) cause objectionable bottom deposits.

As mentioned previously, the mixing zone does not stimulate the growth of phytoplankton. Additionally, light does not penetrate to the bottom of the channel (i.e., the Secchi depths recorded are less than the river depths), so there is little to no benthic algae, as evidenced by the lack of scrapers in the BMI community. Therefore, the mixing zone does not cause objectionable bottom deposits.

The mixing zone shall not (9) cause nuisance.

The mixing zone does not cause a nuisance, as described for SIP requirements 5–8 above.

The mixing zone shall not (10) dominate the receiving water body or overlap a mixing zone from different outfalls.

As described above, the mixing zone represents only a fraction of a percent of the entire water body. RWCF effluent makes up less than 1% of the water, on a long-term average, throughout the majority of the Delta. Finally, the requested mixing zone does not overlap any other mixing zones.

The mixing zone shall not (11) be allowed at or near any drinking water intake.

No drinking water intakes are currently located in the immediate vicinity of the RWCF mixing zone. The nearest drinking water intake is more than 10 miles downstream of the discharge and mixing zone.

6 CONCLUSION

This study finds that the existing effluent quality and in-river conditions with regards to nitrate plus nitrite levels are consistent with the 11 requirements of the SIP. Hence, the granting of the requested dilution credit and associated mixing zone in the renewed NPDES permit, that would allow existing conditions to continue, would be consistent with the SIP and protective of receiving water beneficial uses.

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Appendix A

Quality Assurance and Quality Control

QUALITY ASSURANCE AND QUALITY CONTROL

Sample collection, sample preparation, and sample analysis will undergo rigorous quality assurance and quality control. Specific quality assurance procedures for tasks outlined in this Work Plan are described below.

Sample Collection

Collection, handling, and transport of sediment samples will follow standard operating procedures for the collection of grab samples. These procedures include the use of clean sampling equipment. Samples will be collected on the upstream side of the boat away from the motor and, if possible, outboard motors will be shut off during sampling.

Nitrate plus nitrite, total nitrogen, phosphorous, dissolved ortho-phosphate, and algae samples will be capped and stored in chilled coolers for transport to the analytical laboratory for processing. At the time of sample collection, typical field conditions will be logged, including start and end time of sampling, tidal stage, direction of flow, and GPS positioning of sample location. Multiple grab samples may be required to obtain representative samples and complete volumes. All algae, water, and BMI samples will be transported and delivered under proper chain of custody.

A Benthic Macroinvertebrate Field Data Sheet (Barbour et al. 1999) will be completed for each transect. The Ponar dredge will be cleansed of any debris and/or organisms by repeatedly rinsing the dredge in the surface water and removing any clinging organisms by hand nozzle prior to sampling the subsequent transect. Upon retrieval of BMI samples, the dredge will be examined for the following criteria:

- Complete closure of the grab jaws,
- No evidence of sediment washout through the grab doors,
- An even distribution of sediment in the grab,
- Minimum disturbance of the sediment surface,
- Minimum overall sediment depth appropriate for the sediment type:
- 4 cm in coarse sands and gravel;
- 5 cm in medium sands;
- 7 cm in fine sands; and
- 10 cm in silty sands, silts, and clay.

If these criteria are met, the sample will be retained. If not, the dredge contents will be discarded and another sample will be taken at an adjacent location within a few feet of the discarded sample. Copies of all field data sheets, databases, and summary reports will be retained by RBI. RBI will maintain all data in a comprehensive database.

Sample Analyses

All phytoplankton identifications and biovolume measurements will be conducted by a qualified phycologist. Outside taxonomists will be utilized for taxonomic verifications when necessary. All samples are initially test mounted for counting density before final mounting. Any major questionable taxonomic identifications are noted in the database during counting, and indicated on the report as uncertain for taxonomic clarity. If sufficient sample is available, samples will sent out to other taxonomists for taxonomic confirmation. All biovolume calculations will be verified by comparing with current scientific literature, and, if necessary, with outside consultants.

Standard analytical method quality assurance protocols will be employed in nitrate plus nitrite sample analysis. Depending on the method, internal quality control checks may include method blanks, matrix spike and spike duplicates, surrogates, and sample duplicates. Standard quality control objectives for precision and accuracy will be utilized. All analytical reporting will undergo quality assurance and quality control verification by laboratory staff, as well as by the preparers of this SAP. For BMI samples, CSBP quality assurance protocols will be performed on a minimum of 10% of the samples to ensure a 90% removal rate of organisms.

Appendix B

DSM2 Near Field Simulation Results

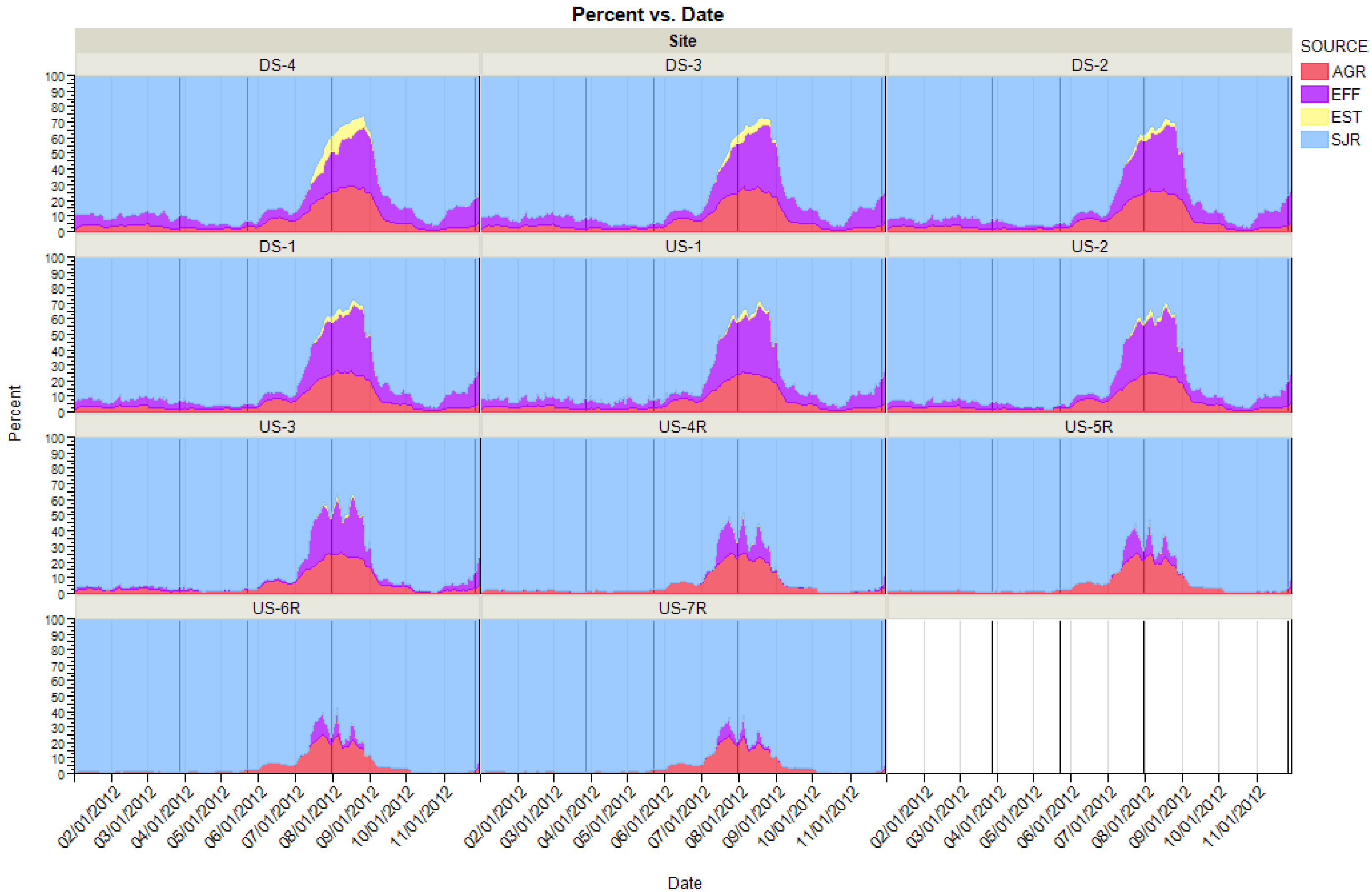


Figure B-1. Source water fraction time series for January through November, 2012, for the eleven sample sites. Darker vertical lines represent days of algae/nutrient sampling.

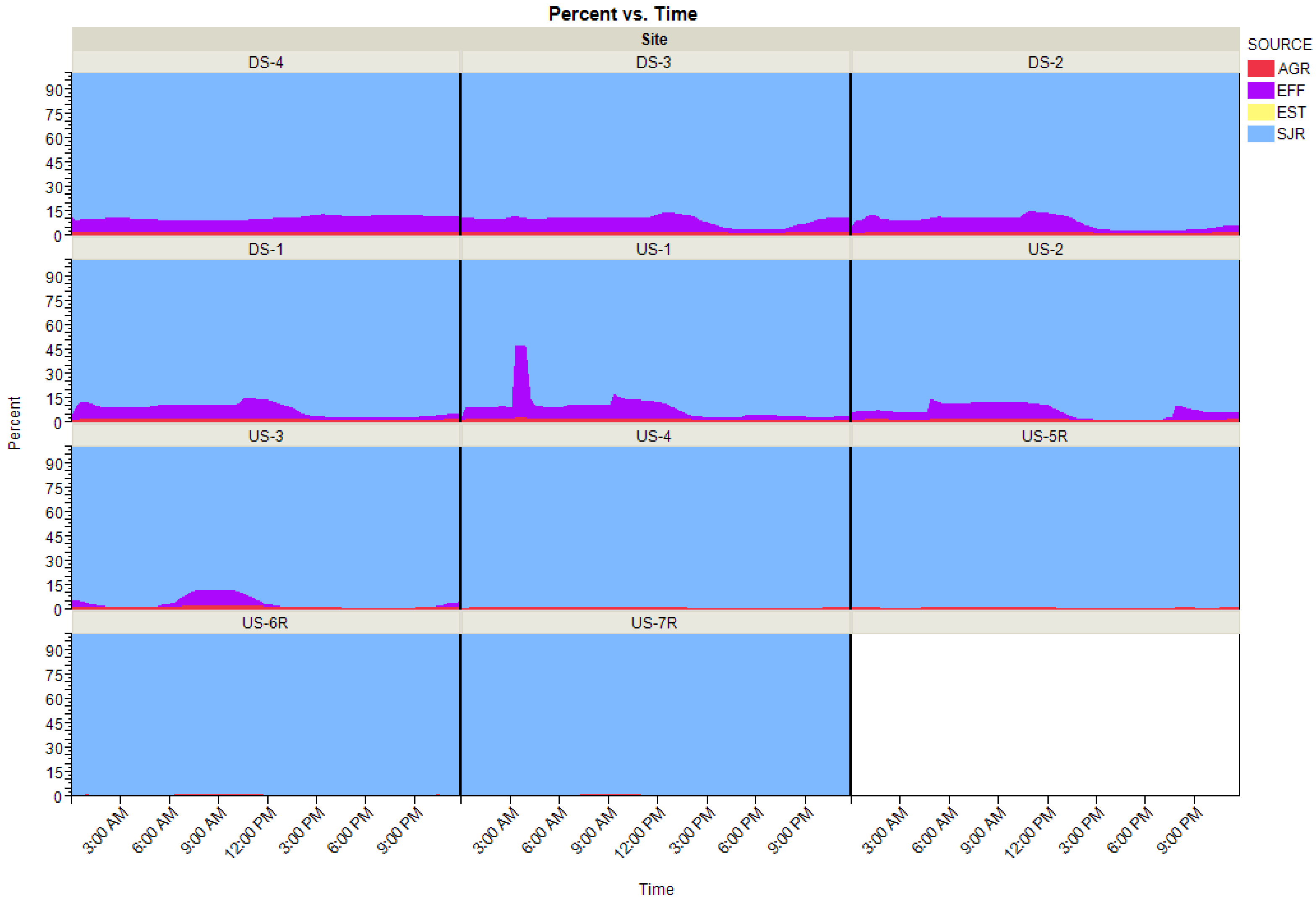


Figure B-2. Source water fraction vs. time of day for the eleven sample sites for March 28, 2012.

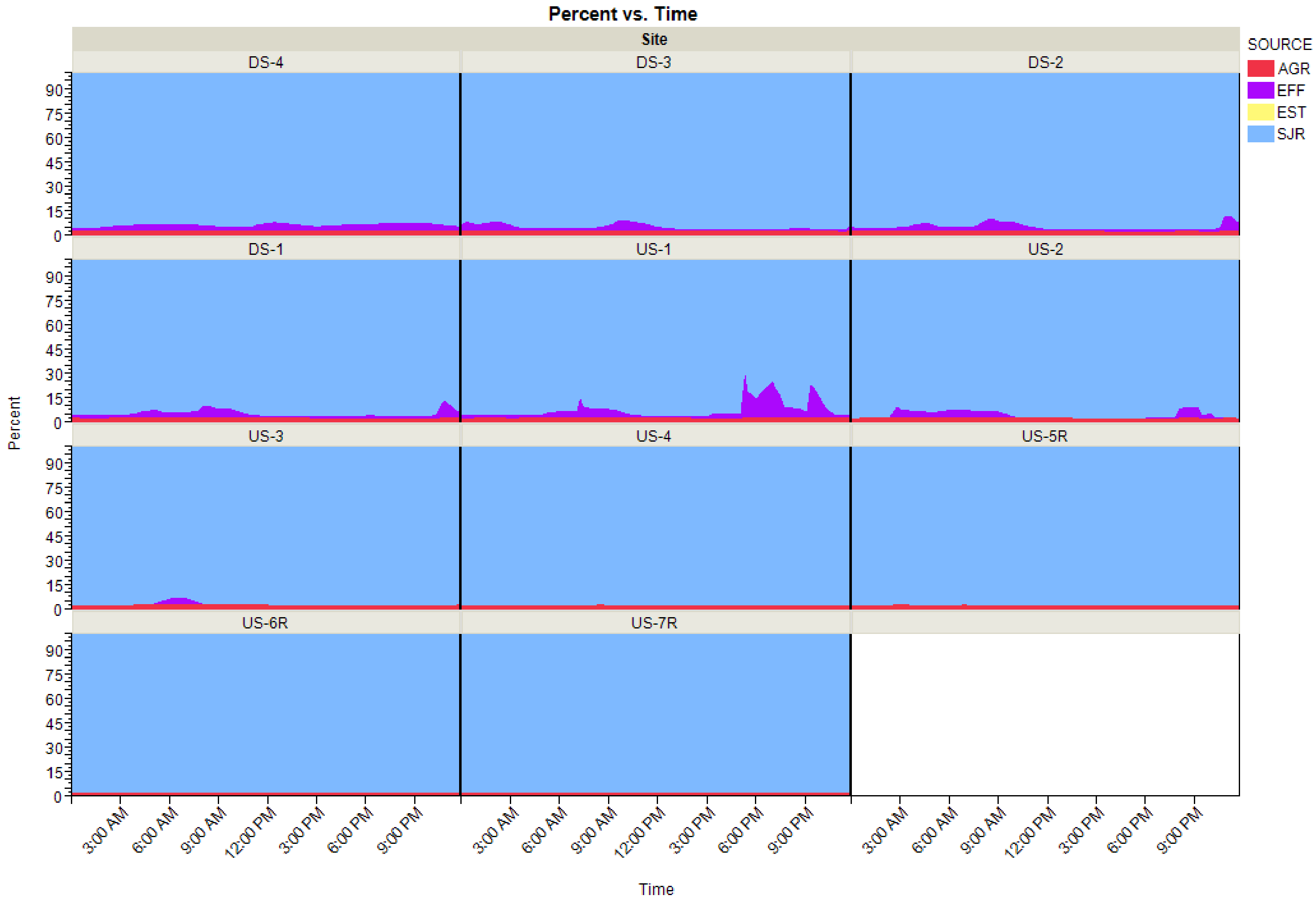


Figure B-3. Source water fraction vs. time of day for the eleven sample sites for May 23, 2012.

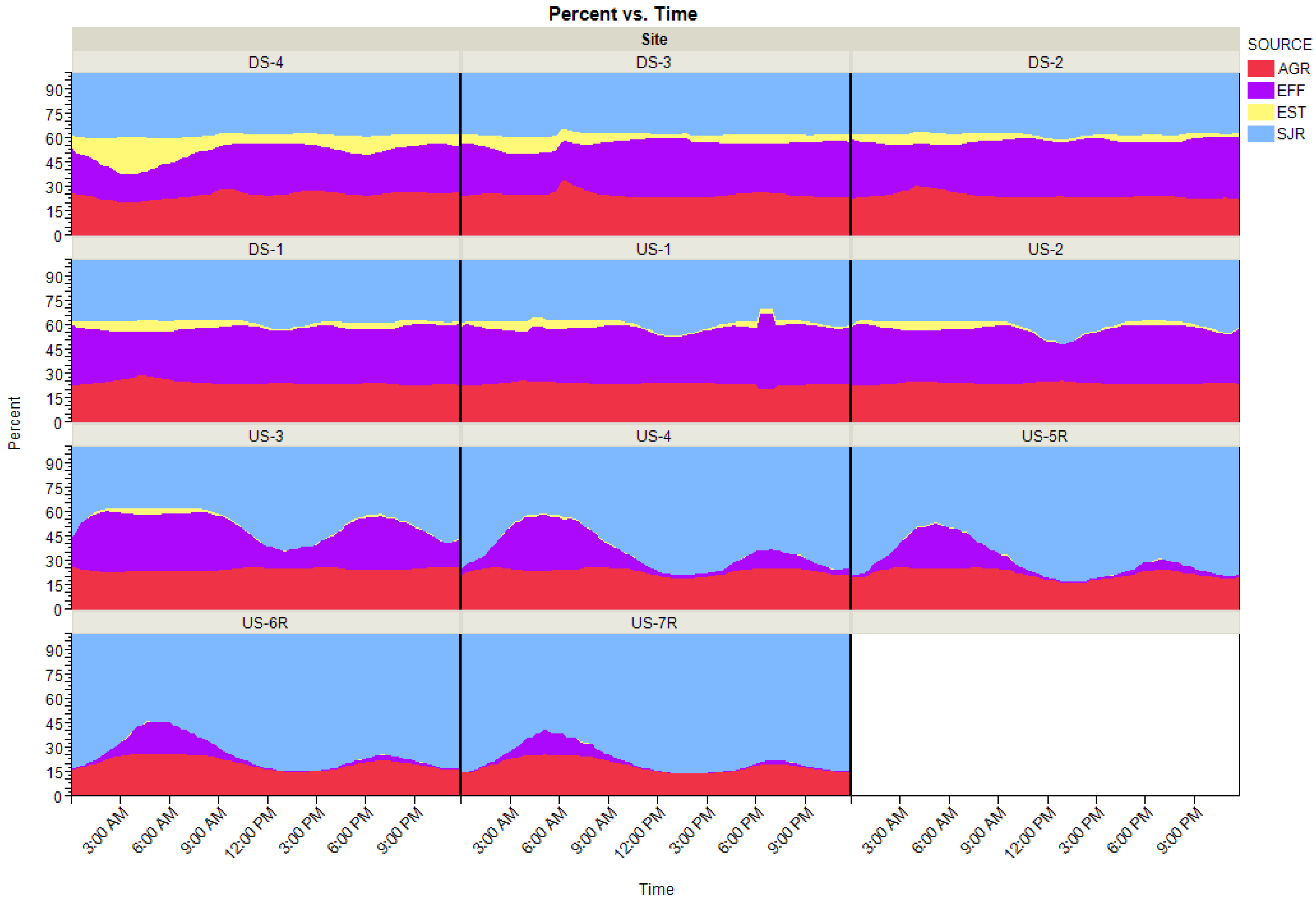


Figure B-4. Source water fraction vs. time of day for the eleven sample sites for July 31, 2012.

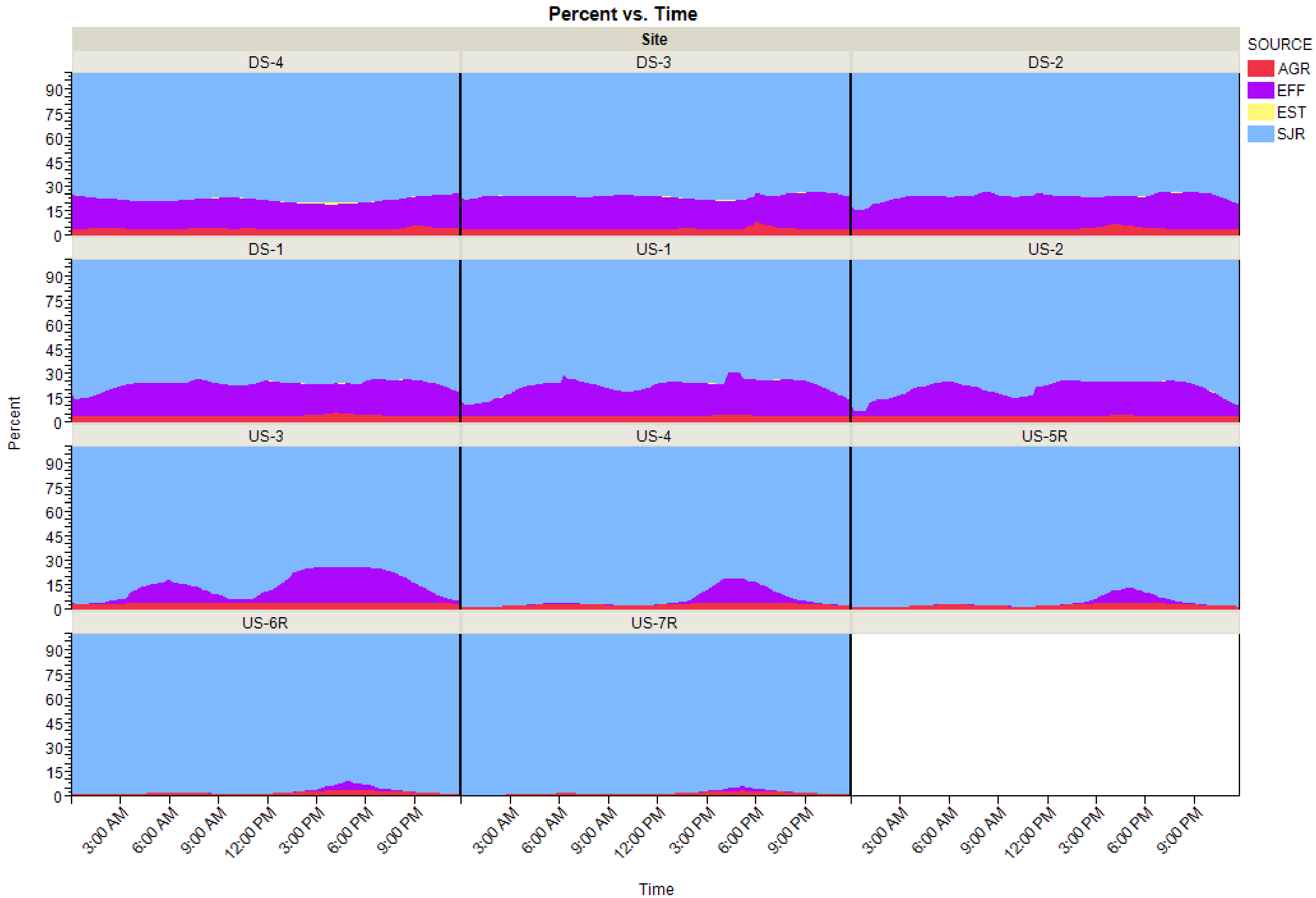
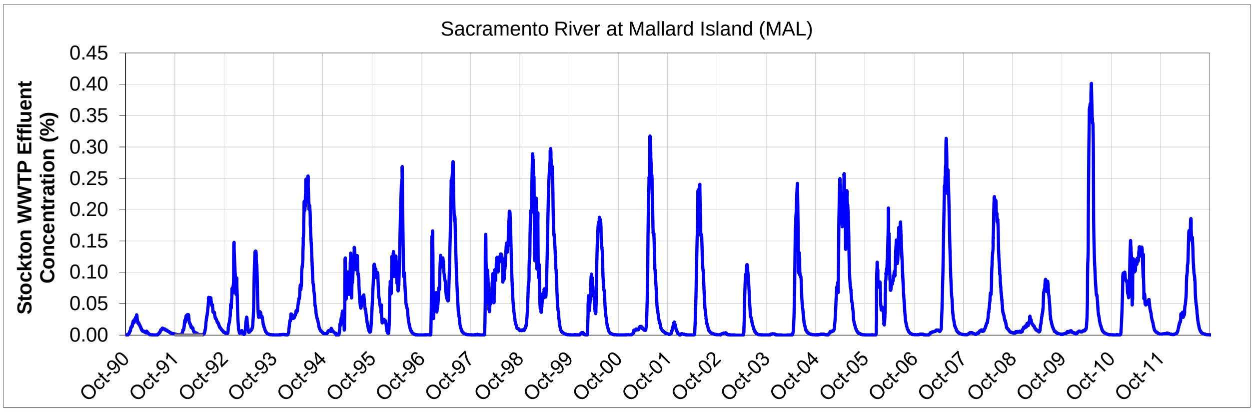


Figure B-5. Source water fraction vs. time of day for the eleven sample sites for November 27, 2012.

Appendix C

DSM2 Results: Effluent Fractions at Far-Field Delta Sites



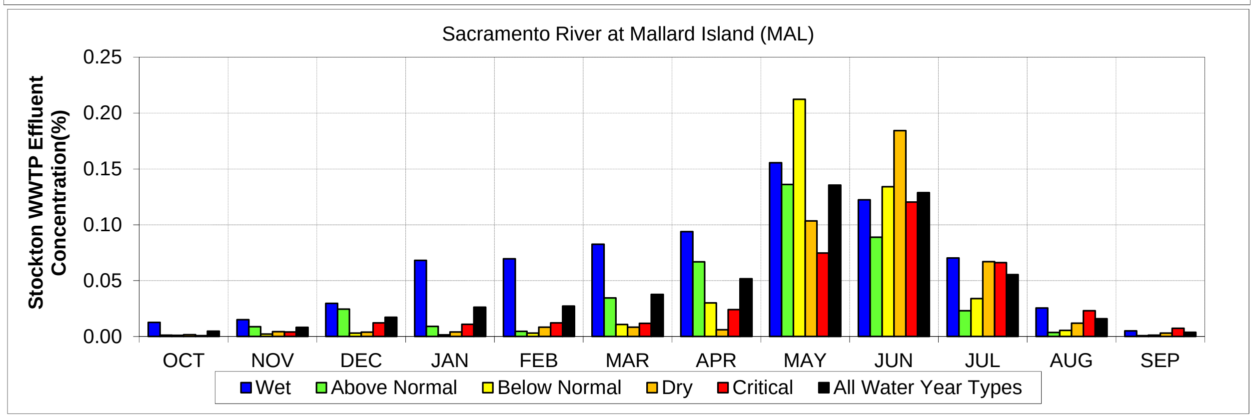
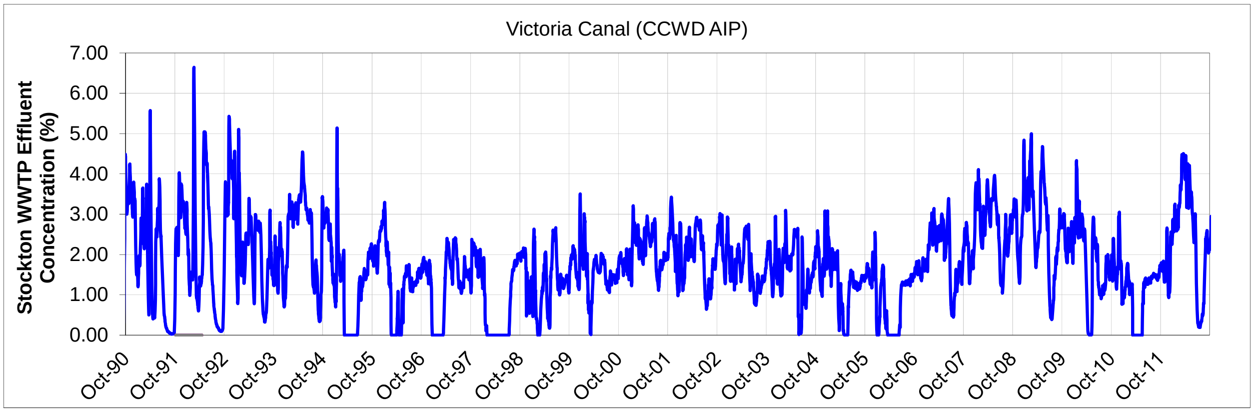


Figure C-1. Time-series and monthly average Stockton RWCF effluent concentrations for Sac. River at Mallard Island.



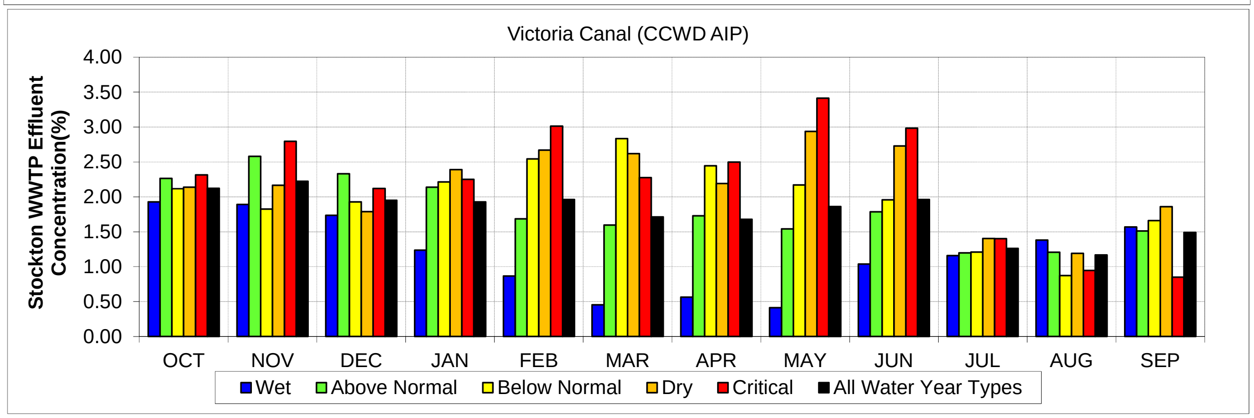
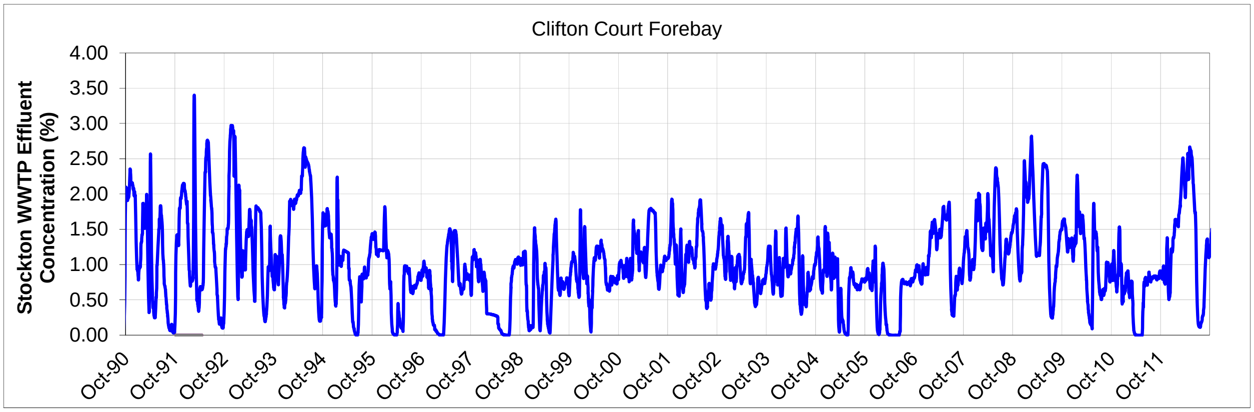


Figure C-2. Time-series and monthly average Stockton RWCF effluent concentrations for CCWD Victoria Canal Intake.



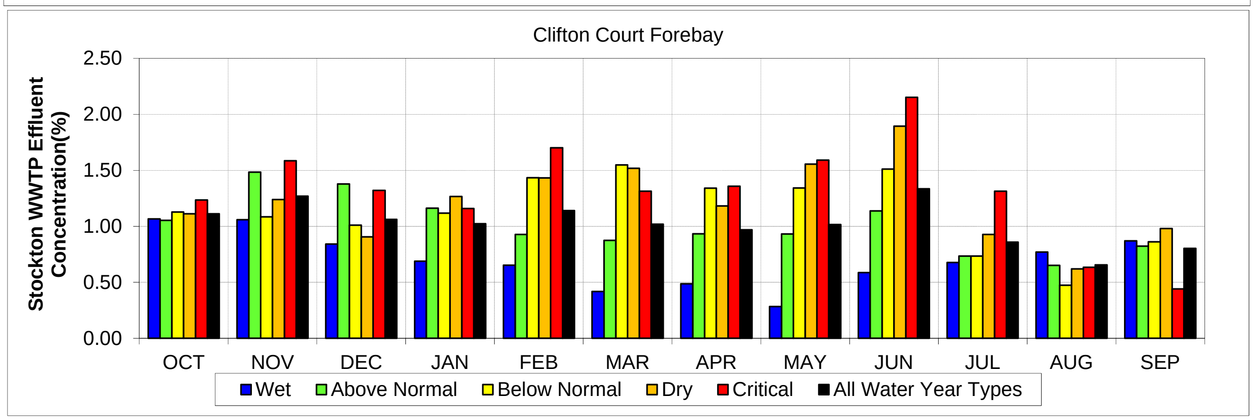


Figure C-3. Time-series and monthly average Stockton RWCF effluent concentrations for Clifton Court Forebay.

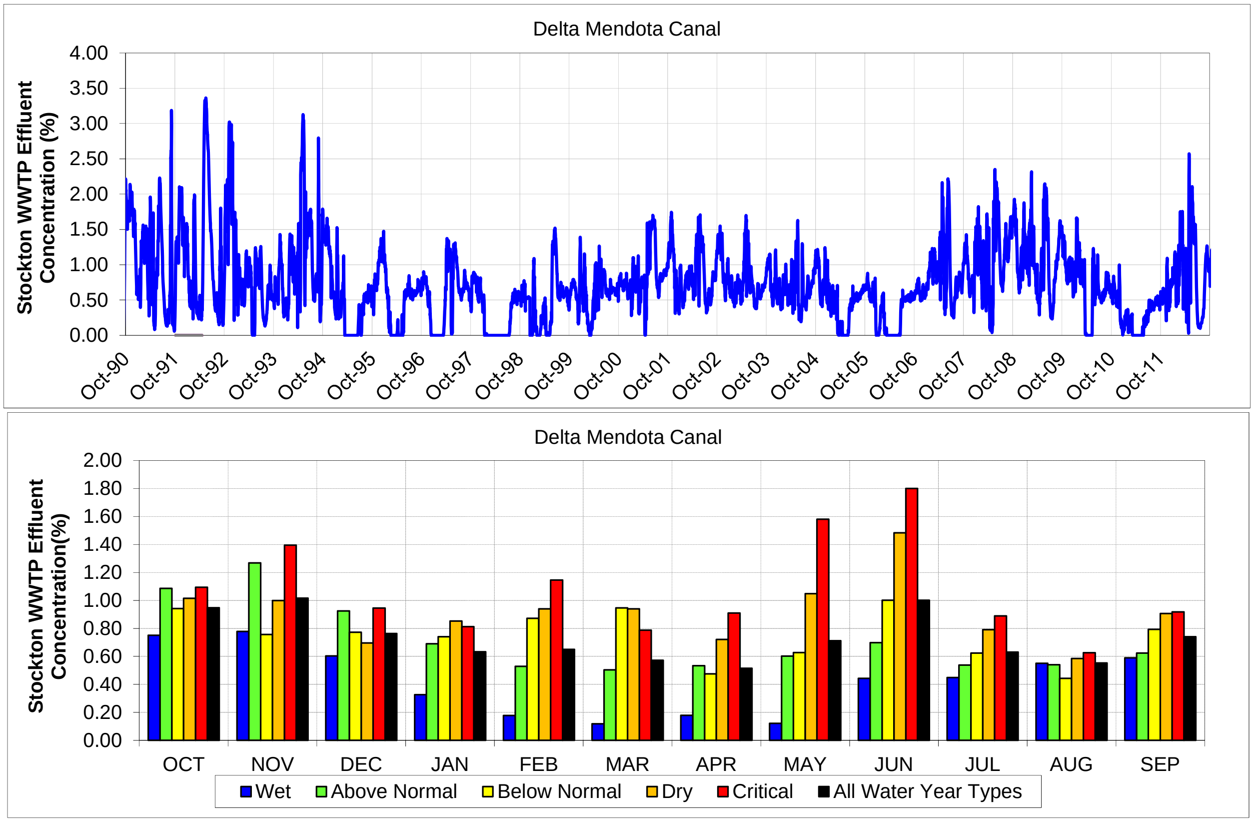


Figure C-4. Time-series and monthly average Stockton RWCF effluent concentrations for Delta Mendota Canal.

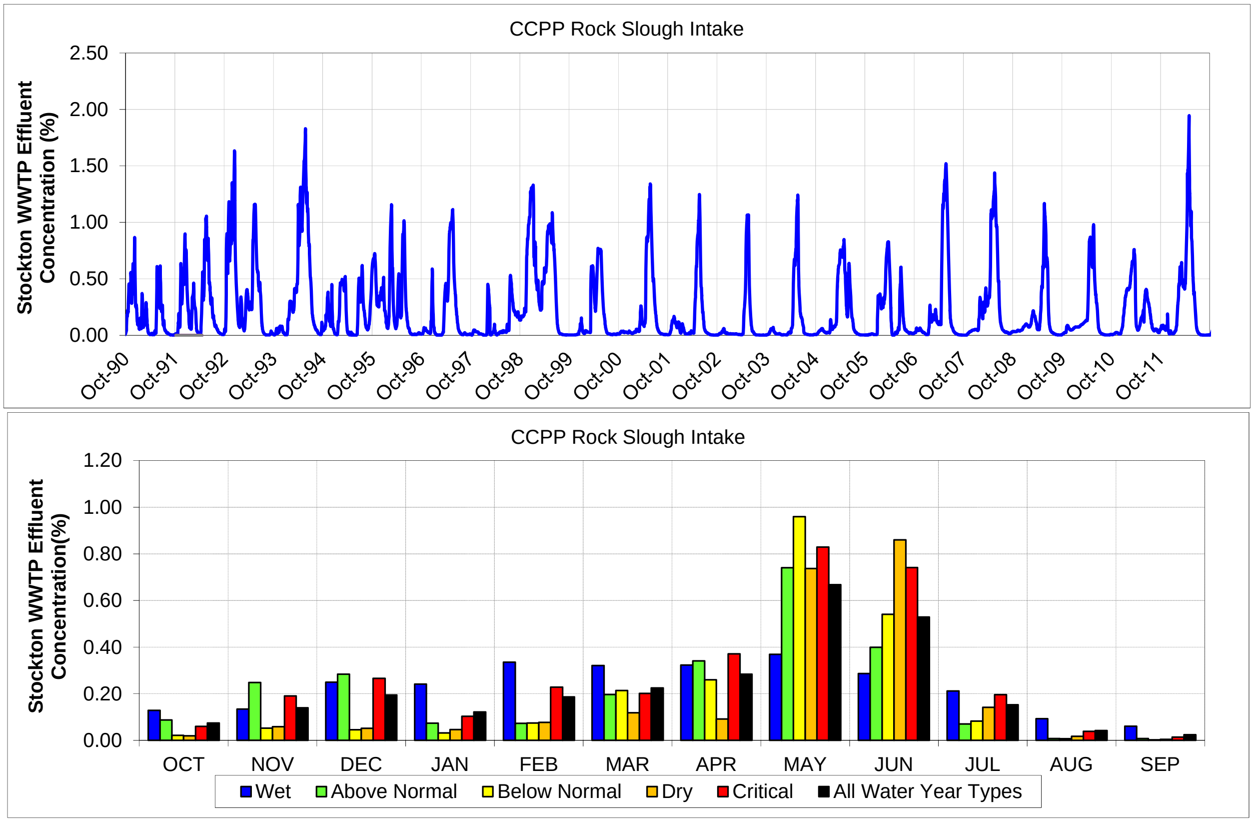


Figure C-5. Time-series and monthly average Stockton RWCF effluent concentrations for CCWD Intake at Rock Slough.

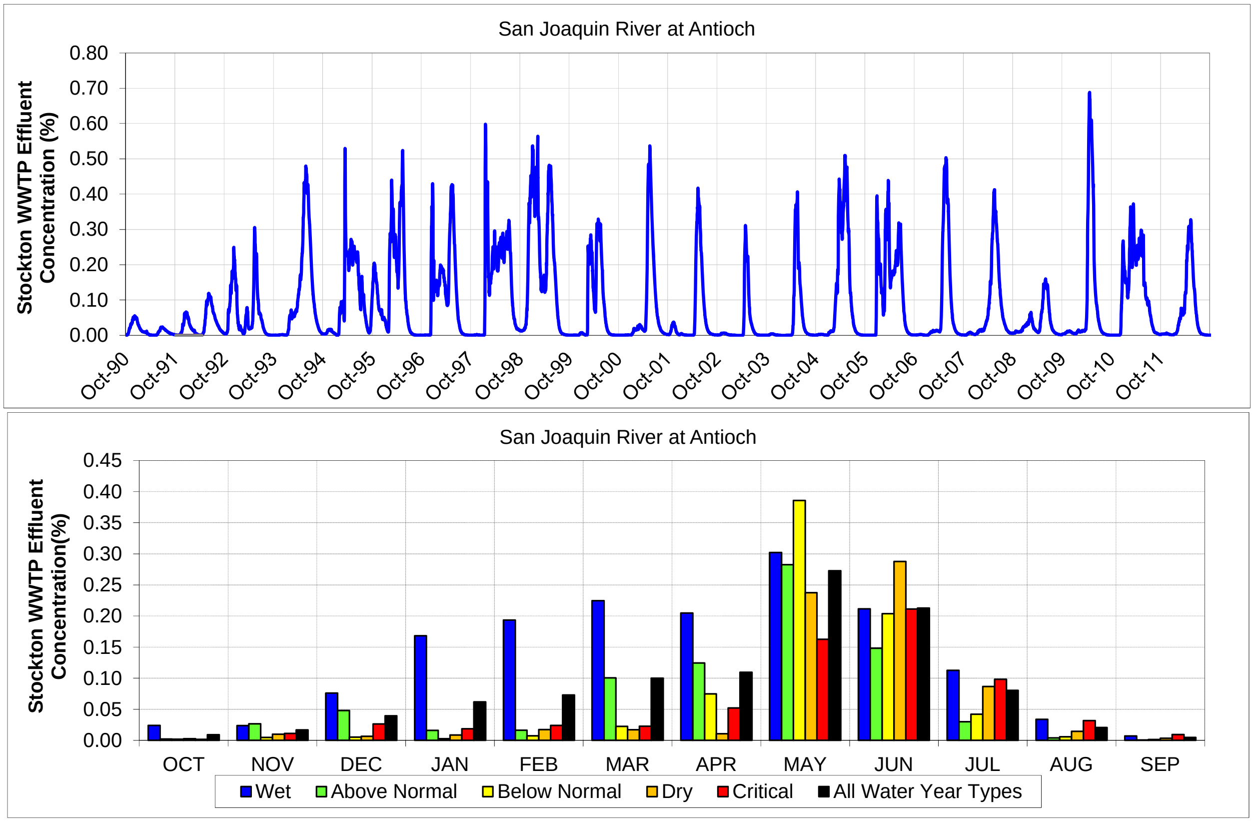


Figure C-6. Time-series and monthly average Stockton RWCF effluent concentrations for San Joaquin River at Antioch.

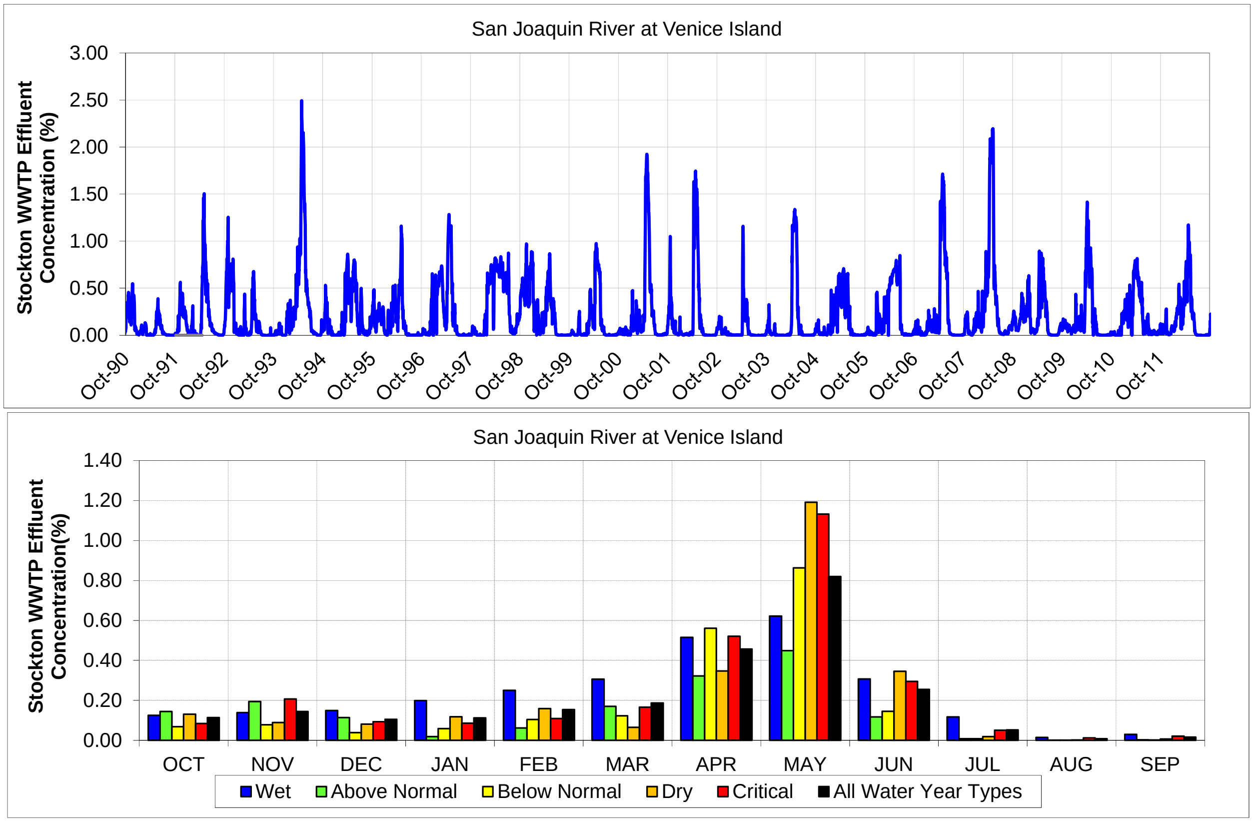


Figure C-7. Time-series and monthly average Stockton RWCF effluent concentrations for San Joaquin River at Venice Island.

Appendix D

Tables and Figures of Physical, Chemical, and Algae Measurements

Table D-1. Temperature, dissolved oxygen, electrical conductivity, and Secchi depth measured upstream and downstream of the RWCF outfall in the San Joaquin River from March 2012 through January 2013.

		Sampling Site (River Mile)										
Constituent	Month	DS-4 (39)	DS-3 (40)	DS-2 (40.75)	DS-1 (41)	US-1 (41.5)	US-2 (41.75)	US-3 (43)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
Temperature (°C)	March	14.6	14.6	14.7	14.7	14.8	15.0	15.3	15.5	15.4	15.3	15.3
	May	20.7	20.5	20.3	20.3	20.2	20.2	19.9	19.9	19.7	19.7	19.5
	July	25.9	25.7	25.8	25.8	26.0	26.1	26.3	26.5	26.5	26.8	26.7
	September	22.6	21.8	21.8	21.7	21.7	21.9	22.2	22.3	22.2	22.2	22.0
	November	14.6	14.4	14.3	14.3	14.2	14.1	13.9	13.5	13.5	13.4	13.4
	January	9.9	10.2	10.2	10.3	10.3	10.5	10.8	10.7	10.7	11.1	10.7
Dissolved Oxygen (mg/L)	March	8.32	8.06	8.74	7.94	8.22	8.47	9.23	9.48	9.65	9.64	9.72
	September	7.35	8.18	8.56	9.03	9.27	9.97	9.64	9.95	10.79	11.31	8.82
	November	8.28	8.55	8.30	8.25	7.96	8.16	8.31	8.51	8.46	8.37	8.91
	November	8.01	7.99	7.88	8.45	8.34	8.32	8.68	8.04	8.88	8.40	9.62
Electrical Conductivity (µS/cm)	March	991	990	960	955	938	916	986	1011	1002	995	988
	May	435	381	384	380	369	368	371	371	373	374	372
	July	727	672	681	641	588	556	569	646	668	672	668
	September	711	649	620	618	606	635	777	774	766	754	739
	November	788	808	809	810	809	809	807	784	782	781	779
	November	1001	1016	1017	1015	1019	1031		1054	1063	1066	1067
Secchi Depth (ft)	March	4.5	4.0	4.5	4.0	3.5	3.0	2.5	3.0	2.5	2.5	2.5
	May	1.9	1.2	1.2	1.3	1.3	1.2	2.0	1.8	1.9	1.5	1.8
	July	3.0	2.8	2.5	2.3	2.3	2.4	1.5	2.0	2.0	2.0	1.8
	September	3.6	3.1	3.0	2.2	2.2	2.3	1.9	2.1	1.9	1.8	1.9
	November	4.5	5.7	5.5	5.5	5.5	6.0	6.5	6.0	6.5	6.3	6.5
	November	4.8	6.0	5.8	6.0	5.0	5.0	5.0	5.0	5.3	5.3	5.0

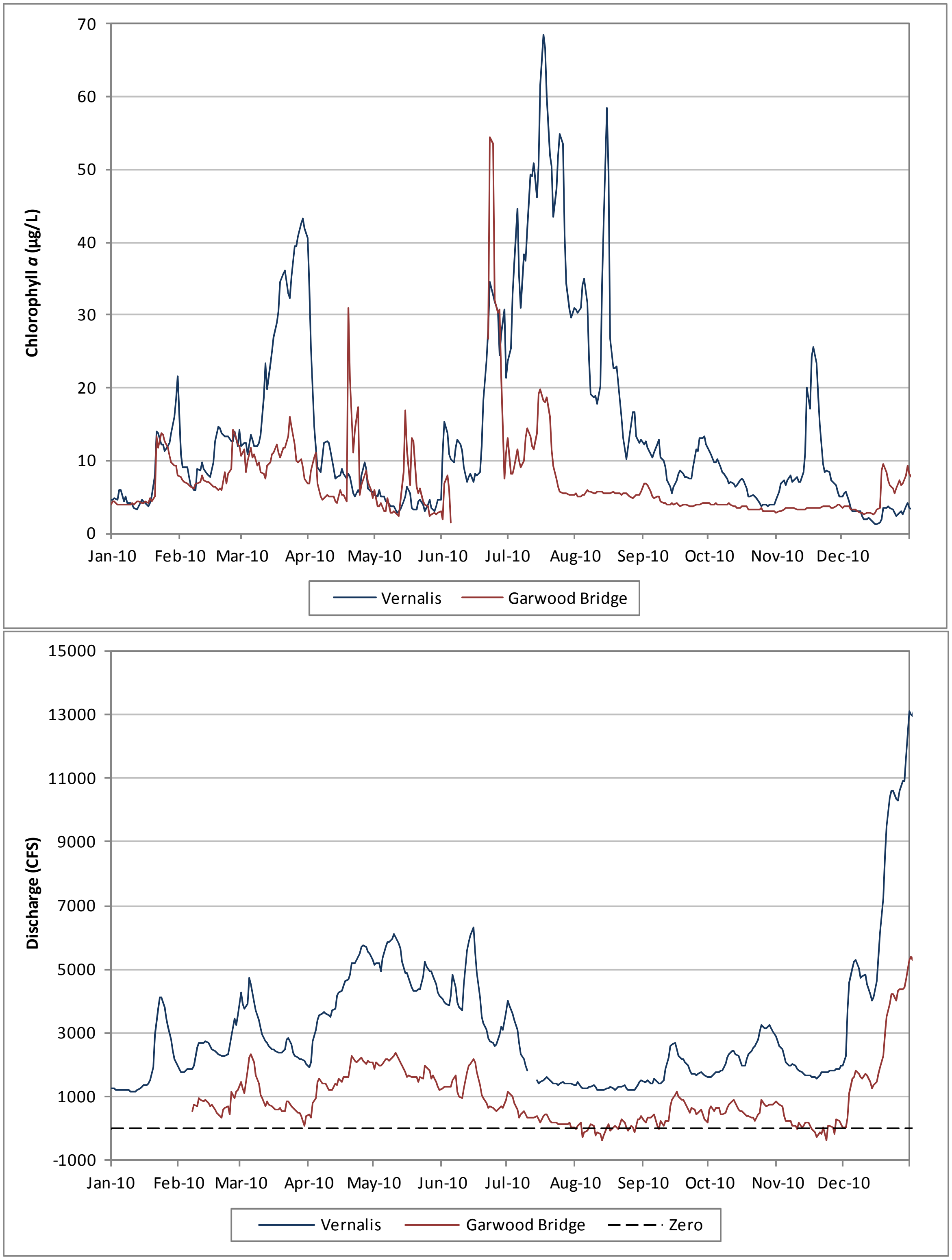
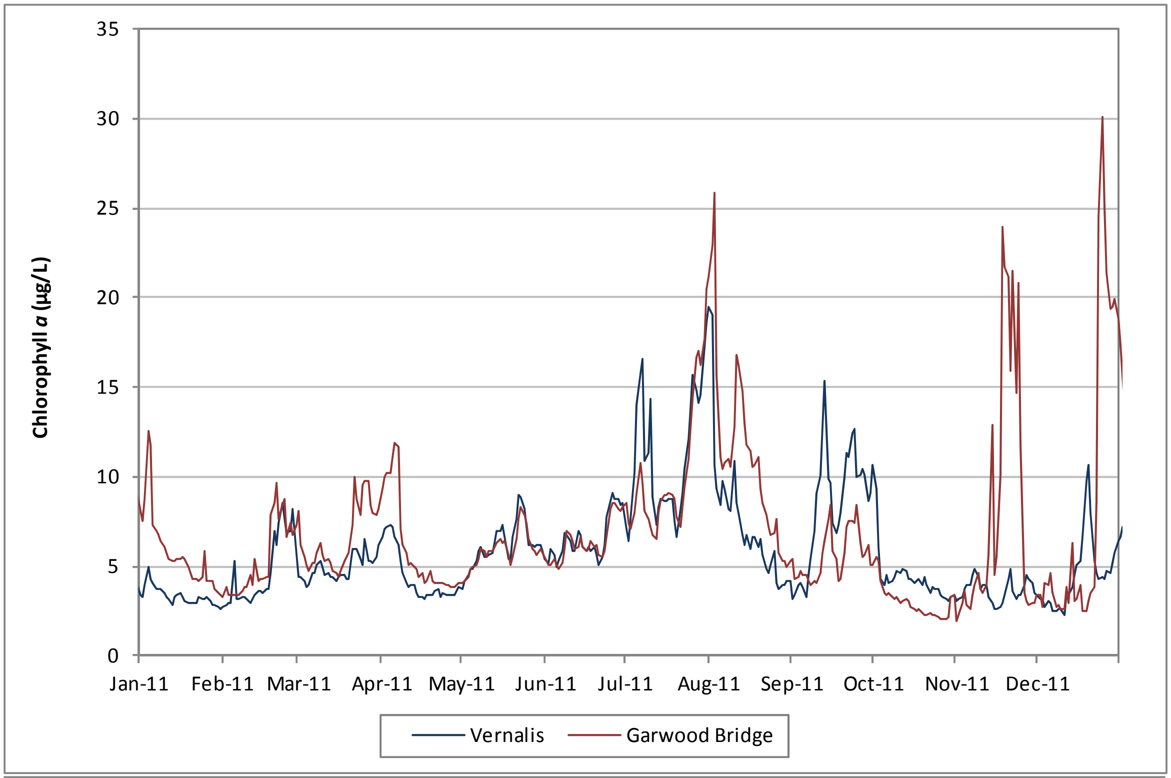


Figure D-1. Chlorophyll *a* (top) and discharge (bottom) measured for the San Joaquin River at Vernalis and Garwood Bridge for January 2010 through December 2010. Data were compiled from CDEC.



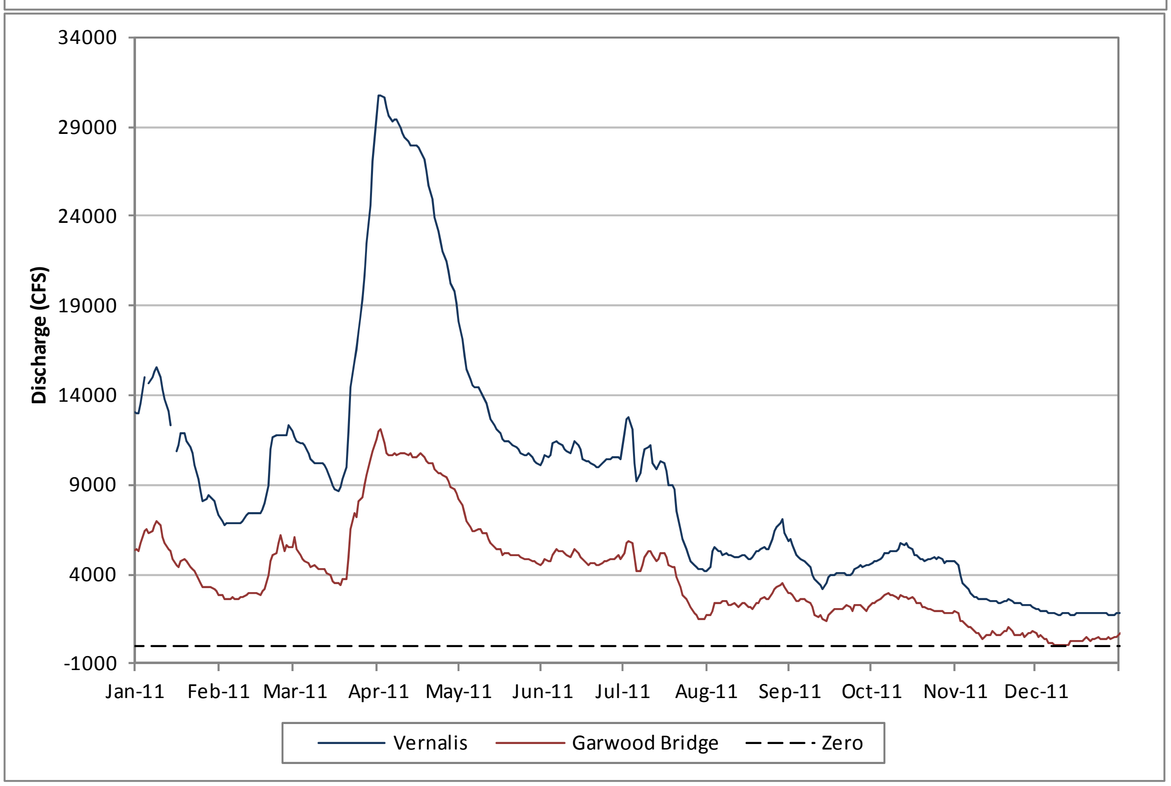


Figure D-2. Chlorophyll *a* (top) and discharge (bottom) measured for the San Joaquin River at Vernalis and Garwood Bridge for January 2011 through December 2011. Data were compiled from CDEC.

Table D-2. Nitrate+nitrite (measured and modeled), total Kjeldahl nitrogen (measured), and total-nitrogen (measured) concentrations in water samples taken upstream and downstream of the RWCF outfall in the San Joaquin River from March 2012 through November 2012.

Constituent	Month	Sampling Site (River Mile)						
		DS-3 (40)	DS-1 (41)	US-1 (41.5)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
Nitrate+Nitrite (mg/L-N)	March	4.00	4.50	3.10	1.70	1.70	1.70	1.70
	May	0.93	0.84	0.80	0.78	0.78	0.79	0.79
	July	2.50	2.50	2.70	2.70	2.40	2.30	2.10
	November	5.60	5.70	5.80	4.00	3.20	2.90	2.50
Nitrate+Nitrite Modeled (mg/L-N)	March	3.95	3.21	2.57	1.71	1.71	1.71	1.71
	May	0.97	0.98	0.98	0.81	0.81	0.81	0.81
	July	5.07	4.89	4.41	2.12	2.05	2.03	2.02
	November	6.27	6.38	6.00	2.49	2.47	2.46	2.46
Total Kjeldahl Nitrogen (mg/L-N)	March	0.64	0.68	0.57	0.53	0.56	0.42	0.60
	May	0.46	0.44	0.47	0.44	0.43	0.49	0.48
	July	0.55	0.60	0.59	0.67	0.68	0.75	0.82
	November	0.52	0.57	0.47	0.46	0.59	0.43	0.35
Total Nitrogen (mg/L-N)	March	4.6	5.2	3.7	2.2	2.3	2.1	2.3
	May	1.4	1.2	1.3	1.2	1.2	1.3	1.3
	July	3.1	3.1	3.3	3.4	3.1	3.1	2.9
	November	6.1	6.3	6.3	4.5	3.8	3.3	2.9

Table D-3. Dissolved-phosphate and total-phosphorus concentrations, and N:P (atoms N/atoms P) measured in water samples taken upstream and downstream of the RWCF outfall in the San Joaquin River from March 2012 through November 2012.

Constituent	Month	Sampling Site (River Mile)						
		DS-3 (40)	DS-1 (41)	US-1 (41.5)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
Dissolved Phosphate (mg/L-P)	March	0.21	0.25	0.19	0.12	0.11	0.11	0.10
	May	ND	ND	ND	ND	ND	ND	ND
	July	0.42	0.39	0.42	0.28	0.24	0.23	0.20
	November	0.46	0.42	0.46	0.32	0.27	0.23	0.20
Total Phosphorus (mg/L-P)	March	0.28	0.29	0.25	0.19	0.19	0.18	0.20
	May	0.15	0.14	0.15	0.12	0.11	0.15	0.13
	July	0.52	0.53	0.56	0.45	0.35	0.36	0.38
	November	0.54	0.52	0.55	0.38	0.30	0.28	0.24
Total Nitrogen / Total Phosphorus (atoms N / atoms P)	March	36	40	33	26	27	26	25
	May	21	19	19	22	24	19	22
	July	13	13	13	17	20	19	17
	November	25	27	25	26	28	26	27

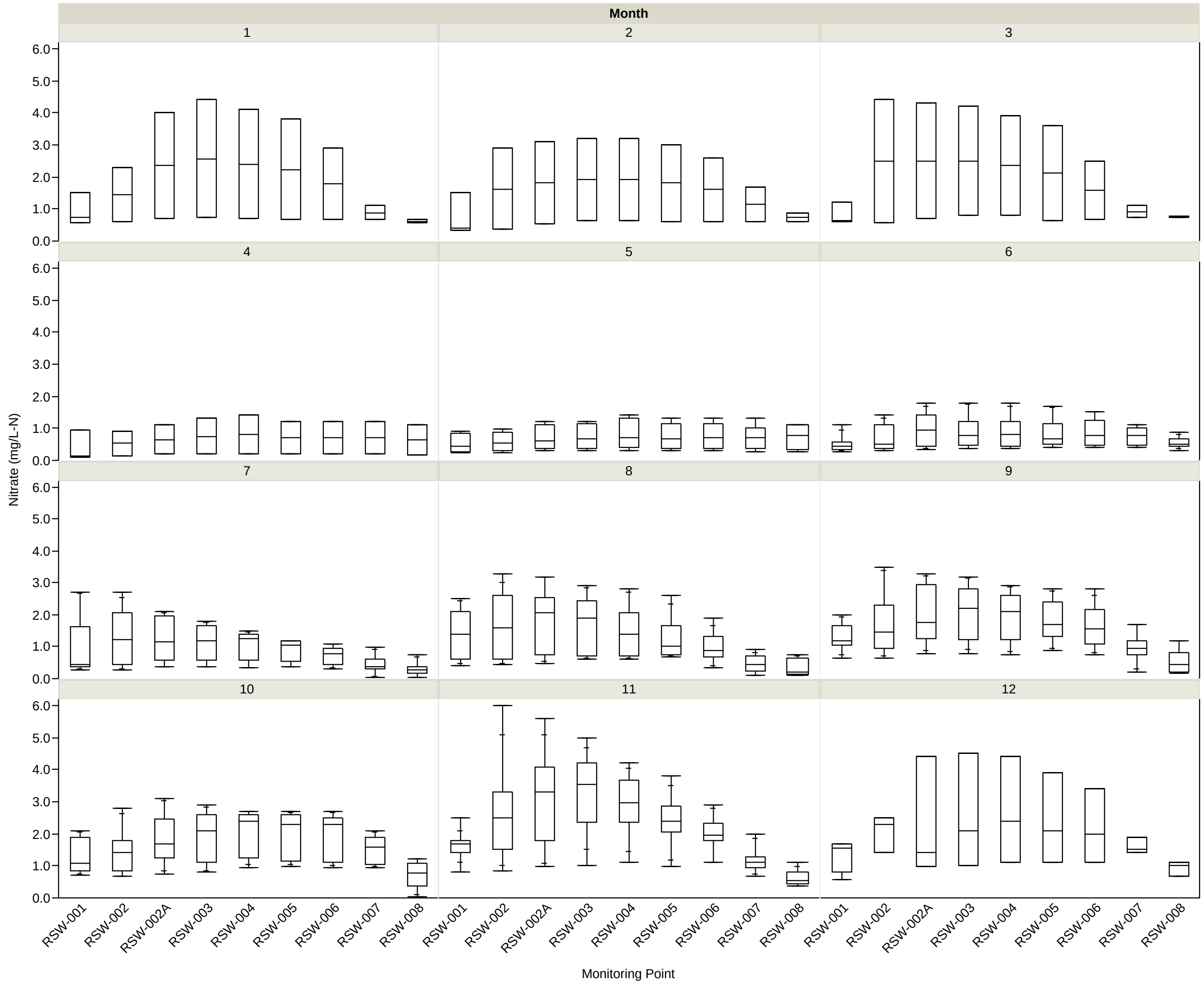


Figure D-3. Monthly quantile box plots of nitrate concentrations measured in samples from the RWCF's nine receiving-water monitoring sites (RSW-001 to RSW-008) on the San Joaquin River during the period January 2010 through December 2012. Samples were taken by RWCF staff in fulfillment of the NPDES permit's receiving water monitoring requirements, and the eight receiving water monitoring sites are defined in Table D-4.

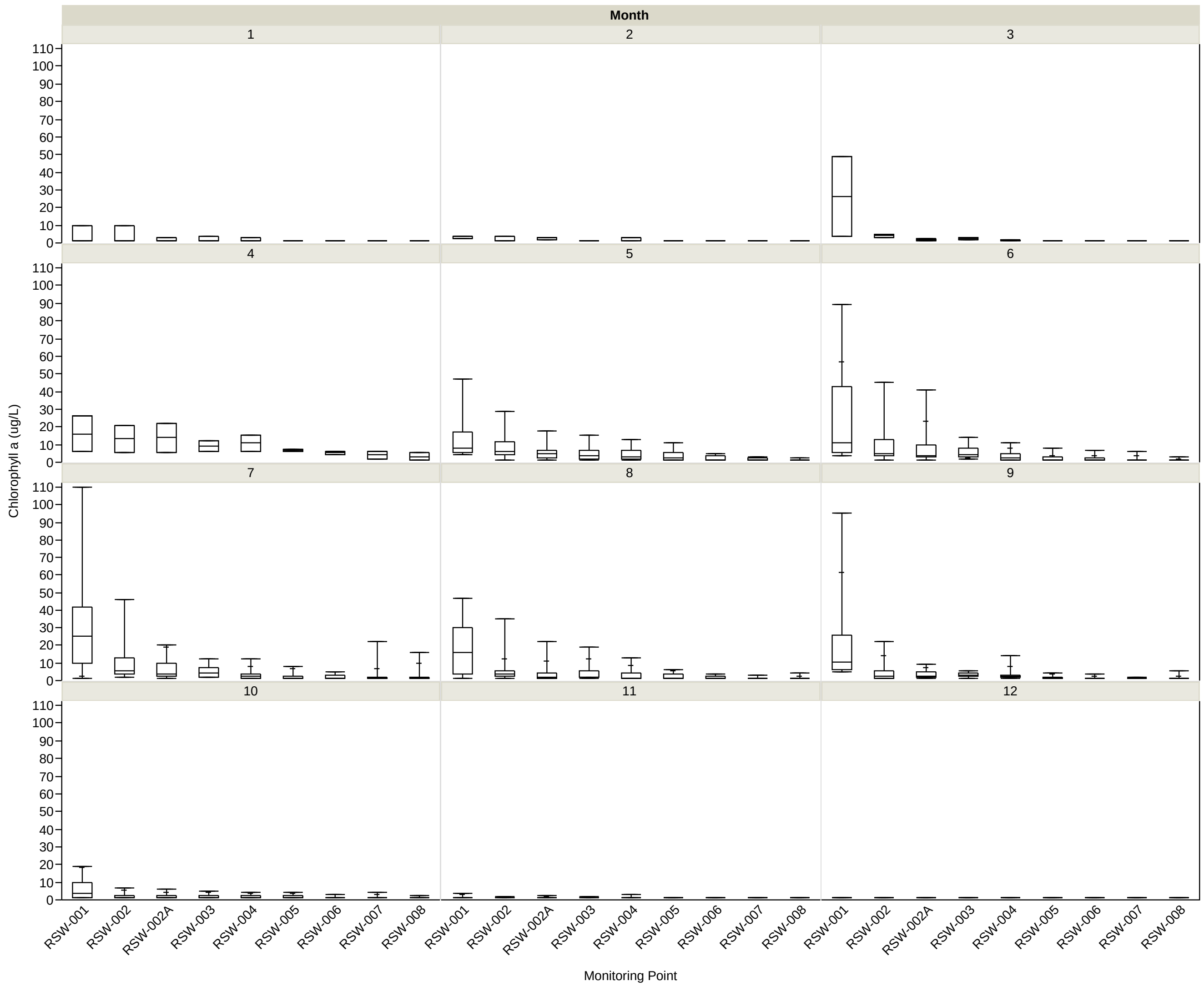


Figure D-4. Monthly quantile box plots of chlorophyll *a* concentrations measured in samples from the RWCF's nine receiving-water monitoring sites (RSW-001 to RSW-008) on the San Joaquin River during the period January 2010 through December 2012. Samples were taken by RWCF staff in fulfillment of the NPDES permit's receiving water monitoring requirements, and the eight receiving water monitoring sites are defined in Table D-4.

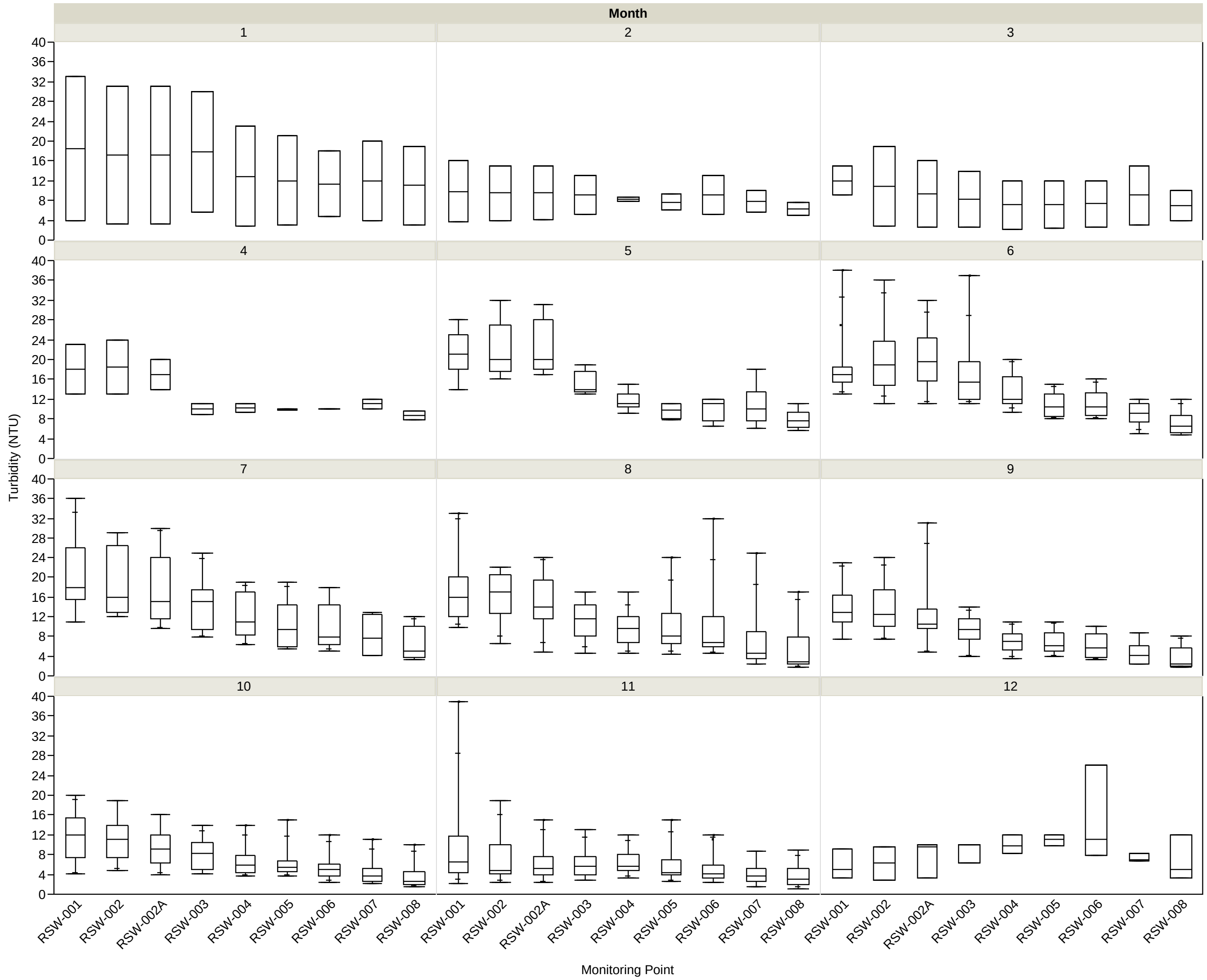


Figure D-5. Monthly quantile box plots of turbidity measured in samples from the RWCF's nine receiving-water monitoring sites (RSW-001 to RSW-008) on the San Joaquin River during the period January 2010 through December 2012. Samples were taken by RWCF staff in fulfillment of the NPDES permit's receiving water monitoring requirements, and the eight receiving water monitoring sites are defined in Table D-4.

Table D-4. Description and location of the RWCF receiving water monitoring sites.

Monitoring Location Name	Monitoring Location Description	Approximate River Mile
RSW-001	San Joaquin River and Bowman Road, 8.0 miles south of Discharge Point No. 001.	47.6
RSW-002	San Joaquin River and Highway 4, 0.5 miles south of Discharge Point No. 001.	42.2
RSW-002A	San Joaquin River and Burns Cutoff, 0.5 miles north of Discharge Point No. 001.	40.7
RSW-003	San Joaquin River at Deep Water Channel, 1.5 miles north of Discharge Point No. 001.	39.7
RSW-004	San Joaquin River at Light 45, 2.5 miles north of Discharge Point No. 001.	38.7
RSW-005	San Joaquin River at Light 41, 3.5 miles north of Discharge Point No. 001.	37.7
RSW-006	San Joaquin River at Light 36, 5.0 miles north of Discharge Point No. 001.	36.2
RSW-007	San Joaquin River at Light 24, 7.3 miles north of Discharge Point No. 001.	33.9
RSW-008	San Joaquin River at Light 18, 9.0 miles north of Discharge Point No. 001.	32.2

Table D-5. Algal biomass by phyla (division) in water samples taken upstream and downstream of the RWCF outfall in the San Joaquin River during March through November 2012.

Month	Phylum	Sampling Site (River Mile)						
		DS-3 (40)	DS-1 (41)	US-1 (41.5)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
March	Diatoms	0.217	1.073	0.229	2.787	1.430	3.939	2.409
	Green algae	0.071	0.044	0.036	0.221	0.296	0.282	0.155
	Chrysophytes	0.040	0.010	0.032	0.012	0.009	0.003	0.006
	Cryptophytes	0.062	0.026	0.020	0.122	0.165	0.040	0.212
	Cyanobacteria	0.267	0.070	0.080	0.099	0.070	0.036	0.146
	Euglenophytes	0.135	0.072	0.021	0.069	0.123	0.079	0.087
	Pyrrhophytes	--	--	--	0.013	--	0.046	0.015
	Miscellaneous	--	--	0.001	--	--	--	--
May	Diatoms	4.169	5.471	5.535	7.939	9.303	8.451	9.219
	Green algae	0.156	0.241	0.082	0.078	0.146	0.139	0.197
	Chrysophytes	--	--	--	0.001	--	--	--
	Cryptophytes	0.130	0.131	0.070	0.082	0.089	0.123	0.051
	Cyanobacteria	0.131	0.053	0.100	0.056	0.259	0.033	0.239
	Euglenophytes	--	--	--	0.090	--	--	0.012
	Pyrrhophytes	--	--	--	--	--	0.004	--
	Miscellaneous	--	--	--	--	--	--	--
July	Diatoms	0.212	0.215	0.116	1.507	3.602	2.870	10.578
	Green algae	0.005	0.015	0.018	0.027	0.036	0.054	0.064
	Chrysophytes	0.001	0.001	--	--	--	--	--
	Cryptophytes	0.011	0.044	0.027	0.018	0.119	0.090	0.025
	Cyanobacteria	0.364	0.178	0.178	0.269	0.375	0.307	0.429
	Euglenophytes	--	--	--	--	--	--	--
	Pyrrhophytes	--	--	--	--	--	--	--
	Miscellaneous	--	--	--	--	--	--	--
November	Diatoms	0.022	0.006	0.020	0.002	0.015	0.013	0.016
	Green algae	0.014	0.016	0.011	0.012	0.026	0.024	0.063
	Chrysophytes	0.005	0.001	0.004	0.002	--	--	--
	Cryptophytes	0.037	0.058	0.060	0.059	0.081	0.100	0.119
	Cyanobacteria	0.064	0.064	0.048	0.039	0.007	0.008	0.011
	Euglenophytes	0.007	0.003	0.002	--	--	--	--
	Pyrrhophytes	--	--	--	--	--	--	--
	Miscellaneous	--	--	--	--	--	--	--

Table D-6. Algal density by phyla (division) in water samples taken upstream and downstream of the RWCF outfall in the San Joaquin River during March through November 2012.

Month	Phylum	Sampling Site (River Mile)						
		DS-3 (40)	DS-1 (41)	US-1 (41.5)	US-4 (45)	US-5R (46)	US-6R (46.5)	US-7R (47.5)
March	Diatoms	1127	679	1053	5524	5507	8330	5378
	Green algae	1363	640	386	3476	3288	2890	3746
	Chrysophytes	189	104	125	322	303	114	202
	Cryptophytes	265	344	227	530	871	303	555
	Cyanobacteria	10193	3813	3112	4474	6554	13137	11661
	Euglenophytes	57	26	7	16	76	15	50
	Pyrrhophytes	--	--	--	38	--	76	25
	Miscellaneous	--	--	8	--	--	--	--
May	Diatoms	9957	9342	11242	10799	18626	13116	19770
	Green algae	1278	2234	1202	1409	2817	1590	4645
	Chrysophytes	--	---	--	45	--	--	--
	Cryptophytes	227	227	182	136	91	136	202
	Cyanobacteria	3449	5111	5446	4687	2650	4435	34573
	Euglenophytes	--	--	--	23	--	--	6
	Pyrrhophytes	--	--	--	--	--	7	--
	Miscellaneous	--	--	--	--	--	--	--
July	Diatoms	304	889	334	2705	5252	4074	8493
	Green algae	220	669	635	593	1659	1092	893
	Chrysophytes	17	42	--	--	--	--	--
	Cryptophytes	152	466	339	169	494	423	330
	Cyanobacteria	23437	19596	12060	8259	37678	19957	22932
	Euglenophytes	--	--	--	--	--	--	--
	Pyrrhophytes	--	--	--	--	--	--	--
	Miscellaneous	--	--	--	--	--	--	--
November	Diatoms	104	76	97	17	18	25	55
	Green algae	118	237	220	373	1016	949	1499
	Chrysophytes	59	34	271	76	--	--	--
	Cryptophytes	982	1059	1224	1660	2016	2562	3328
	Cyanobacteria	4693	4446	5635	2450	4065	4743	6437
	Euglenophytes	8	4	4	--	--	--	--
	Pyrrhophytes	--	--	--	--	--	--	--
	Miscellaneous	--	--	--	--	--	--	--

Appendix E

BMI Detailed Results

Table E-1. Biological metrics for BMI samples collected at seven sites in the San Joaquin River on March 28, 2012.

Biological Metric	Response to Degradation	Site						
		DS-3	DS-1	US-1	US-4	US-5R	US-6R	US-7R
Richness								
Taxonomic	Decrease	8	5	5	8	7	11	10
EPT	Decrease	0	0	0	0	0	0	0
Ephemeroptera	Decrease	0	0	0	0	0	0	0
Plecoptera	Decrease	0	0	0	0	0	0	0
Trichoptera	Decrease	0	0	0	0	0	0	0
Coleoptera	Decrease	0	0	0	0	0	0	0
Predator	Decrease	2	0	0	2	1	3	2
Composition								
EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sensitive EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Shannon Diversity Index	Decrease	1.4	1.1	1.2	1.0	0.9	1.3	0.9
Dominant Taxa (%)	Increase	41	52	51	61	73	59	79
Non-insect Taxa (%)	Variable	88	100	80	88	86	82	70
Tolerance/Intolerance Measures								
Tolerance Value	Increase	6.6	5.2	7.2	5.0	4.9	5.1	4.5
Intolerant Organisms (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Intolerant Taxa (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tolerant Organisms (%)	Increase	40	8.0	34	6.6	12	14	4.7
Tolerant Taxa (%)	Increase	38	40	40	38	43	27	30
Functional Feeding Groups								
Collector-gatherers (%)	Increase	20	42	61	67	19	31	15
Collector-filterers (%)	Increase	77	58	39	32	81	68	82
Scrapers (%)	Variable	0.6	0.0	0.0	0.0	0.0	0.2	0.0
Predators (%)	Variable	2.2	0.0	0.0	1.0	0.2	1.0	2.5
Shredders (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other (%)	Variable	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other								
Abundance	Decrease	179	450	41	2862	2440	3800	529

Table E-2. Biological metrics for BMI samples collected at seven sites in the San Joaquin River on May 23, 2012.

Biological Metric	Response to Degradation	Site						
		DS-3	DS-1	US-1	US-4	US-5R	US-6R	US-7R
Richness								
Taxonomic	Decrease	5	5	4	9	7	9	5
EPT	Decrease	0	0	0	0	0	0	0
Ephemeroptera	Decrease	0	0	0	0	0	0	0
Plecoptera	Decrease	0	0	0	0	0	0	0
Trichoptera	Decrease	0	0	0	0	0	0	0
Coleoptera	Decrease	0	0	0	0	0	0	0
Predator	Decrease	0	1	0	2	2	2	0
Composition								
EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sensitive EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Shannon Diversity Index	Decrease	0.6	0.7	0.4	0.6	0.6	0.6	0.4
Dominant Taxa (%)	Increase	84	70	87	81	87	86	90
Non-insect Taxa (%)	Variable	100	100	100	100	100	78	100
Tolerance/Intolerance Measures								
Tolerance Value	Increase	4.3	4.6	4.3	4.4	4.3	4.2	4.3
Intolerant Organisms (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Intolerant Taxa (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tolerant Organisms (%)	Increase	1.4	1.2	1.7	1.4	2.6	0.7	2.7
Tolerant Taxa (%)	Increase	20	40	25	44	29	22	20
Functional Feeding Groups								
Collector-gatherers (%)	Increase	14	29	11	17	7	11	6
Collector-filterers (%)	Increase	86	71	89	82	89	87	94
Scrapers (%)	Variable	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Predators (%)	Variable	0.0	0.2	0.0	0.5	4.2	2.3	0.0
Shredders (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other (%)	Variable	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other								
Abundance	Decrease	1932	3552	5770	1300	712	920	110

Table E-3. Biological metrics for BMI samples collected at seven sites in the San Joaquin River on July 31, 2012.

Biological Metric	Response to Degradation	Site						
		DS-3	DS-1	US-1	US-4	US-5R	US-6R	US-7R
Richness								
Taxonomic	Decrease	6	4	4	5	8	8	8
EPT	Decrease	0	0	0	0	0	0	0
Ephemeroptera	Decrease	0	0	0	0	0	0	0
Plecoptera	Decrease	0	0	0	0	0	0	0
Trichoptera	Decrease	0	0	0	0	0	0	0
Coleoptera	Decrease	0	0	0	0	0	0	0
Predator	Decrease	0	0	0	0	2	1	1
Composition								
EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sensitive EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Shannon Diversity Index	Decrease	1.4	1.3	1.0	1.2	1.1	1.3	1.3
Dominant Taxa (%)	Increase	41	37	59	52	58	47	53
Non-insect Taxa (%)	Variable	83	100	100	100	75	75	88
Tolerance/Intolerance Measures								
Tolerance Value	Increase	6.6	6.2	5.2	5.7	5.7	6.0	5.5
Intolerant Organisms (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Intolerant Taxa (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tolerant Organisms (%)	Increase	25	20	9.3	7.8	10	16	7.3
Tolerant Taxa (%)	Increase	17	25	25	40	25	38	38
Functional Feeding Groups								
Collector-gatherers (%)	Increase	67	61	62	69	84	78	78
Collector-filterers (%)	Increase	32	39	38	31	14	22	16
Scrapers (%)	Variable	1.2	0.0	0.0	0.0	0.0	0.3	0.2
Predators (%)	Variable	0.0	0.0	0.0	0.0	1.6	0.3	5.7
Shredders (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other (%)	Variable	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other								
Abundance	Decrease	81	122	268	385	451	341	615

Table E-4. Biological metrics for BMI samples collected at seven sites in the San Joaquin River on November 27, 2012.

Biological Metric	**Response to Degradation**	**Site**						
		DS-3	**DS-1**	**US-1**	**US-4**	**US-5R**	**US-6R**	**US-7R**
Richness								
Taxonomic	Decrease	4	4	6	6	5	7	5
EPT	Decrease	0	0	0	0	0	0	0
Ephemeroptera	Decrease	0	0	0	0	0	0	0
Plecoptera	Decrease	0	0	0	0	0	0	0
Trichoptera	Decrease	0	0	0	0	0	0	0
Coleoptera	Decrease	0	0	0	0	0	0	0
Predator	Decrease	0	0	0	1	0	2	1
Composition								
EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sensitive EPT Index (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Shannon Diversity Index	Decrease	0.9	1.3	0.8	1.2	1.0	0.9	1.3
Dominant Taxa (%)	Increase	71	46	78	50	54	62	47
Non-insect Taxa (%)	Variable	100	100	83	100	100	86	100
Tolerance/Intolerance Measures								
Tolerance Value	Increase	5.8	5.8	5.5	6.5	7.8	6.7	7.5
Intolerant Organisms (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Intolerant Taxa (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tolerant Organisms (%)	Increase	15	12	6.7	29	55	33	47
Tolerant Taxa (%)	Increase	25	25	33	50	40	43	20
Functional Feeding Groups								
Collector-gatherers (%)	Increase	81	64	92	60	43	66	43
Collector-filterers (%)	Increase	19	36	7	39	56	33	51
Scrapers (%)	Variable	0.0	0.0	1.0	0.0	0.0	0.0	0.0
Predators (%)	Variable	0.0	0.0	0.0	0.2	0.0	1.1	6.4
Shredders (%)	Decrease	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other (%)	Variable	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other								
Abundance	Decrease	78	284	104	1754	443	752	314

Appendix F

Submerged/Emergent Vegetation Photos

A.

B.

C.

Figure F-1. Sampling location DS-4 during the months of: (A) May 2012, (B) September 2012, and (C) January 2013.

A.

B.

C.

Figure F-2. Sampling location DS-3 during the months of: (A) May 2012, (B) September 2012, and (C) November 2012.

A.

B.

C.

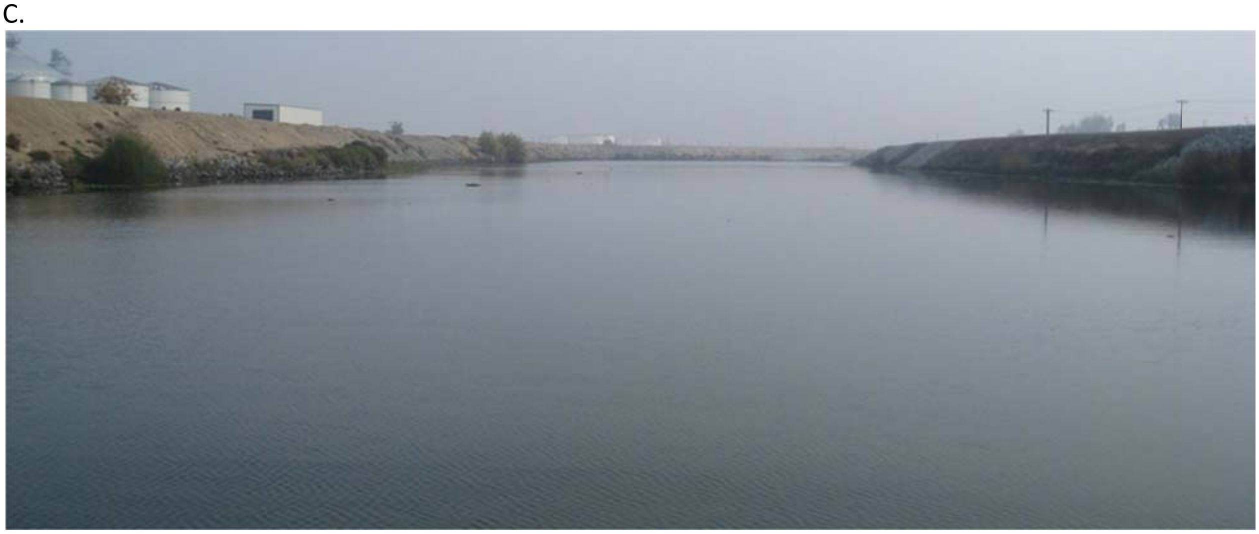


Figure F-3. Sampling location DS-2 during the months of: (A) March 2012, (B) July 2012, and (C) November 2012.

A.

B.

C.

Figure F-4. Sampling location DS-1 during the months of: (A) May 2012, (B) November 2012, and (C) January 2013.

A.

B.

C.

Figure F-5. Sampling location US-1 during the months of: (A) May 2012, (B) September 2012, and (C) January 2013.

A.

B.

C.

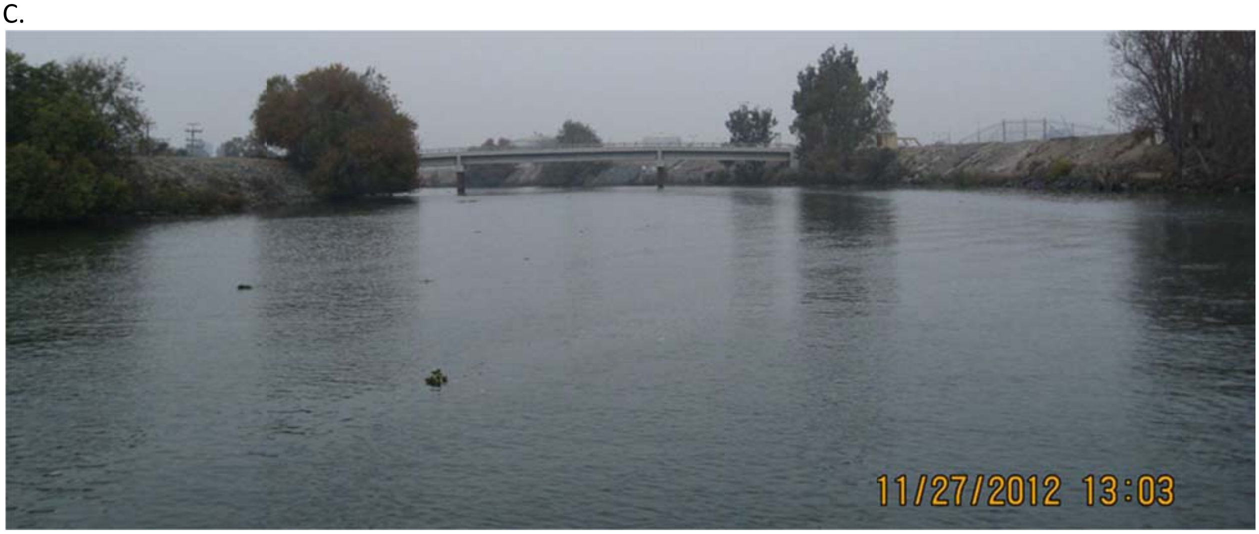


Figure F-6. Sampling location US-2 during the months of: (A) May 2012, (B) July 2012, and (C) November 2012.

A.

B.

C.

Figure F-7. Sampling location US-3 during the months of: (A) May 2012, (B) September 2012, and (C) January 2013.

A.

B.

C.

Figure F-8. Sampling location US-4 during the months of: (A) March 2012, (B) July 2012, and (C) November 2012.

A.

B.

C.

Figure F-9. Sampling location US-5R during the months of: (A) May 2012, (B) September 2012, and (C) January 2013.

A.

B.

C.

Figure F-10. Sampling location US-6R during the months of: (A) March 2012, (B) September 2012, and (C) November 2012.

A.

B.

C.

Figure F-11. Sampling location US-7R during the months of: (A) March 2012, (B) July 2012, and (C) January 2013.

Appendix G

Statistical Analysis of Environmental Parameters, Algal Biomass and Density

Methodology

A mixed General Linear Model (GLM) was used to evaluate differences in measures of water quality constituents and algal biomass and density across the reference sites and the sites within the mixing zone. In this analysis, date of sampling was treated as a categorical random variable, based on the concept that the dates sampled are a sample of all possible dates that could be sampled.

One challenge in this analysis is the status of site US-4. Hydrodynamic modeling indicates that this site is on the boundary between the reference sites and those sites where effluent from the Stockton RWCF is predicted to mix with river water. As such, the analysis was conducted treating this site as a "reference" site, and also by dropping this site entirely from the analysis. Final means for "reference sites" and "impact sites" were computed dropping this site from the analysis.

The results of the GLM analysis for water quality constituents are described in section 4.3.1 (Table 10), and the results for algal metrics are described in section 4.3.4 (Table 14 and Table 15

Table 15).

The influence of measurable environmental parameters on algal taxa biomass and density was evaluated via multiple regression, using Akaikie's Information Criterion (corrected for small sample size = AICc) as means of comparing models, the results of which are presented below. The use of AICc for model comparison is founded on the concept that the explanatory capacity of models needs to be balanced with the parsimony or simplicity of the model. Thus, simpler models are given preference over more complex models when their explanatory capacity is similar. In technical terms, model goodness of fit in AICc is measured by the model's likelihood (assuming a normal distribution), which is then "penalized" for the number of parameters estimated in the model. In comparing one model to another for a particular dependent variable, the "better" model has a lower AICc value. AICc was computed using the mixed procedure in SAS v9.3. Each set of models for a particular dependent variable were ranked by AICc values. We also computed the R^2 for each model, but emphasize that this is provided purely to indicate the amount of variance explained (explanatory power) by each model, and not as an index for model selection.

Algal taxa biomass and density were selected as the dependent variables for the regression analysis, and taxa which were consistently present at each sampling location and during each sampling event were included (e.g., diatoms, cyanobacteria, green algae, and cryptophytes). Environmental parameters used as factors in the regression analysis were nitrate, dissolved phosphate, total phosphorus, Secchi depth, water temperature (temperature), modeled river velocity (velocity), and modeled % effluent. Because total-N was overwhelmingly composed of nitrate, it was omitted as a factor from the regression analysis. An initial model screening also showed TKN was a poor predictor of algal taxa biomass or density, thus TKN was omitted from the final regression analysis. The environmental parameters were also subject to an initial correlation assessment to aid interpretation of the multiple regression results.

Results

Regression analysis of the impact of the environmental parameters on algal biomass and density is limited by the high degree of correlation among the measured environmental parameters (Table G-1). In particular, nitrate was significantly, positively correlated at a value above 0.7 with total nitrogen, dissolved phosphate, total phosphorus, and Secchi depth. Nitrate was also significantly, negatively correlated with velocity. Similarly, dissolved phosphate and total phosphorus were significantly correlated with nitrate, total nitrogen, effluent fraction, and with each other at levels above 0.7. They were also correlated with Secchi disk clarity (R^2 of 0.37 and 0.57, respectively). Velocity was significantly, negatively correlated with dissolved phosphate and total phosphorus (-0.85 and -0.80, respectively). Secchi disk clarity was correlated with a number of the other variables, but was particularly highly correlated with water temperature. Although temperature was significantly correlated with several variables, correlation near or above 0.5 only occurred for Secchi depth and TKN. Given the high degree of correlation among the environmental parameters and given that sample size was relatively small, the following results of the linear regression analysis should be interpreted cautiously.

Tables G-2 through G-7 provide summary statistics (AICc, model rank, and R^2) for each candidate model relating environmental parameters to algal biomass and density. Model ranks for algal biomass varied widely across taxa. For green algae biomass, the top models included dissolved phosphate, velocity, and the combination of temperature and Secchi depth. Top models for cryptophytes biomass also included phosphorus, velocity, and also included the combination of nitrate and dissolved phosphate. Top models for cyanobacteria biomass included temperature, temperature and dissolved phosphate combination, and the combination of temperature and dissolved phosphate and nitrate. Diatom biomass and total algal biomass model ranking were similar. These models were generally more complex than for the other algal taxa and included the combination of nitrate and dissolved phosphate and Secchi depth and temperature and velocity. This model without velocity and this model without nitrate or dissolved phosphate also ranked high for diatom biomass and total algal biomass.

Ranking of models for algal density showed a high degree of similarity across taxa (Table G-6). The model including nitrate, dissolved phosphate, Secchi depth, temperature and velocity ranked highest for all taxa, and the second highest ranking model contained the same factors except for percent effluent substituted for water velocity. The third highest ranking model included all of these factors except excluded percent effluent or velocity.

In general, the explanatory power (R^2) of a model increases as additional factors are added, and as expected, the most complex models evaluated herein had the highest R^2 values. The loss in explanatory power as variables are dropped provides some insight into the relative importance of the environmental parameters. However, because many of the environmental parameters showed some degree of correlation, the amount of variance explained is not completely additive. Nonetheless, a highly parameterized model which included nutrients (nitrate and dissolved phosphate) and physical factors (Secchi depth, velocity, and temperature) was compared both with a model which consisted only of nutrients and with another model consisting only of physical factors. The comparison shows that physical factors contributed the bulk of explanatory capacity for nearly all algal taxa biomass and density measures (Table G-8). Notably, physical factors accounted for nearly all of the variance in cyanobacteria and diatom biomass and density.

A complementary analysis was conducted using the partial correlation (based on type II sums of squares) for each factor in the highly-parameterized model of nitrate, dissolved phosphate, Secchi depth, temperature, and velocity (Table G-9 and Table G-10). The partial correlation for each factor can be regarded as the increase in correlation upon addition of that factor to the model which includes all other factors. The specific factors showing the greatest contribution to R^2, and thus having the greatest explanatory power, were physical factors (except for green algae density). Most notably, temperature had the greatest influence on algal taxa biomass. Velocity was also a major factor driving diatom biomass and density. Nitrate and dissolved phosphate contributed relatively small amounts to the variability of cyanobacteria and diatom biomass and density. Taken as a whole, the modeling and partial correlation analyses attribute the change in algal biomass and density to temperature, velocity, and light (as measured by Secchi depth).

Table G-1. Correlation, associated p-value, and sample size among environmental parameters and algal taxa biomass.

	Total Algal Biomass	Diatom Biomass	Green Algae Biomass	Cyanobacteria Biomass	Cryptophyte Biomass	Nitrate	TKN	Total N	Dissolved Phosphate	Total P	Secchi Depth	Temperature	N:P	% Effluent	Velocity	
Total Algal Biomass	1.00	1.00	0.47	0.33	0.16	-0.71	-0.02	-0.70	-0.74	-0.60	-0.68	0.36	-0.28	-0.48	0.75	Coefficient
		<0.0001	0.0115	0.0914	0.4296	<0.0001	0.9207	<0.0001	<0.0001	0.0008	<0.0001	0.063	0.1449	0.0089	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Diatom Biomass	1.00	1.00	0.45	0.31	0.14	-0.70	-0.03	-0.69	-0.73	-0.59	-0.67	0.35	-0.29	-0.48	0.75	Coefficient
	<0.0001		0.0171	0.1125	0.4878	<0.0001	0.867	<0.0001	<0.0001	0.001	0.0001	0.0676	0.1408	0.0097	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Green Algae Biomass	0.47	0.45	1.00	-0.16	0.53	-0.61	-0.30	-0.63	-0.71	-0.71	-0.46	-0.15	0.07	-0.49	0.67	Coefficient
	0.0115	0.0171		0.4101	0.0041	0.0005	0.1169	0.0003	<0.0001	<0.0001	0.0133	0.4428	0.7358	0.0079	0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Cyanobacteria Biomass	0.33	0.31	-0.16	1.00	-0.24	-0.17	0.65	-0.12	0.08	0.23	-0.49	0.76	-0.43	0.16	-0.12	Coefficient
	0.0914	0.1125	0.4101		0.2152	0.3903	0.0002	0.5511	0.6766	0.2477	0.0079	<0.0001	0.0225	0.4126	0.5488	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Cryptophyte Biomass	0.16	0.14	0.53	-0.24	1.00	-0.41	-0.24	-0.42	-0.51	-0.54	-0.12	-0.23	0.07	-0.50	0.33	Coefficient
	0.4296	0.4878	0.0041	0.2152		0.0292	0.2223	0.0249	0.0053	0.0029	0.5353	0.2433	0.7328	0.0073	0.0851	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Nitrate	-0.71	-0.70	-0.61	-0.17	-0.41	1.00	0.22	1.00	0.84	0.74	0.75	-0.38	0.41	0.49	-0.70	Coefficient
	<0.0001	<0.0001	0.0005	0.3903	0.0292		0.2666	<0.0001	<0.0001	<0.0001	<0.0001	0.0436	0.0285	0.0079	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
TKN	-0.02	-0.03	-0.30	0.65	-0.24	0.22	1.00	0.29	0.32	0.40	-0.21	0.49	-0.07	0.13	-0.38	Coefficient
	0.9207	0.867	0.1169	0.0002	0.2223	0.2666		0.141	0.1005	0.0367	0.2944	0.0084	0.7217	0.5071	0.0489	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Total N	-0.70	-0.69	-0.63	-0.12	-0.42	1.00	0.29	1.00	0.85	0.76	0.72	-0.34	0.40	0.49	-0.72	Coefficient
	<0.0001	<0.0001	0.0003	0.5511	0.0249	<0.0001	0.141		<0.0001	<0.0001	<0.0001	0.0772	0.0356	0.0078	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Dissolved Phosphate	-0.74	-0.73	-0.71	0.08	-0.51	0.84	0.32	0.85	1.00	0.96	0.57	0.03	-0.06	0.75	-0.85	Coefficient
	<0.0001	<0.0001	<0.0001	0.6766	0.0053	<0.0001	0.1005	<0.0001		<0.0001	0.0017	0.8935	0.77	<0.0001	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Total P	-0.60	-0.59	-0.71	0.23	-0.54	0.74	0.40	0.76	0.96	1.00	0.37	0.24	-0.27	0.78	-0.80	Coefficient
	0.0008	0.001	<0.0001	0.2477	0.0029	<0.0001	0.0367	<0.0001	<0.0001		0.051	0.2154	0.1706	<0.0001	<0.0001	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Secchi Depth	-0.68	-0.67	-0.46	-0.49	-0.12	0.75	-0.21	0.72	0.57	0.37	1.00	-0.78	0.52	0.14	-0.59	Coefficient
	<0.0001	0.0001	0.0133	0.0079	0.5353	<0.0001	0.2944	<0.0001	0.0017	0.051		<0.0001	0.0046	0.4673	0.001	P-value
	28	28	28	28	28	28	28	28	28	28	66	66	28	28	28	Sample size
Temperature	0.36	0.35	-0.15	0.76	-0.23	-0.38	0.49	-0.34	0.03	0.24	-0.78	1.00	-0.81	0.22	-0.04	Coefficient
	0.063	0.0676	0.4428	<0.0001	0.2433	0.0436	0.0084	0.0772	0.8935	0.2154	<0.0001		<0.0001	0.2583	0.8415	P-value
	28	28	28	28	28	28	28	28	28	28	66	66	28	28	28	Sample size
N:P	-0.28	-0.29	0.07	-0.43	0.07	0.41	-0.07	0.40	-0.06	-0.27	0.52	-0.81	1.00	-0.29	0.03	Coefficient
	0.1449	0.1408	0.7358	0.0225	0.7328	0.0285	0.7217	0.0356	0.77	0.1706	0.0046	<0.0001		0.1402	0.8823	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
% Effluent	-0.48	-0.48	-0.49	0.16	-0.50	0.49	0.13	0.49	0.75	0.78	0.14	0.22	-0.29	1.00	-0.45	Coefficient
	0.0089	0.0097	0.0079	0.4126	0.0073	0.0079	0.5071	0.0078	<0.0001	<0.0001	0.4673	0.2583	0.1402		0.0175	P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size
Velocity	0.75	0.75	0.67	-0.12	0.33	-0.70	-0.38	-0.72	-0.85	-0.80	-0.59	-0.04	0.03	-0.45	1.00	Coefficient
	<0.0001	<0.0001	0.0001	0.5488	0.0851	<0.0001	0.0489	<0.0001	<0.0001	<0.0001	0.001	0.8415	0.8823	0.0175		P-value
	28	28	28	28	28	28	28	28	28	28	28	28	28	28	28	Sample size

Table G-2. AICc values for models relating environmental parameters to biomass of algal taxa.

Model Description	AICc Values				
	Green Algae Biomass	Cryptophyte Biomass	Cyanobacteria Biomass	Diatom Biomass	Total Algal Biomass
Linear temperature	-40.0	-71.3	-44.8	147.3	148.4
Quadratic temperature	-39.4	-60.4	-34.3	136.1	137.1
Secchi depth	-47.7	-72.4	-31.2	133.4	133.7
Nitrate	-54.2	-77.1	-25.1	130.9	131.4
Dissolved phosphate	-64.8	-84.8	-29.1	124.1	124.6
Dissolved phosphate, nitrate	-58.2	-77.4	-29.1	122.0	122.3
Dissolved phosphate, nitrate, temperature	-51.7	-70.1	-39.3	115.1	115.0
Dissolved phosphate, nitrate, Secchi depth	-51.2	-71.2	-31.5	118.0	117.7
Dissolved phosphate, nitrate, Secchi depth, temperature	-55.2	-62.9	-33.4	114.4	114.4
Nitrate, temperature	-52.7	-72.4	-38.7	133.2	133.6
Dissolved phosphate, temperature	-55.5	-75.4	-42.2	118.4	118.3
% effluent	-45.0	-75.6	-21.1	145.5	146.6
Velocity	-61.3	-79.4	-28.8	123.1	124.0
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	-47.3	-52.8	-23.6	117.7	117.8
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	-53.8	-62.0	-32.1	99.8	100.6
N:P ratio	-39.1	-69.5	-26.8	149.0	150.3
Temperature, Secchi depth	-58.8	-67.7	-38.0	134.4	134.4
Temperature, Secchi depth, velocity	-56.1	-64.7	-36.5	114.1	114.9

Table G-3. Ranking of models relating environmental parameters to biomass of algal taxa.

Model Description	Model Rank				
	Green Algae Biomass	Cryptophyte Biomass	Cyanobacteria Biomass	Diatom Biomass	Total Algal Biomass
Linear temperature	16	9	1	17	17
Quadratic temperature	17	17	7	15	15
Secchi depth	13	8	11	13	13
Nitrate	8	4	16	11	11
Dissolved phosphate	1	1	13	10	10
Dissolved phosphate, nitrate	4	3	13	8	8
Dissolved phosphate, nitrate, temperature	11	11	3	4	4
Dissolved phosphate, nitrate, Secchi depth	12	10	10	6	5
Dissolved phosphate, nitrate, Secchi depth, temperature	7	15	8	3	2
Nitrate, temperature	10	8	4	12	12
Dissolved phosphate, temperature	6	6	2	7	7
% effluent	15	5	18	16	16
Velocity	2	2	14	9	9
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	14	18	17	5	6
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	9	16	9	1	1
N:P ratio	18	12	15	18	18
Temperature, Secchi depth	3	13	5	14	14
Temperature, Secchi depth, velocity	5	14	6	2	3

Table G-4. Variance explained (R^2) by various models relating environmental parameters to biomass of algal taxa.

Model Description	R^2 Values				
	Green Algae Biomass	Cryptophyte Biomass	Cyanobacteria Biomass	Diatom Biomass	Total Algal Biomass
Linear temperature	0.02	0.05	0.58	0.12	0.13
Quadratic temperature	0.38	0.13	0.61	0.54	0.54
Secchi depth	0.21	0.01	0.24	0.44	0.46
Nitrate	0.38	0.17	0.03	0.49	0.50
Dissolved phosphate	0.51	0.26	0.01	0.53	0.54
Dissolved phosphate, nitrate	0.51	0.26	0.19	0.56	0.57
Dissolved phosphate, nitrate, temperature	0.56	0.35	0.63	0.70	0.71
Dissolved phosphate, nitrate, Secchi depth	0.51	0.32	0.43	0.63	0.65
Dissolved phosphate, nitrate, Secchi depth, temperature	0.72	0.35	0.63	0.70	0.71
Nitrate, temperature	0.55	0.35	0.60	0.50	0.51
Dissolved phosphate, temperature	0.53	0.31	0.59	0.67	0.68
% effluent	0.24	0.25	0.03	0.23	0.23
Velocity	0.45	0.11	0.01	0.56	0.56
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	0.75	0.38	0.63	0.70	0.71
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	0.73	0.40	0.63	0.81	0.81
N:P ratio	0.00	0.00	0.18	0.08	0.08
Temperature, Secchi depth	0.65	0.21	0.59	0.47	0.49
Temperature, Secchi depth, velocity	0.65	0.21	0.59	0.72	0.72

Table G-5. AICc values for models relating environmental parameters to density of algal taxa.

Model Description	AICc Values				
	Green Algae Density	Cryptophyte Density	Cyanobacteria Density	Diatom Density	Total Algal Density
Linear temperature	455.7	425.7	551.4	535.7	568.4
Quadratic temperature	441.4	406.7	539.4	496.6	554.5
Secchi depth	449.6	401.4	557.7	519.1	564.1
Nitrate	440.7	428.4	560.0	511.9	565.5
Dissolved phosphate	432.9	425.0	556.2	500.9	566.0
Dissolved phosphate, nitrate	419.9	411.6	532.8	484.2	542.7
Dissolved phosphate, nitrate, temperature	406.9	390.4	513.4	468.0	523.8
Dissolved phosphate, nitrate, Secchi depth	407.8	355.2	514.1	465.8	523.7
Dissolved phosphate, nitrate, Secchi depth, temperature	391.9	343.4	496.1	452.2	506.3
Nitrate, temperature	424.6	413.7	535.5	499.6	546.9
Dissolved phosphate, temperature	423.5	406.7	531.0	484.1	542.0
% effluent	449.8	435.7	564.4	531.7	577.6
Velocity	433.9	420.5	556.8	492.5	566.1
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	381.5	335.6	483.0	439.7	492.7
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	374.7	323.7	474.7	416.6	482.9
N:P ratio	456.5	433.0	558.6	536.3	572.6
Temperature, Secchi depth	429.0	392.8	535.1	500.2	548.5
Temperature, Secchi depth, velocity	410.3	377.1	514.0	456.8	523.5

Table G-6. Ranking of models relating environmental parameters to density of algal taxa.

Model Description	Model Rank				
	Green Algae Density	Cryptophyte Density	Cyanobacteria Density	Diatom Density	Total Algal Density
Linear temperature	17	15	12	17	16
Quadratic temperature	14	10	11	10	11
Secchi depth	15	8	15	15	12
Nitrate	13	16	17	14	13
Dissolved phosphate	11	14	13	13	14
Dissolved phosphate, nitrate	7	11	8	8	8
Dissolved phosphate, nitrate, temperature	4	6	4	6	6
Dissolved phosphate, nitrate, Secchi depth	5	4	6	5	5
Dissolved phosphate, nitrate, Secchi depth, temperature	3	3	3	3	3
Nitrate, temperature	9	12	10	11	9
Dissolved phosphate, temperature	8	10	7	7	7
% effluent	16	18	18	16	18
Velocity	12	13	14	9	15
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	2	2	2	2	2
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	1	1	1	1	1
N:P ratio	18	17	16	18	17
Temperature, Secchi depth	10	7	9	12	10
Temperature, Secchi depth, velocity	6	5	5	4	4

Table G-7. Variance explained (R^2) by various models relating environmental parameters to density of algal taxa.

Model Description	R^2 Values				
	Green Algae Density	Cryptophyte Density	Cyanobacteria Density	Diatom Density	Total Algal Density
Linear temperature	0.01	0.28	0.37	0.05	0.27
Quadratic temperature	0.26	0.57	0.40	0.70	0.34
Secchi depth	0.15	0.70	0.13	0.46	0.33
Nitrate	0.39	0.13	0.04	0.58	0.29
Dissolved phosphate	0.46	0.09	0.01	0.67	0.14
Dissolved phosphate, nitrate	0.48	0.13	0.22	0.69	0.31
Dissolved phosphate, nitrate, temperature	0.54	0.47	0.37	0.74	0.42
Dissolved phosphate, nitrate, Secchi depth	0.48	0.87	0.30	0.74	0.37
Dissolved phosphate, nitrate, Secchi depth, temperature	0.58	0.88	0.38	0.75	0.42
Nitrate, temperature	0.54	0.31	0.37	0.59	0.41
Dissolved phosphate, temperature	0.47	0.38	0.37	0.74	0.42
% effluent	0.26	0.01	0.02	0.24	0.03
Velocity	0.45	0.25	0.01	0.77	0.15
Dissolved phosphate, nitrate, Secchi depth, temperature, % effluent	0.62	0.89	0.38	0.76	0.42
Dissolved phosphate, nitrate, Secchi depth, temperature, velocity	0.59	0.91	0.38	0.88	0.46
N:P ratio	0.00	0.07	0.18	0.05	0.17
Temperature, Secchi depth	0.45	0.70	0.38	0.58	0.36
Temperature, Secchi depth, velocity	0.49	0.70	0.38	0.84	0.45

Table G-8. Summary of R^2 for various models relating nutrients and physical factors to algal taxa biomass and density.

Algal Metric	Model Description	R^2 Values				
		Green Algae	Cryptophytes	Cyanobacteria	Diatoms	Total Algae
Biomass	Nutrients + Physical Factors	0.73	0.40	0.63	0.81	0.81
	Nutrients (nitrate, dissolved phosphate)	0.51	0.26	0.19	0.56	0.57
	Physical Factors (temperature, Secchi depth, velocity)	0.65	0.21	0.59	0.72	0.72
Density	Nutrients + Physical Factors	0.59	0.91	0.38	0.88	0.46
	Nutrients (nitrate, dissolved phosphate)	0.48	0.13	0.22	0.69	0.31
	Physical Factors (temperature, Secchi depth, velocity)	0.49	0.70	0.38	0.84	0.45

Table G-9. Partial R^2 values for factors in the linear regression model relating nutrients (nitrate, dissolved phosphate) and physical factors (Secchi depth, temperature, velocity) to algal taxa biomass.

Model Factor	Partial R^2 Values				
	Green Algae Biomass	Cryptophyte Biomass	Cyanobacteria Biomass	Diatom Biomass	Total Algal Biomass
Nitrate	0.181	0.065	0.092	0.160	0.151
Dissolved phosphate	0.032	0.014	0.062	0.306	0.302
Secchi depth	0.362	0.021	0.000	0.195	0.161
Temperature	0.433	0.099	0.259	0.440	0.417
Velocity	0.048	0.084	0.010	0.360	0.336

Table G-10. Partial R^2 values for factors in the linear regression model relating nutrients (nitrate, dissolved phosphate) and physical factors (Secchi depth, temperature, velocity) to algal taxa density.

Model Factor	Partial R^2 Values				
	Green Algae Density	Cryptophyte Density	Cyanobacteria Density	Diatom Density	Total Algal Density
Nitrate	0.154	0.372	0.000	0.047	0.001
Dissolved phosphate	0.026	0.044	0.001	0.192	0.005
Secchi depth	0.043	0.691	0.007	0.125	0.028
Temperature	0.135	0.024	0.107	0.307	0.135
Velocity	0.013	0.200	0.003	0.519	0.068