Design Tools for Vegetated Treatment Systems/ Constructed Wetlands/ Woodchip Bioreactors

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PG&E Fund Grant #2005-0735

1. Introduction

There is growing interest among stakeholders regarding the efficacy of constructed treatment wetlands to reduce nutrient and sediment loads from agricultural runoff in Central Coast watersheds. An important consideration in determining whether to utilize wetland treatment is the area of land required to install a wetland, due primarily to the cost of removing this land from production and also to the cost of installation. The capital cost of artificial wetland construction is not inexpensive (Kadlec and Wallace 2009) and land prices in Monterey County are high. However, wetland operation and maintenance costs are low when compared with other wastewater treatment technologies because they employ energies from the sun, wind, water, plants and microbes for chemical degradation (Kadlec and Knight 1996). The purpose of this study was to develop a user friendly model for sizing constructed wetlands to determine the water surface area needed for treating runoff containing "estimated" discharge concentrations of nitrate to local water quality standards. The model allows growers, resource conservation agencies, and wetland designers to input runoff nitrate concentration, discharge, water temperature, and targeted nitrate output concentration, and the model output returns the wetland area required to reduce nitrate concentration to targeted levels. Also included in the output is a table with the median size in acres for four different wetland types, the nominal hydraulic retention time, nitrate load reduction, and concentration reduction. Model output does not include service areas for the wetland, such as roadways for periodic maintenance.

We identified specific nutrients to include in the wetland sizing study based on the magnitude of discrepancy between regulatory water quality objectives and the monitored water quality findings. We found total nitrogen, nitrate, unionized ammonia and phosphate to be the nutrients of concern in the region. The EPA drinking water standard for nitrate as N is 10 mg/L and matches the numeric target established by the CCRWQCB for municipal and groundwater primary uses (RWQCB 2010). The Central Coast Ambient Monitoring Program (CCAMP, http://www.ccamp.info) display of water quality data shows the Salinas watershed nitrate as N concentration occasionally exceeds 150 mg/L with median concentrations ranging from less than 1 mg/L (309ATS, 309DSA, 309NAC, 309GRN, 309KNG, 309LOK, 309NAC, etc) to above 35 mg/L at some locations (56 mg/L ,309UQA; 41.7 mg/L, 309ASB; 64.4 mg/L, 309BLA; 35.0 mg/L, 309ESP; 37.1 mg/L, 309NOS; 41.2 mg/L, 309SDR; 42.4 mg/L, 309SOS). The median nitrate concentration at 25 out of 50 (50%) Salinas watershed sites displayed in CCAMP WQ data exceeded the WQ objective of 10 mg/L. The maximum concentration at each site shows that 34 out of 50 (68%) monitoring sites in Salinas had at least one measured concentration above the water quality objective of 10 mg/L. Based on the frequency of nitrate concentrations detected above the CCRWQCB target for surface water and the concern for ground water contamination above the drinking water standard, we focused wetland sizing estimates provided by the model on area needs to reduce nitrate. Other chemicals may be removed from the water simultaneously as it undergoes treatment through the wetland; however our model does not estimate the sizing requirements for these contaminants, for reasons explained below.

Other nitrogenous compounds detected in local waters at concentrations exceeding regional water quality objectives and known to be removed by wetlands are ammonia and total nitrogen. The numeric target for unionized ammonia as N established by the CCRWQCB for inland surface waters is based on an aquatic toxicity objective of 0.025 mg/L (RWQCB 2010). CCAMP monitoring results for unionized ammonia as N were measured at a maximum of 3.34 mg/L (309ALG) with the median concentrations ranging between 0.0004 mg/L (309NAC, 309SAS, 309SDR) to above 0.03 mg/L at 2 monitoring sites (309AGL, 309CRR). The maximum concentration at each site shows that 32 out of 47 (68%) monitoring sites in Salinas had measured concentrations above the water quality objective of 0.025 mg/L. Total nitrogen in local surface waters has been occasionally monitored at concentrations over 100 mg/L in the lower Salinas watershed with mean total nitrogen concentrations at monitoring locations reported between about 1 mg/L and over 40 mg/L (RWQCB 2010). The goal for total nitrogen (TN) recommended as a preliminary numeric nutrient target to prevent excessive algal growth in the "Lower Salinas River watershed nutrient TMDL" is between 1.4 mg/L and 6.0 mg/L depending on water temperature and percent shading (RWQCB 2010). Due to the complexity of nitrogen cycling in wetlands and the transformation from one form of nitrogen compound to another, we did not attempt to model removal of other nitrogen containing constituents besides nitrate.

Our investigation of four types of constructed wetlands, also called vegetated treatment systems (VTS), revealed performance differences between types in regard to their rate of chemical transformation and therefore considerable differences in the area required for water treatment. In addition to differences between VTS types, a range of performance for removal of each water contaminant is found within each wetland type (Kadlec 2009). Two types of wetlands, free water surface (FWS) wetlands and horizontal subsurface flow (HSSF) wetlands, have been used for waste water treatment for over 50 years and a large number of these wetlands have been researched and their performance characterized (Kadlec and Wallace 2009, Vymazal 2011). We developed a probability distribution for the range of performance for these two wetland types based on Kadlec's (2009) comparison of performance of a large number of wetlands. This range is displayed as model output in the form of a histogram of areas based on a Monte Carlo random draw of 1000 possible decay rates from the probability distribution. A third type of wetland is sometimes called a woodchip horizontal subsurface flow wetland and other times a woodchip bioreactor. Woodchip bioreactors are a more recent development and the information to compare a large number of systems was not available. The earliest report of woodchip bioreactors nitrate removal was a 1994 study by Blowes et al. researching 55 gallon containers in series filled with woodchips. Very few studies of actual on-farm working woodchip bioreactors are available, thus we were not able to model a probability distribution for the performance of this type of VTS. Instead we portray five different areas on the model output graphic display based on the range of performance observed in woodchips reactors. We engaged in field research of the performance of local vegetated ditches in Salinas over a 3 year period from 2007 through 2009 (Largay et al. 2008, LosHuertos and Krone-Davis 2010). Insufficient research of nitrate reduction in vegetated agricultural ditches exists to develop a probability distribution and the model instead displays a range of three values based on this research.

Of the four wetland types reviewed, superior performance regarding nitrate reduction with the least area is achieved by woodchip bioreactors. The area required for a four foot deep woodchip bioreactor to treat a 20 gpm discharge of 60 mg/L nitrate as N to the drinking water standard of 10 mg/L ranged between 4200 ft^2 to 37000 ft^{2}, depending on the unique performance of different wetlands. Comparing the median performance of all wetland types using these same model inputs, the area required by woodchip bioreactors was 13.4%of that required for HSSF wetlands using gravel substrate and only 7.2% of the size required for an FWS wetland. Although the per unit area cost of FWS is lower than that of woodchip bioreactors, the lower area requirements of the woodchip reactor make it the more cost effective treatment option. The woodchip bioreactors also have the advantage over FWS wetlands of not harboring mosquitoes and attracting lower biodiversity, thus providing an advantage when these systems are located adjacent to fields of leafy greens where food safety may be a concern. However, if the goal is the highest provision of ecosystem services and greatest societal benefit, FWS wetlands provide a rich array of biodiversity and when publically located can become sites for recreation, education and public enjoyment (Mitsch and Gosselink 2000b).

In addition to nitrogen, both phosphorus and sediment are also constituents of concern in California's Central Coast surface water. The preliminary goal for total phosphorus (TP) in the Salinas TMDL preliminary report is between 0.024 and 0.088 mg/L depending on temperature and shading (RWQCB 2010). CCAMP online reporting shows ortho-phosphate as P was reported with medians above 0.088 for all but 2 out of 28 (93%) of sites monitored in the lower Salinas watershed (http://www.ccamp.info/_2010/). Although sediment can be removed by wetlands, the sediment remains in the wetland and will eventually clog the pore space between the media (including woodchips) in subsurface flow wetlands. Water flow will restricted to open spaces, resulting in lower treatment effectiveness. As the woodchip bioreactors are the most efficacious wetland for nitrate treatment, we recommend removal of sediment from the water in advance to prevent clogging and maintain effectiveness. Phosphorus removal in woodchip reactors has not been researched, however in FWS wetlands, it is primarily removed through adherence to sediment particles (Vymazal 2006).

2. Model Development and Parameter Determination

We originally proposed to use the DNDC model for wetland sizing as it is used in agricultural ecosystems to model carbon and nitrogen dynamics (ISEOS 2007). There is a component of the model that predicts nitrate and ammonia fluxes based on nitrification, denitrification and fermentation sub-models, and the model includes a wetland tool. There are many input parameters for this model including site, climate, soil, farm management, crop, tillage, fertilization, weeding, manure amendment, flooding, irrigation, grazing and cutting (ISEOS 2007). After testing the model and discussing output issues with the model authors, we determined that this was not the best model for local wetland sizing both because of the extensive input requirements and because it was not truly operable for wetland design purposes in its current stage of development. However, there are aspects of the DNDC model that pertain to nutrient management in wetland systems that are incorporated into the more user friendly wetland sizing model we developed, primarily from tanks in series concepts used in chemical engineering and applied to wetland by Kadlec and Wallace (2009). Tanks-in-series theory incorporates wetland hydrology and chemical transformation into a steady state model of contaminant reduction.

2.1. Variables pertaining to VTS design to treat nitrate

Several conditions influence the rate of nitrate transformation in wetlands and vegetated treatment systems. Our model for determining VTS size requirements incorporates some of these conditions into variables entered as input values by the model user and others as working variables in the model code. Yet other conditions are explained in this report and design guidelines are provided to help designers optimize wetland performance. In addition to explained parameters that influence performance, many unexplained performance differences remain. Despite years of study, wetlands are complex ecological systems and much remains to be discovered regarding their performance (Kadlec and Wallace 2009).

2.1.1. Dentrification

Nitrate removal in wetlands occurs through the processes of plant uptake and microbial denitrification; however denitrification accounts for the majority of removal at high nitrate inlet concentrations (Ingersoll and Baker 1998). Whereas the nitrogen taken up by plants can be released back to the wetland during senescence, the removal by denitrification as N_2 gas by microbial respiration is permanent (Ingersoll and Baker 1998, McGehan 2005). Denitrification in wetlands occurs in the anoxic zones of sediments and on the biofilms adhering to plant surfaces and to substrates (van Oostrom 1995, Kadlec and Knight 1996). In a comparison of nitrate removal by macrophyte uptake versus denitrification, Lin et al.'s (2002) study found 4-11% removal was by uptake whereas denitrification accounted for 89-96% removal. This does not infer macrophytes are unimportant in VTS systems. Lin et al.'s (2002) comparison of nitrate removal in planted and unplanted microcosms revealed that those planted with macrophytes removed greater amounts of nitrate than unplanted wetland microcosms containing only algae, with macrophyte presence accounting for 0.36 to 0.92 g m^2/day of the total removal of 0.63 to 1.26 g m^2/day. This greater removal was not due to direct uptake by the macrophytes, but to the increased microbial surface area provided by macrophyte plant stems and leaves as well as to the increased carbon available for microbial utilization as plants died and decomposed (Lin et al. 2002).

Denitrification occurs by microbial respiration using oxidized nitrogen as an electron acceptor when oxygen is limited (Mitsch and Gosselink 2000). Denitrification by heterotrophic bacteria occurs under anaerobic conditions with nitrate acting as a terminal electron acceptor and giving off nitrogen gas as the product of the reaction (Mitsch and Gosselink 2000):

$$C_6H_{12}O_6 + 4NO_3^- \longrightarrow 6CO_2 + H_2O + 2N_2$$

Denitrification to N_2 is the most prevalent pathway for nitrate reduction, however N_2O, a greenhouse gas can also be produced during nitrate reduction (Mitsch and Gosselink 2000, McGechan 2005). The production of N_2O is a concern because its greenhouse gas potential for global warming is 300 times that of CO_2. In their review of N_2O production in 15 wetlands, Kadlec and Wallace (2009) identified a mean emission of about 2.2% of the nitrogen load and a median of only 0.3%. A wetland located in Estonia had N_2O emissions that were 7.6% or the nitrogen supplied, whereas a wetland in China released only 0.10 % of the nitrogen supplied to N_2O. Although there is considerable variability, N_2O production is generally low in wetlands, and proper design can help minimize its release. Future research will elaborate the factors associated with favoring N_2 production and reducing N_2O production, with current indication that plant type and temperature are influential (Kadlec and Wallace 2009).

Denitrification by both anearobic and facultative microorganisms occurs both in the water-saturated soils of wetlands and on epiphytic films covering the immersed surfaces of macrophytes growing in the wetland. Denitrification can happen in different soil types including those with organic and mineral compositions, although in some cases the acidity of peat soils can hinder the process (Gosselink and Mitsch 1994). In mineral soils, the carbon for this process is gained by the decomposition of plant detritus that accumulates on top of the sediment layer. For this reason, young wetlands, without this layer of detritus, do not have the carbon source needed for denitrifying bacteria to engage in respiration and therefore have low denitrification rates (Kadlec and Wallace 2009). Denitrification occurs in the sediment layer below the contact layer with water, deep enough that oxygen is not transported to the layer. In the presence of oxygen, facultative bacteria will instead use O_2 for respiration because of its higher oxidation potential (Kadlec and Wallace 2009).

Plant type and density influence the rate of denitrification and will be further explored as a separate topic later in this section. Further evidence of the importance of plant surfaces as locations for denitrification is the finding by Bourgues and Hart (2007) that reaction rates on epithetic films are comparable with those of anaerobic sediments (Bourgues and Hart 2007). The presence of microscale anaerobic conditions in bio- films on plant surfaces allows for the growth of denitrifying bacteria within a community containing microalgae and other bacterial populations (Freeman and Lock 1995, Bourgeus and Hart 2007). The denitrifying bacteria growing on epiphytic films are embedded in a matrix of mucus and sugars, which can provide them a short-term energy supply when external carbon is limited (Freeman and Loch 1995). However for maximum denitrification to occur, sufficient carbon must be available to drive the microbial processes. Although adding external carbon to the wetland can increase denitrification, a significant amount of the carbon can also be lost to microbial oxidation (Lin et al. 2002).

Ingersoll and Baker (1998) suggested a ratio of carbon to nitrogen of 5:1 to prevent carbon limitation and to maximize the performance of denitrifying bacteria. Hume et al. (2002) found that not only the amount of carbon but also the type of carbon is important to the denitrification rate, with acid soluble carbohydrates promoting higher rates of denitrification. Hume et al. (2002) also reported an increase in denitrification rates occur at higher C:N ratios, even up to 10:1 ratio. If we assume a 5:1 ratio and optimum planting with high carbohydrate material, the wetland itself may supply the carbon needs of the system, depending on the nitrate loading rate and the wetland productivity. Carbon in decomposing and dead plant matter comprises about 40% of the dry biomass and wetland productivity varies from 500 to 2000 g(C) / m^2 yr (Baker 1998, Hume et al. 2002). A wetland with an inlet concentration of 30 mg/L nitrate as N at a discharge rate of 100 m^3/day would receive a total of 1,095,000 gm/yr of nitrate as N. To maximize the denitrification assuming the 5:1 ratio, a total of 5,475,000 gm/yr of carbon would be required. In order to supply its own carbon from plant decay, the wetland size would need to be 2700 m^2 assuming a highly productive wetland and 11,000 m^2 assuming a low productivity wetland. Wetlands that do not have sufficient carbon production internally to fuel the denitrification process will benefit from carbon addition from external sources such as fructose, shredded plant material, or sawdust (Baker 1998, van Oostrom 1998).

2.1.2. Wetland Type

Different designs for constructed wetlands include free water surface wetlands (FWS) , horizontal. subsurface flow wetlands (HSSF) and vertical flow (VF; Kadlec 2009). FWS wetlands have open water containing submerged or emergent plants and are similar in appearance to natural marshes (Kadlec and Wallace 2009). HSSF are composed of a gravel, woodchip, or soil substrate planted with wetland vegetation where the water table is below the surface (Kadlec and Wallace 2009, Leverenz et al. 2010). Because the performance of HSSF wetlands with woodchip media is considerably better than with non-carbon substrates, this will be considered in the model as its own wetland type. Treatment in HSSF occurs as the water flows through the substrate material and around plant roots. Vertical flow wetlands are less common and are used primarily for treatment of ammonia, therefore these systems will not be reviewed in this paper. Typically, FWS wetlands have a water depth of 0.3 meters and HSSF have a bed depth of 0.6 meters, however for FWS 95% or more of the depth is water content and for HSSF only 60% is water (Kadlec 2009). If we remove the substrate portion of the HSSF the "effective" water depths of FWS and HSSF are 0.3 m and 0.24 m respectively.

There are performance differences between HSSF and FWS wetlands when compared on an areal basis for most water quality parameters (Kadlec 2009). They are also different in terms of the habitat they provide. As the water is not exposed to air in HSSF, they do not provide mosquito breeding habitat (Kadlec and Wallace 2009). FWS tend to have habitat for greater biodiversity (Kadlec 2009), which may be a disadvantage for on-farm treatment. Furthermore, Kadlec's (2009) review of approximately 30 wetlands of both types estimated a background concentration of fecal coliform[1] of 40 in FWS; whereas for HSSF wetlands pathogen background concentration is 0, because there is minimal or no wildlife interaction.

When comparing data sets from a large number of treatment systems, Kadlec (2009) found differences in the median removal rate by wetland type and also considerable overlap in the results achieved by FWS and HSSF for most contaminants. As shown in Table 1, high performance variability exists between wetlands of the same type. To compare the performance of FWS to HSSF systems for nutrient removal, Kadlec determined the areal rate coefficients for each system for each analyte. This comparison shows evidence of better performance of HSSF in regard to nitrate and phosphorus removal whereas FWS generally demonstrated better ammonia removal . Dentrification areal decay rate constants computed for nitrate in 72 FWS wetlands ranged from about 2 - 55 m/yr and for 216 HSSF wetlands in a range from about 3 - 72 m/yr with median values of about 27 m/yr and 42 m/yr respectively.

[1] Units were not given in this paper.

Table 1. Kadlec (2009) compared the performance of many FWS and HSSF wetlands and determined areal decay rate coefficients for nutrients. The larger the decay constant (k_a), the less the wetland treatment area required. Performance is similar between wetland types, however there is a considerable range of wetland performance.

	FWS				HSSF			
	Decay Rate k_a (m/yr)				Decay Rate k_a (m/yr)			
Nutrient	low	high	median	number of wetlands	low	high	median	number of wetlands
Nitrate	2	55	27	72	3	72	42	216
Ammonia	0	82	14.7	118	0	62	11.4	214
TN	0	39	12.6	116	2	30	8.4	123
TP	0	25	10	282	2	33	6	82

Leverentz et al. (2010) found the HSSF rock substrate typical in HSSF design prevents plant debris from entering the submerged area where treatment takes place, thus making them inefficient for denitrification due to carbon limitation (Leverenz et al. 2010). By using woodchips for the substrate in place of gravel, denitrifying bacteria gain both the carbon source they require for metabolism as well as increased surface area on which to grow (Leverenz et al. 2010). After two years of operation, the woodchip HSSF wetland demonstrated first-order volumetric reaction constants of 1.41 and 1.3 day^{-1} for planted and unplanted systems respectively. The bed depth was 0.85 meters and porosity was 0.58. Using these values, the volumetric decay rate converts to an areal decay rate of 250 m/yr, nearly 6 times the median value for HSSF wetlands in the data bases reviewed by Kadlec (2009). The woodchip HSSF was operated over 2 years, with a decline in the first-order decay rate from 2.6 to 1.4 day^{-1} as the woodchip media aged. As steady state was not reached, the long term performance of this type of system is uncertain (Leverenz et al. 2010). This study was conducted in fiberglass containers on a relatively small scale, with an expectation of performance loss associated with sizing up to a full scale system.

Moorman et al. (2010) studied the nitrate reduction performance of a woodchip bioreactor in a field setting over 9 years and calculated a first order volumetric decay rate of 1.08 ± 0.19 day^{-1}. In the woodchip bioreactors studied by Moorman et al. (2010), wood loss occurred over the nine years of operation, with greater loss in top layers not consistently saturated with water. The woodchip half life was estimated at 4.6 years (Moorman et al. 2010). Robertson (2010) reported that woodchips loose 50% of their denitrifying potential after the first year but then stabilize to a period of consistent performance lasting 5-15 years, whereas Christianson (2011) reports a lifespan of 15-20 years.

Woodchip bioreactors are sometimes installed with liners and other times without in areas such as edge of field buffer strips or grassed areas to avoid displacing production (Christianson 2011). They can be planted or unplanted with only small differences in performance (Leverenz 2010). They have been installed in parallel containers in the banks of waterways (Blowes et al. 1994). One woodchip reactors was installed directly in a stream bed that had been previously trenched for agricultural drainage (Roberston 2010). A large variety of chips including discarded wood pallets, pine chips, hardwood and redwood have proven effective (Leverenz personal communication 2011, Schipper 2010).

HSSF are more expensive to install per unit area than FWS wetlands, costing roughly 3.3 times as much (Kadlec 2009); however the lesser area required to achieve the same nitrate removal more than offsets this cost when using woodchip media. The capital cost estimate for construction of both types of wetlands in 2006 dollars reported by Kadlec (2009) after reviewing a large number of constructed wetlands installed in the United States is proportional to the wetland area by the following formula:

$$\text{FWS: } C = 194A^{0.690},\ R^2 = 0.79,\ 0.03 < A < 10{,}000$$

$$\text{HSSF: } C = 652A^{0.704},\ R^2 = 0.75,\ 0.005 < A < 20$$

where C is the cost (2006 USD in thousands) and A is the land area (hectares). Cost estimates for a large number of woodchip HSSF are not available for comparison, however Christianson and Helmers (2011) report costs in the range of $7000 to $10,000 for on-farm systems in Iowa covering an area of 1000 to 3000 ft^2. For a 4000 ft^2 bioreactor in California, Leverenz and King (2010) reported an estimated initial cost of $20,000. Whether permitting or land is included in these estimates is not clear.

2.1.2. Hydrology

The hydrology of wetlands is an often ignored yet important parameter in quantifying wetland performance (Kadlec and Wallace 2009). Water entering the inlet of a wetland does not all follow the same travel path nor spend the same amount of time in the wetland system, as would be predicted by plug flow models. Plug flow assumes the same flow rate of all water entering the wetland, such that water would flow down the channel like a plug. While plug-flow assumptions work well for mesocosm studies, these assumptions do not apply to full scale wetlands where the forces of advection and dispersion effect the residence time distribution function of water undergoing wetland treatment. If plug flow conditions are assumed inappropriately for some extrapolations, performance discrepancies occur (Kadlec and Wallace 2009). Because nitrate undergoes treatment by microbes in the wetland, the residence time of the water in the wetland influences the amount of transformation of nitrate to nitrogen gas that occurs. In the context of hydrologic considerations of wetlands, a "number of tanks in series (NTIS)" model can be used to describe the residence time distribution function (Kadlec and Knight 1996). The number of tanks in series (NTIS) model for contaminant removal is described by the equation

$$\frac{C_o}{C_i} = \left(1 + \frac{k_v \bar{t}}{N}\right)^{-N}$$

where the C_i is inlet concentration (mg/L), C_o is the outlet concentration (mg/L), k_v is the volumetric removal rate constant (d^{-1}), N is the number of tanks in series (dimensionless), and $\bar{t}$ is the mean hydraulic retention time (d) (Clark 2009). The NTIS model distributes water entering the wetland inlet in accord with a residence time distribution to the outlet in the shape of a gamma function as shown in Figure 1. The mean hydraulic residence time is always less than the nominal hydraulic residence time, which is calculated by dividing wetland volume by discharge.

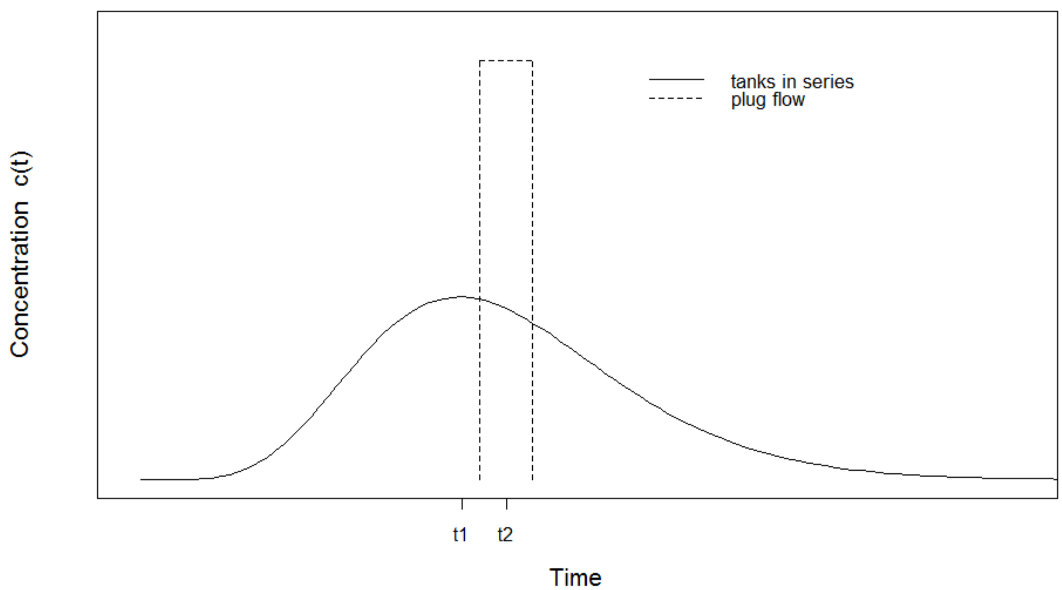


Figure 1. The residence time distribution (RTD) of a tanks in series model as compared with a plug flow model. Tracer tests are used in wetland studies to determine the residence time distribution function of individual wetlands in terms of N (number of tanks) and mean residence time. The tanks in series model residence time distribution has the shape of a gamma function, where the maximum concentration occurs at t1. The mean residence time for plug flow and NTIS models is t2.

Different wetlands have different hydraulic efficiencies depending on their morphology, plant arrangement and other factors effecting water flow patterns (Kadlec and Wallace 2009). The number of tanks in series is representative of hydraulic efficiency, with greater N values representing increased efficiency and N =∞ representing plug flow conditions (Kadlec and Wallace 2009, Clark 2009). Kadlec (2005) reviewed N values for 30 FWS wetlands, finding a mean and standard error for N of 4.5 ± 0.4.

Wetlands can be designed for higher hydraulic efficiencies and higher N values. Worman and Kronnas (2005) analyzed the effect of water conveyance, plant location and wetland shape on the level of nitrogen removal, using a modeling approach. They developed a tanks-in-series model that included nitrogen reduction and rugosity. The modeling results indicate slower flow and lower N values, could be caused by friction as water moves through plants or by variations in wetland depth. The wetland aspect (length / width ratio) is also important, with higher wetland efficiency gained at higher aspect, i.e. long narrow wetlands. In wide wetlands, water flows rapidly through a central channel and slower along the edges, increasing the spread of the residence time distribution (higher N) and decreasing contaminant removal performance. To maximize performance effectiveness, Worman and Kronnas (2005) recommend wetland designs to achieve a higher aspect ratio, i.e. length to width, which could be accomplished by partitioning the wetland into several channels such that parallel ponds are formed or by using multiple inlets and outlets. They estimated that an aspect ratio of 13.6 compared with 1.9 would result in halving the ratio of outlet concentration/ inlet concentration . Their model results show single small inlets tend to cause more channeling of the flow path and lower efficiency. Persson et al. (1999) also utilized a modeling approach to hydrology with similar findings. They recommended adding baffles to wetland ponds to increase the length of the flow path and improve hydraulic efficiency. Their model also showed that planting only on the channel edges and allowing a deeper midsection greatly diminished hydraulic efficiency and predicted the greatest hydraulic efficiency can be accomplished by a bathymetry that favors uniform flow conditions across the wetland cross section. They suggested this could be achieved by either a trapezoidal cross section or by banded bathymetry involving different depths and recommended full vegetation for both.

2.1.3. Reaction Kinetics

Kadlec's (2005) review of a database of 65 wetlands and a number of microcosm studies found the data strongly supports first order removal of nitrate. Gebremariam and Beutel (2008) confirmed first order reaction in mesocosms planted with cattails (*Typha sp.)* and bulrushes (*Scirpus sp.).* In the context of tanks in series hydrology, a first order removal of nitrate can be represented by the equation (Kadlec 2005):

$$\frac{C_o}{C_i} = \left(1 + \frac{k_a}{Nq}\right)^{-N}$$

where C_i is the inlet concentration (mg/L), C_o is the outlet concentration (mg/L), k_a is the areal reaction rate constant (m/yr), N is the number of tanks in series, and q is the hydraulic loading rate (m/yr). Calculations of the area rate constant exhibit a considerable range from near 0 m/yr to greater than 60 m/yr with a median removal rate of about 34-35 m/yr for FWS wetlands (Kadlec and Knight 1996, Kadlec 2005).

2.1.4. Temperature

Wetland performance in treating nitrate is temperature dependent, with higher conversion to N_2 at more elevated temperatures (Kadlec and Knight 1996, Vymazal 2007, Beutel et al. 2009). In a wetland study in Central Washington, Beutel et al. (2009) reported removals 2-4 times higher in summer than in cooler months. The effect of temperature on denitrification rates is described by the Arrhenius equation

$$k_T = k_{20}\theta^{T-20}$$

where k_T (day^{-1}) is the rate coefficient for Dentrification at the water temperature, k_{20} (day^{-1}) is the denitrification rate at 20 C˚, θ is the denitrification coefficient, and T is the water temperature (˚C) (Kadlec and Knight 1996). Studies of the coefficient (θ) report a range from 0.984 to 1.12 for wetlands (Kadlec and Knight 1996, Buetel et al. 2009, Kadlec 2009, Leverenz et al. 2010). Our wetland sizing uses the theta value proposed by Kadlec (2005) of 1.088 in his broad review of wetland performance, with the exception of the woodchip HSSF wetland the theta value or 1.1 defined by the research of Leverentz et al. 2010. A temperature drop of 8˚ C reduces the decay rate by about 50%.

Because nitrate removal rate is temperature dependent, wetland performance is acknowledged to be seasonally related. As an example, Spieles and Mitsch (1998) modeled nitrate removal in a constructed wetland systems receiving low nitrate concentrations (4.6 mg/L) from river water and the second receiving medium nitrate concentrations (12.4 mg/L) from municipal waste water, finding a k_v of 0.4 day^{-1} decay rate for both constructed wetlands. Nitrate removal was more efficient in the summer than other seasons due to higher outside temperature. During flooding events, nitrate removal was negative, i.e. the wetlands contributed nitrate to their surround.

2.1.5. pH

Denitrifying bacteria are most effective in a pH range 6.5 <pH < 7.5 (Kadlec and Wallace 2009). There is a diurnal pattern of pH in those areas of FWS systems with high algal production, i.e. in open waters. This pH swing is based on CO_2 utilization for photosynthesis during daylight hours and CO_2 production through respiration during night time hours. This diurnal pattern typically causes a variation from a neutral pH of 7.0 to a pH of 9.0, such that the higher pH is outside the conditions for maximum denitrification. These diel pH changes are not seen in the densely vegetated areas of wetlands but tend to occur in sparsely vegetated areas or open water areas such as deep zones, where algal growth occurs (Kadlec and Wallace 2009). The loss in effectiveness in denitrification during this cycle can be overcome by designing the wetland system to have high macrophyte density.

Wetlands with below surface water flow do not exhibit pH fluctuation and instead display consistent pH close to neutral, within the ideal range for maximizing denitrification.

2.1.6. Plant Type and Plant Location

The type, diversity and density of plants influence nitrate removal rates in wetlands. All plants, including submergent, floating, and emergent macrophytes, are beneficial compared with open water because they provide surface area for denitrifying bacteria growth and supply carbon for their respiration. The best performing species for local growing conditions on the Central Coast species has not been determined, however research findings from other locations can inform plant choice. Zhu and Sikora's microcosm study demonstrated increased removal rates in wetland cells with canary grass (*Phalaris arundinacea*) and reed compared with those planted with bulrush[2] (*Scirpus atrovirens georianus*) and cattail (*Typha latifolia*). Lin et al. (2002) found the nitrate removal rate increased by about 20% within wetland microcosm cells planted with Uganda grass (*Penniserum purpureum)* compared with cells planted with common reeds (*Phragmites australis),* day flowers *(Commelina communis),* water spinach *(Ipomea aquatica)* or water lettuce *(Pistia stratiotes).* They attributed the better performance of Uganda grass (*Penniserum purpureum)* to the release of increased amounts of organic carbon from the roots. Gebremarium and Beutel (2008) found cattail (*Typha sp.*) planted mesocosms outperformed bulrush (*Scirpus sp.*) mesocosms in nitrate areal removal rates by 25%. Similar performance differences were observed by Hernandez and Mitsch (2007) who reported higher cold water extractractable organic matter in the content of soils underlying cattails (*Typha* sp.) as compared with softstem bulrushes

[2] This genus is undergoing taxonomical revision, and this particular species may be Scirpus or Schoenoplectus.

(*Schoenoplectus tabernaemontani*). This type of organic matter is a good indicator of bioavailability and was likely related to the lower lignin content of cattails (Hernandez and Mitsch 2007). Hume et al. (2002) also explored different qualities of carbon related to different plant types in their study of the addition of different plant litters to wetland microcosms, finding a stronger correlation between acid-soluble carbohydrates C:N ratios than total C:N ratios. Similarly, Dodla et al. (2008) reported increased denitrification potential in soils containing organic carbon in the form of polysaccharides (long chained carbohydrates) as compared with phenolic forms.

The amount as well as type of plant material present for decomposition is important to denitrification and nitrate removal. Hume et al.'s (2002) computation of the first order decay when dry weight addition of litter to achieve a 2:1 ratio of C:N resulted in volumetric decay rate constants for duckweed (*Lemna* sp., $k = 0.47\ day^{-1}$), cattail (*Typha* sp., $k = 0.41\ day^{-1}$), and pennywort (*Hydrocotyle* sp., $k = 0.37\ day^{-1}$) that were about twice that of bulrush (*Scirpus* sp., $k = 0.21\ day^{-1}$). Hume et al. (2002) additionally found that higher ratios of C:N addition resulted in higher rate constants. Cattail litter addition at a ratio of 10:1 C:N had a decay rate of $1.42\ day^{-1}$ compared with an addition ratio of 2:1 with a substantially lower decay rate of $0.41\ day^{-1}$.

Unvegetated open water has low nitrate removal and low k values for nitrate removal constants (Arrheimer and Wittgren 2002), as do wetlands with woody species such as shrubs and trees (Kadlec 2005). The results of a study of denitrification enzyme activity by McGill et al. (2010) suggest that more constant denitrification over seasons and a variety of soil conditions occur when there is greater plant diversity. Due to the importance of carbon availability to nitrate reduction, Weisner et al. (2004) suggested interspersing emergent vegetation with submergent vegetation and open water areas to allow dead plant material to better reach the water column where it can be utilized by denitrifying bacteria as it decays. In a review of 65 wetlands, Kadlec (2005) concluded that fully vegetated systems with submergent and/or emergent vegetation or a banded pattern should be considered for nitrate reduction purposes in FWS wetlands. Not enough information is available locally to recommend plant types or to model their influence on decay rate, although we strongly recommend following Kadlec's suggestion for design.

2.1.7. Dissolved Oxygen (DO)

Dissolved oxygen (DO) is important to denitrification rates because if oxygen is available for facultative microbial metabolism, it is preferred over nitrate (Mitsch and Gosselink 2000). Only in oxygen poor conditions, will facultative bacteria utilize NO_3 because the redox potential of nitrate is higher and requires more energy. Dissolved oxygen is present both in the water column and in wetland soils in varying concentrations. Generally the water column of wetlands have a vertical gradient of DO, with high DO at the water-air interface and low DO at the water-soil interface (Kadlec and Knight 1996). Wetland soils also have a DO gradient, with a top aerobic layer, containing low DO, underlain by anaerobic soils (Mitsch and Gosselink 2000). Furthermore, the water column DO is influenced by plant type. Algae photosynthesis can increase DO above the saturation level (>100%) in open waters, whereas dense macrophytes block the sunlight needed for algal growth. Therefore lower DO levels are found in densely vegetated regions of wetlands.

Denitrification in wetlands occurs in the low DO areas, i.e. in the anoxic layer of soils and in the thin film on plant surfaces (Freeman and Loch 1995, Bourgues and Hart 2007). Because denitrification occurs in specific micro-habitats, it is unclear how much DO in the water column influences the rate of denitrification. The lower denitrification rates seen in open water areas could be due to higher DO, increased acidity (see pH), reduced carbon availability, lack of plant surface area for microbes or a combination of all these factors. Lin et al. (2008) noted that higher DO did not alter or inhibit denitrification in comparisons of nitrate removal rate over several trials in pilot scale wetlands. By contrast, Gebremarium and Buetel's (2008) comparison of nitrate removal in cattail vs. bulrush mesocosm found higher removal and lower DO in those with cattails. DO in the bulrush mesocosm exhibited a diurnal cycle with a range between 0.5 mg/L and 2 mg/L while DO in the cattail was always less than 0.3 mg/L . As cattails produce liter that is more easily biodegraded than bulrushes, carbon in a more available form may have driven the higher decay rates and also brought about the lower DO, i.e. as both oxygen and nitrate were used by bacteria (Gebremarium and Buetel 2008). Weisner et al. (1994) found emergent macrophyte species produce more organic material than submerged varieties, however the organic material was in a form more resistant to decomposition. Additionally the submerged vegetation produced higher DO during the daytime, which may have a retarding effect on denitrification; however, submerged plants have the advantage of increased surface area available for the development of epiphytic films. After weighting the different advantages of both plant types, Weisner et al. (1994) recommended wetland design include a pattern of shallower area with emergent species adjacent to deeper areas with submerged species.

We did not use DO as a model parameter due to the many other related variables that would need to be considered in the context of its influence on denitrification. It is more appropriate to incorporate the relevant findings regarding DO into wetland design rather than wetland modeling. As a general design principle for increasing denitrification in constructed wetlands, we recommend that the wetland be densely vegetated and absent of large open areas that would encourage algal production. Dense vegetation should diminish the diurnal pattern of highly elevated DO and acidity swings brought about by algae respiration.

2.1.8. Soil

Wetland soil is hydric soil that can be either mineral or organic (histosols) in nature (Mitsch and Gosselink 2000). Because the anoxic area of the wetland soil is important to denitrification, soil type could be an important influence on this process. However, there have been conflicting findings by different researchers regarding substrate effect on denitrification. Gale et al. (1993) found higher nitrogen removal in mineral than organic soils, however Davidsson and Stahl (2000) observed higher denitrification in peaty soil than sandy loam soil. The Davidsson and Stahl (2000) study found five soils all exhibited denitrification including 2 peat soils, sandy loam, loam and silty loam at amounts ranging from 13% to 73% of the load. Not surprisingly, their study found a positive relationship between soil organic matter and NO_3 reduction.

Dodla et al. (2008) similarly found the total organic carbon content of wetland soils was positively correlated with denitrification rates. Furthermore, their investigation of soil carbon chemistry found that denitrification was influenced by the type of carbons in the soil. Carbon structures from lignin required more energy for decomposition and reduced denitrification rate when compared with carbons from polysaccharides. This research also revealed a relationship between soil profile and the production of N_2O (a global green house gas) in swamps and marshes.

2.1.3. Wetland Maturation

Newly constructed wetlands undergo a period of maturation as the plant density increases, as the soil organic matter content changes, and as antecedent pollutants are potentially released (Kadlec and Wallace 2009). During this maturation process wetland performance is compromised as compared with a mature well vegetated wetland. Hernandez and Mitsch (2007) observed that the amount and type of carbon content of constructed wetland soil changes as the wetland ages in addition to depending on the plant type. Over a ten year period, they observed an increase in denitrification potential by a multiplication factor of 25 in surface sediment as the soil organic matter doubled. Start-up and performance stabilization can take from a few weeks to over two years and achieving optimal performance even longer (Kadlec and Wallace 2009). When wetlands are installed to achieve regulatory requirements, recognition and allowances related to the start-up period are important.

2.1.4. Carbon Addition

Several studies have shown a positive response between denitrification rate and the addition of carbon to vegetated systems (Ragab et al. 1994, Ingersoll and Baker 1998, Robins et al. 2000, Healy et al. 2006, Su and Puls 2007, Largay et al. 2008). Carbon addition is only helpful to promoting denitrification when carbon is a limitation, which can occur when nitrate concentrations are high. On the Central Coast, some waters have high nitrate concentrations and carbon addition may improve decay rates. Different sources of carbon have been added with varying responses in experimental settings. Hein et al. (2010) observed increased denitrification with the addition of both glucose and sawdust. Other carbon sources added to enhance denitrification have included straw (Ragab et al. 1994, Aslan and Turkman 2003), compost from cotton burr and mulch (Su and Puls 2007), starch (Robins et al. 2000), woodchips (Healy et al. 2006) and shredded wetland plant material (Ingersoll and Baker 1998). Largay et al. (2008) increased denitrification by adding sugar and fructose to a vegetated drainage ditch (VTS1) with high nitrate loading. They reported average inlet and outlet nitrate concentrations during sugar addition as 65.0 mg/L and 49.3 mg/L respectively for a concentration reduction of 24%, compared with reductions without sugar addition found in this study averaging 3%.

2.1.5. Decay Rate Constant

In wetland literature decay constants are represented by either areal rate constants k_a (m/yr) or by volumetric decay constants k_v (day^{-1}). Both constants are a measure of the rate of conversion of a chemical from one form to another in accord with first order decay. First order decay is exponential, with the rate of reduction determined by the constant and time by the equation

$$\frac{C_o}{C_i} = \exp\left(-k_v t\right)$$

where C_o (mg/L) is the outlet concentration, C_i (mg/L) is the inlet concentration, k_v is the volumetric decay constant (day^{-1}) and t is time (day). The only difference for the areal constant is that the exponent is divided by wetland depth

$$\frac{C_o}{C_i} = \exp\left(-\frac{k_a t}{h}\right)$$

where h is the water depth (m), k_a is the areal decay rate (m/yr) and time is expressed in terms of years. If the wetland depth is known, then conversion between the two constants is straightforward. If depth is unknown or not reported in journal articles, we assumed an FWS depth of 0.3 meters (Kadlec's 2005 estimate of a typical FWS) in order to allow comparisons between studies.

The research of Lin et al. (2008) suggests nitrate decay rates may be valid only within a range of hydraulic load and that the wetland capacity may be overwhelmed, resulting in diminished performance above a peak loading rate. Their results showed increased nitrate removal rate to a peak of 0.9 g N /m^2 d as the hydraulic load increased from 0.0 up to 0.12 m/d and a sharp decline when the hydraulic load was increased beyond this, with a diminished removal rate down to 0.7 g N /m^2 d at a hydraulic load of 0.27 m/day. The peaking at a maximum value was understandable due to limited carbon availability, however the subsequent decline is more difficult to explain. Lin et al. (2008) proposed that it could be caused by inhibition due to higher dilution of organic carbon and a short hydraulic residence time, insufficient for denitrification. It is important the wetland be designed appropriate to the hydraulic load to both maximize nitrate load removal and achieve outlet concentration goals. Kadlec (2009) reports a hydraulic loading rate ranging from 0.2 cm/day to 80 cm/day for 205 FWS wetlands, with a medina hydraulic load of 3 cm/d. For 624 HSSF wetlands hydraulic load ranged from 2 to 80 cm/d, with a median of 7 cm/d.

2.1.6. Hydraulic Conductivity of HSSF VTS

In HSSF wetlands, the water experiences friction as it moves through the bed media. Motivating water flow through the system requires gravitational force based on a sloped bottom, or could alternatively be pumped at the outlet. Hydraulic conductivity is a measure of the flow characteristics through the media expressed by Darcy's law:

$$K = \frac{QL}{A\Delta h}$$

where K is the hydraulic conductivity (cm/s), Q is discharge (cm^3/s), L is the length of the wetland, A (cm) is the areal cross section, and Δh is the drop in water level (cm) across the length of the wetland. The needed drop in elevation or degree of slope can be computed when K is known. A number of factors influence the value of K including the size and roughness of the woodchips and whether pea gravel is added to the woodchip media (Christianson et al. 2010). Woodchip media with a particle size ranging from 1.5 to 25 mm exhibited hydraulic conductivity ranging between 7.3 cm/s and 11 cm/s over a series of 5 trials. The addition of 10% pea gravel to the media increased the hydraulic conductivity to 15.4

cm/s, probably due to the greater connectivity of pores (Christianson et al. 2010). Leverenz noted that plant roots decreased compaction of wood media and probably improved hydraulic conductivity in a planted HSSF woodchip system (personal communication).

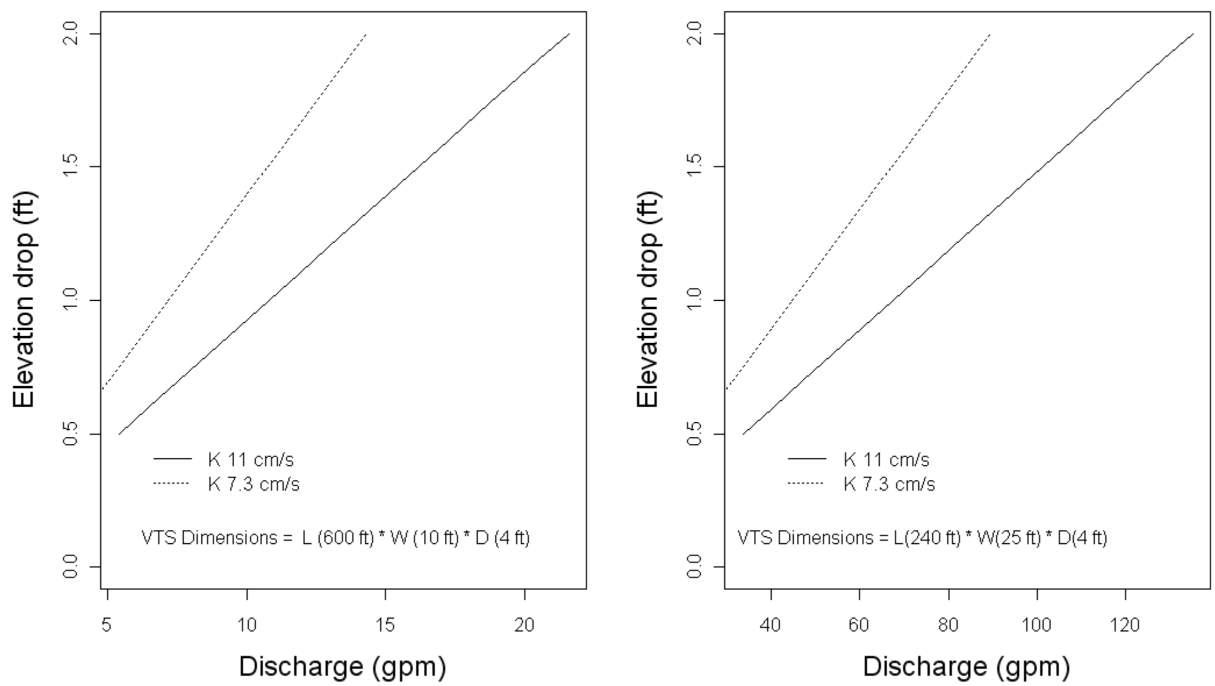


Figure 2: To create flow in HSSF systems requires an elevation drop. For a VTS of the same volume but different proportions, elevation drop is shown as a function of discharge for the range of hydraulic conductivity observed in trials by Christianson et al. 2010.

2.2. Phosphorus

The primary processes of phosphorus removal in constructed wetlands are sorption, precipitation, plant uptake and soil accretion (Vymazal 2006). Phosphorus removal in both FWS and HSSF wetlands is low unless the wetland substrates have high sorption capacity (Vymazal 2006). Kadlec (2009) noted that wetlands with more open surface water have more phytoplankton and this presence brings more phosphorus into the water column as suspended particles, thus increasing phosphorus concentration. Unless special media is chosen to increase phosphorus retention, wetlands generally are not effective at phosphorus removal (Vymazal 2011). The exception is when high concentrations of both phosphorus and suspended sediments are present in inlet waters, such that particulate settling can remove large amounts of phosphorus (Kadlec and Wallace 2009).

Kadlec's (2009) review of large data base of 282 FWS wetlands found rate coefficients for phosphorus removal ranging from 2 m/yr up to 40 m/yr with a median of 10 m/yr. The background concentration for phosphorus in FWS wetlands was 0.002 mg/L, where background concentration represents some portion of the chemical that is resistant to conversion or storage, an association with particulates, or input from groundwater or precipitation. In this same review of 82 HSSF wetlands, Kadlec found a similar range of removal rate coefficients and a median removal rate of 6 m/yr, with no background concentration (Kadlec 2009).

2.3. Ammonia

Ammonia can be carried into constructed wetlands in inlet waters and through atmospheric deposition and can also be produced in the wetland as a step in the sequence of nitrogen processing that occurs (Jordan et al. 2003, Kadlec and Wallace 2009). Ammonia, used in fertilizers in salt form or as a solution, can be present in agricultural runoff and carried to the inlet in these waters or can be volatilized. Ammonia in the atmosphere may be a significant source, as was discovered by Jordan et al. (2003) in their study of a restored wetland in an agricultural watershed in Maryland. This wetland received 30% of its ammonia from atmospheric deposition during precipitation events. Ammonia is produced in wetlands by the ammonification of organic nitrogen and is a product of protein metabolism. For example, ammonia as a byproduct of fish metabolism is a concern of toxicity in aquariums. For these various reasons, wetlands may demonstrate negative removal rates for ammonia, as occurred in 30 out of 208 wetland reviewed by Kadlec and Wallace (2009). Determining ammonia removal rates is further confounded by the low method detection limits for analytical measurement, which can skew the value of low inlet concentrations when they fall below this limit (Kadlec and Wallace 2009).

Transformation of ammonia to other nitrogen forms in wetlands is primarily accomplished by nitrifying bacteria, which convert nitrogen to its oxidized forms as NO_3 or NO_2 (Kadlec and Wallace 2009). The process of nitrification releases H^+ molecules in water and lowers the pH of the water. The relationship of ammonia and ammonium is largely influenced by temperature and pH, with lower pH driving the equilibrium toward higher relative amounts of the ionized form (ammonium NH_4^+) in accord with the chemical equation:

$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^-$.

The ionized form (NH_4^+) is not toxic, whereas the unionized form (NH_3) is toxic at low concentrations. The nitrate or nitrite produced by nitrification of ammonia subsequently undergoes denitrification in the wetland as described previously.

Kadlec and Wallace (2009) recommend the use of the PTIS model for ammonia removal in FWS wetlands. This model is similar to the number of tanks-in-series (NTIS) model discussed earlier, modified by a weathering of the decay rate value. Weathering occurs as water containing a mixture of compounds, such as the various compounds associated with total nitrogen (TN), pass through the wetland and change composition. The net result in the model is that the "apparent" number of tanks is lower than that found during tracer tests, however the formula described in Section 2.1.2 Hydrology is basically the same. To determine the change in concentration for weathered contaminants, rather than determining N by a tracer test, it is renamed P and determined by fitting the data to the model. Both parameters, P and $k_{v.}$ become fitted values in this model (Kadlec and Wallace 2009).

Ammonia reduction in HSSF is typically low because all mechanisms of ammonia reduction require oxygen and oxygen is limited in HSSF systems (Kadlec and Wallace 2009). Nitrification rates compiled from 6 studies ranged from 1.2 g/m^2-d up to 6.8 g/m^2-d with a mean of 3.09 g/m^2-d (Kadlec and Wallace 2009).

2.4. Sediment

Total suspended solids (TSS) or sediment is removed from the water column in wetlands by the processes of settling and filtration, which occurs through slowing the flow rate of water so the sediment drops out. Wetlands do not typically remove all of the TSS, as suspended sediment can be added by internal wetland biochemical processes and resuspension (Kadlec 2009). In FWS, removal occurs when the amount removed exceeds the amount added, which normally occurs at higher loading rates such as those observed in farm runoff. Kadlec (2009) reviewed USEPA wetland data sources and calculated that given a TSS load of 5 gm/m^{2}d, the median outlet concentration based on this data source would be about 20 mg/L.

Incoming TSS is rapidly settled in wetlands and therefore installing a pre-treatment sedimentation pond ahead of HSSF wetlands or a deeper area near the inlet to trap the fastest settling particles in FWS systems is recommended (Kadlec and Wallace 2009). Occasional dredging will be necessary to remove accumulated sediment (Kadlec and Wallace 2009). Because HSSF wetlands are designed for water flow through a bed media, the TSS associated with inlet flows can be retained in the wetland and clog the pore space, causing a reduction in hydraulic conductivity and loss of treatment effectiveness. Sediment accumulation in HSSF wetlands can lead to failure of the bed media and flooding bypass of wetland treatment, so prior removal by a sediment pond when loading is high is important. The TSS outlet concentration in HSSF wetlands is not related to the TSS inlet concentration, as over a wide range of inlet concentrations, the average outlet concentration of 42 mg/L was found for a large number of wetlands from wetland data bases (CWA and WERF), with a 90th percentile outlet limit of 42 mg/L Kadlec and Wallace (2009).

The model recommended for computing removal of TSS in FWS wetlands is a linear regression with an R^2 of 0.65 for logarithmic data (Kadlec and Wallace 2009). There are a complexity of factors related to TSS removal, resuspension and generation in wetlands and which could be modeled as processes. However, the model recommended by Kadlec and Wallace (2009) for predicting outlet concentrations of FWS wetlands is the simple linear regression equation

$$C_o = 1.5 + 0.22\ C_i$$

they developed by fitting a least square regression line to the performance of 142 wetlands where C_o is the outlet concentration (mg/L) and C_i is the inlet concentration (mg/L). These authors also recommend a linear regression equation for computing load reduction of TSS

$$C_o = 2.1 + 3.8\ L_i$$

where L_i is the inlet load (g/m2-d) and 0.22 is a constant [(mg/L)/(g/m2-d)].

3. Wetland Sizing Model

The wetland sizing model estimates the area of a wetland needed to remove nitrate from the water column and achieve a desired outlet concentration based on input variables and working model parameters. The model is written using R statistical software (R Development Core Team). Table 2 shows the input parameters identified and entered into the model by the grower, designer or engineer and the working variables, which were defined through the literature review and analysis of data from local wetlands. The model estimates the wetland water area coverage, however the total area requirement may be greater as the design may require the development of berms and maintenance access paths or roads.

3.1. Model Description and Mathematical Derivation

Our wetland sizing model is founded on the assumption of first order decay of nitrate and a number of tanks in series (NTIS) hydrology, based on the model proposed by Kadlec (2005) for wetland performance:

$$\frac{C_o}{C_i} = \left(1 + \frac{k_a}{Nq}\right)^{-N}$$

where the C_i is inlet concentration (mg/L), C_o is the outlet concentration (mg/L), k_a is the aerial removal rate constant (m/d), N is the number of tanks in series (dimensionless), and q is the hydraulic loading rate (m/d). We incorporated the effect of temperature on wetland performance in accordance with the Arrhenius equation

$$k_a = k_{20}\theta^{T-20}$$

where T is the water temperature (˚C) and k_{20} is the reduction rate constant (d^{-1}) at 20˚C. Furthermore, we used the relationship between hydraulic loading rate and discharge to provide an area parameter by the equation

$$q = \frac{Q}{A}$$

where Q is discharge (m^3/d) and A is wetland area (m^2). By substitution and rearrangement of these equations, using Wolfram Alpha Mathematica, we arrived at the model for estimating wetland area as

$$A = -N\left[\left(\frac{C_o}{C_i}\right)^{\frac{1}{N}} - 1\right]\left(\frac{C_o}{C_i}\right)^{-\frac{1}{N}}\left(\frac{Q}{k_{20}\,\theta^{T-20}}\right)$$

or in the case of chemical "weathering," including changes between different forms of Nitrogen compounds as

$$A = -P\left[\left(\frac{C_o}{C_i}\right)^{\frac{1}{P}} - 1\right]\left(\frac{C_o}{C_i}\right)^{-\frac{1}{P}}\left(\frac{Q}{k_{20}\,\theta^{T-20}}\right).$$

The model parameters identified by the users are inlet concentration, desired outlet concentration, water temperature, discharge and wetland type (i.e. FWS or HSSF). The model variables based on a literature review are the number of tanks in series (N or P) and theta (θ). The value used in the model for theta (θ=1.088) is recommended by Kadlec (2005) in his wetland review, and for woodchip HSSF (θ=1.1) was computed by Leverenz 2010.. The number of tanks in series values for denitrification in FWS wetlands (P = 3) is taken from Kadlec's (2009) median value for 72 wetlands. It should be noted that FWS systems designed for hydraulic efficiency can have a much higher N or P. A tracer study of the Molera Wetland near Moss Landing determined an N=16.5 for this wetland (personal experiment). This relatively high value is due to its design as a long sinuous channel and high length to width ratio (Harris et al. 2007). Designing wetlands for hydraulic efficiency can reduce the area requirements, however at this time the model does not allow for adjustments. At a later time we may modify the WSM to incorporate N or P as a variable that will depend on the wetland morphology chosen by the user. The value for the number of tanks in series (P=6) for HSSF wetlands comes from Kadlec's (2009) review of 216 HSSF wetlands.

We also have included a safety factor (SF) in the model which is a multiplier of 1.8 recommended by Kadlec (2005) to account for stochastic wetland performance. He recommended this factor to safe guard against irregularities in performance, especially when outlet concentration must conform to regulatory requirements. The equation incorporating the safety factor for stochasticity is

$$SA = SF * A$$

where SA is the wetland area (m^2) modified to account for stochastic performance and SF is the stochastic factor (dimensionless) related to an increase in size to avoid non-compliance with outlet concentration goals.

Table 2. Wetland sizing model variables. Note that the reduction constant is not identified as a single value. Based on a literature review, the range of k values as a probability distribution for FWS and HSSF wetlands are used in the model. A range of values for HSSF woodchip wetlands from the literature are used and the local values from a vegetated ditch.

WSM Input Parameters	Symbols	Value	Units	Source
Inlet Concentration	Ci	User Defined	mg/L	User
Outlet Concentration	Co	User Defined	mg/L	User
Volumetric Water Inflow	Q	User Defined	gallons/min	User
Water Temperature	T	User Defined	deg. C.	User
Wetland Type (Listed Below):		User Choice		
Free Water Surface				
Horizontal Sub-surface Flow				
Horizontal Sub-surface Flow with Woodchips				
Vegetated Ditch				
WSM Defined Variables				
Stochastic Variability Factor	SF	1.8	unitless	Kadlec 2005
Arrhenius Temperature Theta (FWS,HSSF,VegDitch)	θ	1.088	unitless	Kadlec 2005
Arrhenius Temperature Theta (Woodchip HSSF)	θ	1.1	unitless	Leverenz et al. 2009
Hydraulic Efficiency FWS	P	3	unitless	Kadlec 2005
Hydraulic Efficiency HSSF	P	6	unitless	Kadlec 2009
Hydraulic Efficiency Vegetated Ditch	N	11	unitless	Largay et al. 2007
Wetland depth for Woodchip HSSF	h	4 & 8	ft	
Areal First Order Reduction Constant	k_{a20}	*	m/yr	Literature and local
Volumetric First Order Reduction Constant	$k_{v\,20}$	*	day^{-1}	Literature
WSM Output Parameters				
Estimated Wetland Area	A	Output	acres	Model Output
Area (with stochastic factor multiplier)	SA	Output	acres	Model Output

The reaction rate constant (k_a) has a considerable effect on the size of treatment wetland needed. There is 50-fold difference in the area needed between the worst (k_a = 2 m/yr) and best performing (k_a = 100 m/yr) FWS wetland treating 45 ppm nitrate as N inlet water to an outlet concentration of 10 ppm. Because there is considerable performance variation between wetlands with a large range of k values reported in the literature, we developed probability density distributions for FWs and HSSF wetland types. This provides a more robust model and allows the user to evaluate the uncertainty related to areal needs. Kadlec's compared 72 FWS wetlands and 216 HSSF wetlands and charted the percentiles associated with denitrification rate for each wetland type. From Kadlec's graph of percentiles, we used a Monte Carlo method based on the decay rate probability distribution. The model randomly draws 10000 k_a values from the probability distribution function and displays a histogram of the wetland areas associated with these k_a values (Appendix B). The cumulative probability distribution for FWS wetlands that showed the best fit with the Kadlec percentile vs. decay plots was a gamma distribution. The shape and scale parameters of the gamma distribution were fitted parameters that achieved the least sum of squared error between the frequency plot and the gamma function (Fig. 2). The cumulative probability distribution for HSSF wetlands is based on 10000 random draws between the upper and lower decay rate for each of the 10% intervals.

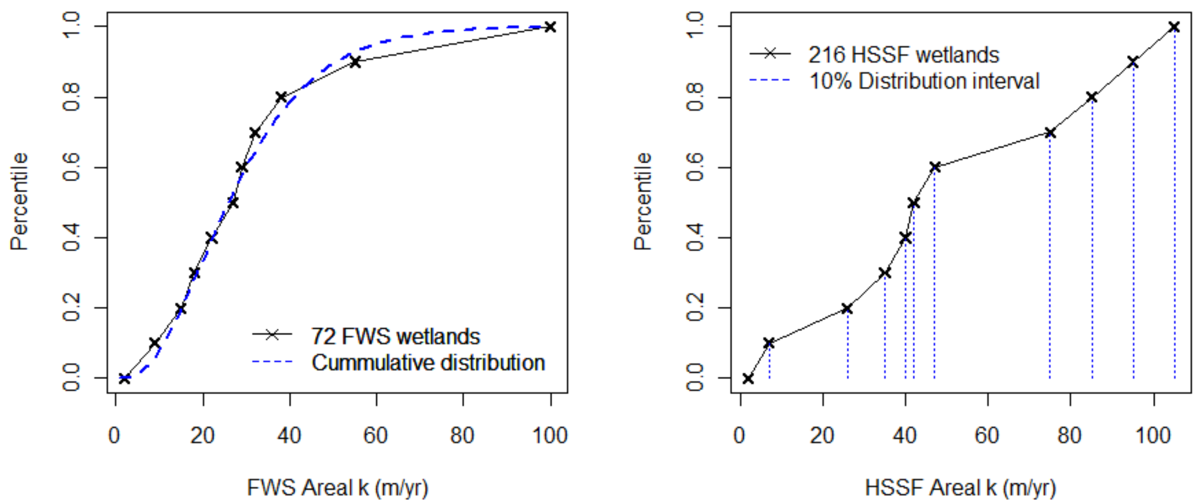


Figure 3. Distributions of FWS and HSSF k_a values modeled are based on percentile graphs of denitrification rates from Kadlec (2009) . Based on the Monte Carlo method, the model randomly draws 10,000 decay rate values from the performance probability distribution over a large number of wetlands. The distribution of k_a values for FWS wetlands uses Monte Carlo random draws from a gamma function. The HSSF wetland k_a values come from Monte Carlo uniform random draws between values at each percentile.

Substitution of woodchip media in place of gravel in HSSF wetlands is a more recent development, first researched by Blowes et al. (1994) in 55 gallon containers. Whereas treatment wetlands have been used for over 50 years for wastewater decontamination (Vymazal 2010), woodchip HSSF have been researched for less than 20 years. Because limited data on HSSF woodchip media denitrification rates is available, a probability distribution for sizing estimates could not be determined due to the absence of a sufficiently large number of wetlands. Furthermore, due to the small number of systems currently operating, there is uncertainty regarding whether first order, zero order or a monad reaction is most relevant to performance. Much of the literature reports reaction rates in zero order terms (Robertson 2010 , Greenan et al. 2009, Schipper et al. 2010), however Leverentz et al. (2010) suggested a better fit from first order rates and identified the need for further research to validate reaction type. A range of volumetric first order decay rate constants from 0.25 to 2.4 day^{-1} have been identified in laboratory and field settings (Leverentz et al. 2010, Robertson 2010, Moorman et al. 2010); however some of these rates were determined in experiments at temperatures greater than 20 deg. C , thus elevating the reaction rate constant (Robertson 2010). Research by Schipper et al. (2010) proposed that reaction rate calculations reported may be artificially low in cases when nitrogen is limited, because the full nitrogen removal capacity is not evaluated. Due to the lack of sufficient number of wetlands to develop a probability distribution for HSSF woodchips, the model instead displays a range of sizes based on the values identified by Moorman et al. (2010) for two in-field woodchip systems in operation for nine years (0.86 and 1.08 day-1), by Leverenz et al. (2010) for a woodchip wetland in operation for two years (1.4 day^{-1}), and the low and high values from a series of trials on media extracted from field woodchip systems (0.25 and 2.2 day^{-1}) by Robertson (2010).

Woodchip media can be used for denitrification without plants, with only slightly lower denitrification rates (Leverenz 2010). A primary value of plant presence is root growth around the woodchip media, which provides structure to the bed and prevents compaction, thus improving hydraulic conductivity through the system (Leverenz 2010). The depth of HSSF with woodchip media is not limited due to suitable conditions for plant growth, as is the case with FWS and HSSF. Whereas FWS depth is limited by plant tolerance of water level and HSSF by root penetration into the media, the woodchips in a woodchip HSSF provide both surface area and carbon for the metabolism of denitrifying bacteria. Therefore woodchip HSSF depth is a design parameter that can be determined based on site considerations. Treatment of discharge can be accomplished by increasing the depth without increasing the area, thus displacing less farmable land. For this reason, the model displays areas calculated based on two different wetland depths (4 and 8 feet). The user can linearly interpolate areas at other depths. To incorporate depth into the model, denitrification rate is calculated in terms of a volumetric constant rather than an areal constant by

$$A = -P\left[\left(\frac{C_o}{C_i}\right)^{\frac{1}{P}} - 1\right]\left(\frac{C_o}{C_i}\right)^{-\frac{1}{P}}\left(\frac{Q}{hk_{v20}\,\theta^{T-20}}\right)$$

where the k_{v20} (day^{-1}) is the volumetric reaction rate and h (m) is the wetland depth. HSSF woodchip wetland depth may be limited by the depth of groundwater or ability of the media to withstand compaction, although construction of the system within a liner may relieve constraints of a high water table. We assumed the P (P=6) to be the same as for other HSSF wetlands, and the same stochasticity factor of 1.8 recommended by Kadlec (2005) for other wetland types.

In addition to reviewing decay rate constants in scientific literature, we determined nitrate decay rate constants for local VTSs based on field monitoring over a 2-3 year period (Table 3). VTS1 and VTS2 are vegetated ditches. VTS3 is a sedimentation pond populated with floating macrophytes with a higher than normal depth (2 m), about 6.5 times the typical FWS wetland depth of 0.3 meters (Kadlec 2009). VTS4 is a new wetland that has not reached maturity and therefore has a relatively low k value compared with anticipated future performance when increased carbon becomes available to fuel denitrification (see Section 2.1.8). We used the range of calculated decay rates from VTS1 in the model for sizing estimates for vegetated ditches. The inconsistent performance between years for VTS2 makes the values uncertain and unreliable. This inconsistency is most likely due to the large hydraulic load, which results in large calculated k-value differences from small differences in inlet and outlet concentrations. The model for vegetated ditches selects all three values from VTS1 and shows a barplot of estimated acreage for each.

Table 3: Derived k_{a20} values for local vegetated treatment systems. VTS1 and VTS2 are vegetated drainage ditches. VTS3 is a vegetated sediment pond. VTS4 is a recently installed FWS wetland. VTS4 decay value is anticipated to increase as the system generates more carbon for denitrification.

	k areal (m/yr)		
	2007	2008	2009
VTS1	11.1	20.3	13.9
VTS2	-35.0	49.0	
VTS3*		0.4	1.1
VTS4		-0.7	2.8

The model was developed in R (R Development Core Team 2011) and is made available on CDs, and the website http: <add later >. Users will need to download R onto their computer in order to make use of the model and install two packages (gWidgets and gWidgetsRGtk2) from a CRAN mirror site. A pdf document describing how to install R, load Packages and input user parameters into the model is included. The R code for the model is shown in Appendix A.

3.2. Example of Model Results

If we compare the area needed for an HSSF wetland to reduce nitrate concentration from 45 mg/L to 10 mg/L with a discharge of 20 gpm, considering two denitrification areal decay rates of 20 m/yr and 42 m/yr, the respective areas required for treatment would be 0.7 acres and 0.4 acres. The assumptions are listed in the inputs and constants columns in Table 3. The model output is displayed for each constant, with area shown in different units for convenience. If no variation and constant wetland performance is assumed, the area is determined by the model inputs in the absence of stochastic considerations and is shown in Table 3 as Area (A). However, wetland performance is not consistent so Kadlec (2009) recommended a stochastic effect (SF) be used to estimate wetland size to incorporate variable performance and not exceed regulatory goals. This Stochastic Factor of 1.8 is multiplied by the area to incorporate natural variation in performance as a safety factor and is shown in Table 3 as Area plus Safety Factor. Table 4 shows the estimated area requirements for the four different wetland types based on the same inputs.

Table 4. Comparison of model results considering two different decay rate constants for nitrate. The wetland area needed for treatment is greater for the lower decay rate.

WSM Inputs			
Input Variable	**Model Name**	**Amount**	**Units**
Inlet Concentration	ConcIn	45	mg/L
Outlet Concentration	ConcOut	10	mg/L
Water Temperature	H2O_Temp	18	deg C
Flow Rate	Q	20	gpm
Wetland Type:	Horizontal Sub-surface Flow		

WSM Outputs k_a = 42 m/yr			WSM Outputs k_a = 20 m/yr		
Output Variable	**Amount**	**Units**	**Output Variable**	**Amount**	**Units**
Concentration Reduction	77.78	%	Concentration Reduction	77.78	%
Load Removed	2.35	g/m2-d	Load Removed	1.12	g/m2-d
Estimated Area	1620.7	m2	Estimated Area	3403.4	m2
Estimated Area	0.40	acres	Estimated Area	0.84	acres
Area plus Safety Factor	2917.26	m2	Area plus Safety Factor	6126.12	m2
Area plus Safety Factor	0.72	acres	Area plus Safety Factor	1.51	acres

Table 5. WSM outputs for all four wetland types using median nitrate decay rates and the same inputs as Table 4. The HSSF woodchip wetland requires the least area and the vegetated ditch the greatest.

Wetland Type	Median Model Denitrification Rate	Estimated Area (Acres)	Area plus Safety Factor (acres)
Free Water Surface	27 m/yr	0.842	1.516
Horizontal Sub-surface Flow	42 m/yr	0.474	0.853
Horizontal Sub-surface Flow with Woodchips (4 ft deep)	1.2 day^{-1}	0.062	0.112
Vegetated Ditch	13.9 m/yr	1.351	2.432

Model results from all four wetland types show that horizontal sub-surface flow wetlands with woodchip media require the least area. Given these inputs, the estimated area of the next smallest wetland (HSSF with gravel media) was greater than 7 times the size of the HSSF with woodchip media. The highest area requirement was for vegetated ditches, with decay based on the rate constants derived from local vegetated ditches.

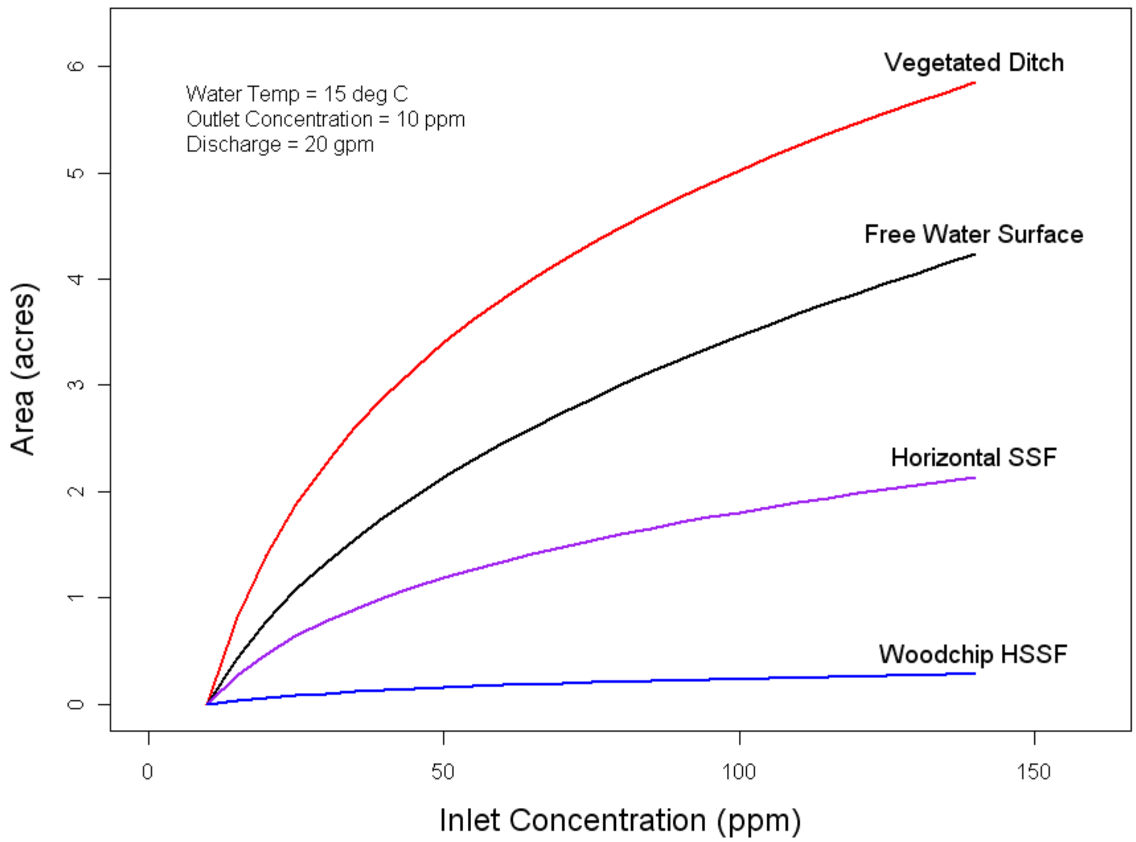


Figure 4. Area requirements for different wetland types at different inlet concentrations. Based on the median decay rate for each type show Woodchip HSSF require the least area. The relatively large area for FWS and HSSF wetlands, could be lessened by designing them specifically for nitrate reduction. Woodchip HSSF area was calculated based on 4 feet depth.

Denitrification rate is temperature sensitive (see Section 2.4.1 Temperature) and the user is required to input the water temperature into the model. Wetland water temperature varies on both an annual and diurnal cycle. The daytime water temperatures of the local vegetated treatment systems we researched are displayed in Appendix C. These water temperatures can be used as a reference for the range of temperatures the user could input into the model. Also available as a reference is the CCAMP data browser at the website: http://www.ccamp.info/_2010/view_data.php. In the data browser from the draw down menus on the CCAMP website, the user should choose the watershed relevant to the location, the data type “Basic Water Quality”, and the Analytes “Water Temperature”. The data browser will then show a display of the sampling locations in the designated watershed and a chart with the temperature range, median and mean at each location. We recommend choosing a stream or creek, rather than a river, close to the potential wetland installation site for water temperature input to the model.

4. Wetland Design and Evaluation Principles

4.1. Design Principles for FWS Wetlands

In addition to the wetland model, there are design principles that could be useful to improve FWS wetland performance for nitrate removal[3] .

1) Maximize FWS wetland hydrologic efficiency through a high length to width ratio (greater than 10), adding berms to create separate wetland compartments, placement of baffles, or use of multiple inlets and outlets (Persson et al. 1999, Worman and Kronnas 2005). Avoid planting only at the wetland edges as this creates hydrologic inefficiency (Persson et al. 1999).
2) Increase the ease of sediment removal as a wetland maintenance activity by employing either a pre-treatment sedimentation pond or constructing a deeper sedimentation area near the inlet to collect sediment. Design the wetland for access to the sediment removal area. Refer to the NRCS (2008) Practice Code 638 for Water and Sediment Control Basins, which recommends sufficient capacity to control run-off from a 10 year frequency storm of 24 hour duration and to store 10 years of sediment accumulation.
3) Vegetation should fully cover FWS wetland area and large open areas promoting algal production should be avoided (Kadlec 2005). We recommend either planting in a banded pattern with the use of both submergent and / or emergent vegetation or planting for full coverage using emergent plants (Weisner et al. 2004, Kadlec 2005). Although the scientific literature does not specify a percent coverage by emergent plant type, we propose about 70% or higher emergent plant coverage.
4) Vegetation choice should be based on plants that grow well in local conditions and will become established with low maintenance (Kadlec and Knight 1996). Plants with higher carbohydrate and lower lignin content [e.g. cattails (*Scirpus* sp *or Schoenoplectus* sp*)* are preferred over

[3] Principles may be modified or added to based on additional information..

bulrushes (*Typha* sp.)] are more bioavailable for microbial denitrification and preferred for maximizing nitrate removal (Hernandez and Mitsch 2007).

5) Inhibition of nitrate reduction may happen above a peak hydraulic or nitrate loading rate, however not enough research exists to validate or quantify the peak for different wetland types. An FWS wetland researched in Lin et al. (2008) was inhibited at hydraulic loading rates above 0.12 m/d.
6) Kadlec (2005) recommends increasing the estimated constructed wetland area by a factor of 1.8 when achieving regulatory goals is important. This is due to normal stochastic performance.
7) Addition of a carbon source (e.g. hay, woodchips, mulch, shredded plant material, sawdust) to FWS wetlands can increase denitrification rates (Ragab et al. 1994, Ingersoll and Baker 1998, Healy et al. 2006, Su and Puls 2007, Largay et al. 2008).
8) Wetland depth in FWS systems is based on plant growth, with an average depth of 0.3 meters. Increasing FWS wetland depth to gain more capacity will not increase the volume of water that can be treated (Kadlec and Wallace 2009).
9) Wetland area estimates can be substantially influenced by water temperature (Kadlec 2005); therefore to be conservative in estimating size, choose water temperatures in the low range for your location. Or if you have seasonal nitrate concentrations, estimate wetland size separately for each season using seasonal water temperatures and inlet nitrate concentrations.

4.2. Design Principles for HSSF Woodchip Wetlands

1) Hydraulic conductivity is an important design consideration for HSSF wetlands, including those with a woodchip substrate. Use Darcy's law to determine the required slope to create a positive flow (Christianson et al. 2010, see Section 2.1.13).
2) Woodchip media has a life of 10-20 years and will need to be replaced periodically (Moorman et al. 2010, Robertson 2010, Christianson and Helmers 2011).
3) Storm water runoff may need to be diverted around the system due to hydraulic limitations.
4) Woodchips should be ¼ to 1 inch in size and should not have a high sawdust content (< 10%) (Wallace and Knight 2006, Christianson and Helmers 2011).
5) Avoid use of treated or preserved wood for woodchips as these may limit denitrifying bacteria (Christianson and Helmers 2011).
6) Sediment can clog pore space between woodchips and should be removed in advance of the VTS system. Use of a sediment control basin is recommended when runoff contains sediment.

Refer to the NRCS (2008) Practice Code 638 for Water and Sediment Control Basins, which recommends sufficient capacity to control run-off from a 10 year frequency storm of 24 hour duration and to store 10 years of sediment accumulation.

7) During low flow periods the woodchip VTS should be monitored for hydrogen sulfide (rotten egg smell) and the outflow structure should be lowered to maximize flow through in order to prevent production of hydrogen sulfide and methyl mercury (Christianson and Helmers 2011).
8) Increasing the depth of Woodchip HSSF systems to decrease area requirements may be feasible, depending on site considerations and costs such as groundwater depth. (Leverentz personal communication).
9) Woodchip VTS can be installed without plants and will successfully remove nitrate with only slightly lower removal rates (Leverenz 2010).

4.3. Evaluation Principles

1) Achieving optimal treatment performance can take more than two years for FWS and HSSF wetlands. Evaluation of performance by regulators and other stakeholders should allow for maturation time (Kadlec and Wallace 2009).
2) During the first 3 months, effluent from Woodchip HSSF wetlands can contain high concentrations of dissolved organic carbon (DOC) as highly soluble lignin is dissolved (. This may cause high biological oxygen demand (BOD). During this period, the amount of dilution by other receiving water should be evaluated and
3) Further research of local wetlands installed for the purpose of nitrate removal on California's Central Coast can enable better calibration of the decay rate constants for each wetland type and validation of the model.

5. Summary of Field Data Collected

Field data was collected for four vegetated treatment systems located on the Central Coast of California in the Salinas and Pajaro River Watersheds over a period of 3 years from 2007 through 2009. Data is summarized seasonally as wetland performance is generally higher in the warmer summer months. For the purposes of this review, seasons were defined as two seasons per year:

- Winter/ Spring = Jan 1 - May 31
- Summer = June 1 - Oct 1

VTS sites were sampled by the permission of the growers on the farms where they were located. Not all VTS systems were sampled for all three years, as VTS4 was not in place for sampling in 2007 and access to VTS2 was not granted in 2009.

VTS sites were primarily chosen for this study because they received drainage water from agriculture. This was important to accomplishing the purpose of the study, i.e. to determine local performance and sizing needed for treatment effectiveness. Growers and landowners also had to grant permission to conduct VTS research on their farms. Socio-economic and food safety concerns are reasons some growers do not want to incorporate VTS systems near agricultural production. More information regarding socio-economic concerns encountered and considered by owners and growers in installing VTS systems are addressed in the Largay et al. (2009) review of many of the same treatment systems.

The most economical form of vegetated treatment is the use of pre-existing drainage systems, with the addition of vegetation to induce treatment. Less economical is the construction of a new system or the increase in land area coverage of an existing system. Three of the four systems we reviewed involved the least cost approach, i.e. addition of plants to drainage ditches (VTS1 and VTS2) and to sediment ponds (VTS3). The problem with the least cost approach is that the VTS systems were undersized to accomplish the load reduction needed to comply with regulatory standards for nitrate. Largay et al.'s (2009) review of these three systems concluded the annualized nitrate loading rate for VTS1 andVTS2 were very high, greater than the median values reported in the literature by a factor of 10 and 3 respectively in 2008. VTS4 was a newly constructed wetland that was built on a working farm, with the grower sacrificing 0.5 acres of productive land to accomplish water quality improvements. The performance of new wetlands is compromised by lack of organic matter to contribute carbon for microbe utilization in using nitrate (Kadlec and Wallace 2009), and although VTS4 demonstrated performance improvement in 2009 over 2008, it has likely not yet reached maturity.

Field measurements, sampling and analysis were done in accordance with the Quality Assurance Project Plan (QAPP). Water quality field measurements were taken with a Hydrolab Sonde and included dissolved oxygen (mg/L), water temperature (°C), pH, specific conductivity (uS/cm), total dissolved solids (g/L), and turbidity (NTU). Nutrient samples were collected and analyzed on a Lachat in accord with the QAPP for the project. Nutrients analyzed included nitrate+nitrite (mg/L), ortho-phosphate (SRP; mg/L), and ammonia (mg/L). Ammonium was calculated from ammonia based on pH and water temperature.

Total suspended sediment samples were collected and analyzed using EPA method is 160.2 "Residue, Non-Filterable (Gravimetric, Dried at 103-105°C)".

5.1. Geographic Location and Site Descriptions

VTS1: Vegetated Drainage Ditch

VTS1 drains both stormwater in the winter months and tile-drained water from approximately 200 acres of irrigated row crops. This system primarily treats tile drain water from a conventional, sprinkler/drip irrigated, mixed vegetable row crop farming operation. However, this ditch also receives some flows from upstream neighbors and stormwater as noted. The entire length of VTS1 Section 1 is 210 m (700 ft) long by 2.1 to 3 m wide by about 0.3 m deep. There is a second section of vegetated drainage ditch for VTS1 downstream of Section 1 after a culvert for a farm road. VTS1 Section 2 is 235 m long with the same width and depth dimensions as Section 1. This system was installed by the RCDMC. Vegetation, hydrolysis, photolysis, and retention time are several key variables that facilitate water quality improvements. Although originally planted with pennywort, other floating vegetation consisting of watercress (*Nasturtium* sp.), pennywort (*Hydrocotyle* sp.), duckweed (*Lemna* sp.), and ditchgrass (*Ruppia* sp.) have become established in VTS1. There were some grasses and one 6 m (20 ft) section of calla lilies close to the water's edge. During both sampling seasons partial die off of the vegetation occurred during the late July through mid-August, perhaps due to herbicides travelling through the ditch. A preferential. flow path developed in the treatment ditch during this same time, allowing for partial bypass of treatment.

Largay et al. (2008) evaluated the hydrology of the VTS ditch by discharge and tracer test measurements with detailed descriptions available in the report to the Central Coast Regional. Water Quality Control Board named "Vegetated Treatment Systems at the Edge of Working Farms: A Tool to Reduce the Yield of Nutrients, Pesticides and Sediment, Monterey County, California." Flow rates in the ditch in 2007 and 2008 ranged from 2 to 24 L/s over the course of the study period. Flow could not be measured in 2009 because the weir system was removed for reasons elaborated in the earlier Largay report. Flow at the outlet was less than inlet flow by about 20%, probably indicating infiltration to ground water. Evapo-transpiration was only 1-2% and was considered to be negligible. The average hydraulic loading rate was estimated at 64 cm/day. A tracer test was conducted and the parameter for tanks in series N was 10 (Largay et al. 2008).

VTS2: Vegetated Drainage Ditch

This vegetative treatment system was implemented in a segment of an already existing agricultural drainage ditch that bisects a large, cut flower, greenhouse nursery operation in the Pajaro River watershed. This greenhouse operation is home to nearly 20 greenhouses. This grower worked with the Natural Resource Conservation Service in 2006 and 2007 to implement a water recycling/reuse program. They recycle and reuse most of their water, however they still have some greenhouse runoff - both tile drained and roof runoff resulting from condensation and stormwater. However, the most significant runoff originates from upstream conventional row-crop agricultural production. All of this runoff is intercepted by a 1000 LF ditch channel that bisects their property. VTS2 was planted with floating pennywort. The pennywort was very well established throughout sampling.

Largay et al. (2008) provided detailed descriptions of size flow and hydraulic load calculations for VTS2 for 2007 and 2008 available in their report to the CCRWQCB. The area required for VTS2 is 0.35 acres and it holds about 0.06 acre-ft of water. Flow at VTS2 varied between 0.1 and 5 L/s during periods of no precipitation. Infiltration to ground water was estimated at 19%. The average hydraulic load was 76 cm/day. Precipitation during the tracer test made determination of the N for this VTS infeasible, however the average residence time of 2.5 days was determined for an inflow discharge of 0.35 L/s (Largay et al. 2008).

VTS3: Vegetated Sediment Detention Ponds

Vegetated Treatment was established in dual, irrigation tailwater recovery ponds to make VTS3.
Tailwater recovery ponds are located downstream of approximately 450 acres of sprinkler irrigated row

crop mixed vegetables. Irrigation runoff from production blocks is conveyed through a series of shallow, v-shaped, agricultural drainage ditches to the dual system tailwater ponds.

Irrigation runoff water enters the system through the upper pond. In this pond sediments settle out near the inflow. Water fills the pond, and is filtered through a dense mat of floating pennywort, a California native riparian species known to float on the waters service and form a dense interwoven mat. This water then leaves the upper pond and enters the lower pond through a small corrugated steel pipe connecting the two ponds.

Hydrological evaluation of VTS3 is explained in the Largay et al. (2008) report to the CCRWQCB, however it is named VTS4 in the earlier report. VTS3 covers 1/3 of an acre, including the two ponds and perimeter road. The ponds are 2 meters deep, however pond depths change by about a foot as variable inlet discharge enters and drains through the system. The theoretical. residence time for the 2-pond system was estimated was 14 days based on calculation of Volume/Q Inflow; however the actual mean residence time is probably half that or less (Largay et al. 2008).

VTS4: Newly Constructed FWS Wetland

This system was composed of two constructed wetlands implemented with technical assistance and funding from the Natural Resource Conservation Service and the RCDSCC. These shallow tailwater recovery ponds together have a 0.5 acre foot print. The VTS is located downstream of 160 acres of sprinkler and drip irrigated organic and conventional row cropped, mixed vegetables. Irrigation runoff from production blocks is primarily tile drained. Tile drained water fills the 1st pond. This water then leaves the 1st pond and enters the 2nd pond through a small culvert connecting the two ponds. Both wetland have approximately the same dimensions of 70 m long by 12 m wide by 0.5 m deep in the center. Their water depth can be varied through the use of a weir structure at the outlet of the second pond, however their approximate combined water capacity is 500 m^3 (0.4 acre-ft) based on a trapezoidal cross sections measured in the field.

During Field Season I, the wetlands were planted in early summer with a rice crop; however this crop was not successful. Prior to Field Season 2, the wetlands were planted with bulrushes (*Scirpus* sp.), which were not well established during the early months of sampling (15% coverage of Pond 1 and 1% coverage of Pond 2); however became more established by the end of the second season's sampling period (70% of Pond 1 and 20% of Pond 2). Both ponds had open waters with filamentous algal growth in these areas as well as in the spaces between the emergent plants.

Table 6. Description of VTS systems and location.

Site Code	Location	Watershed	Description of Farm Operation	Description of VTS
VTS1	T14S, R2E, Section 25; Salinas	Salinas River	Conventional., Irrigated, Mixed Row Crop Vegetables	*Vegetated Treatment System* established in an already existing agricultural drainage ditch. This ditch is 1000 feet long and is broken up into three distinct sections. Each section is divided by a culvert. Runoff comes from a combination of on-farm activities and upstream activities.
VTS2	T12S, R2E, Section 15; Watsonville	Pajaro River	Greenhouse Nursery Operation; Cut flower production	Vegetative treatment system implemented in a segment of an already existing agricultural drainage ditch that bisects a large, cut flower, greenhouse nursery operation in the Pajaro River watershed. Runoff comes from adjacent fields.
VTS3	T15S, R4E, sec27; Salinas	Salinas River	Conventional. and Organic Mixed Vegetables	Established in dual., irrigation tailwater recovery ponds. Tailwater recovery ponds are located downstream of approximately 450 acres of sprinkler irrigated row crop mixed vegetables. Runoff comes from on-farm activities.
VTS4	T12S, R7E, section 29; San Juan Bautista	Pajaro River	Conventional. and Organic Mixed Vegetables	This system was designed to have two large shallow constructed wetlands linked to one another by a culvert. Thus water that flows into the system can be treated in tandem – each pond could have the same treatment (mix of plants for example) or vary depending on water quality goals. Runoff comes from on-farm activities.

5.2. Nitrate Concentration

Nitrate samples were taken at the inlet and the outlet of sites on a weekly basis during sampling periods, once access to the VTS was granted by the owner. Boxplots represent the median sample concentration as well as displaying the concentration ranges at the inlet and outlet (Fig. 3-4). Sample plots at the inlet and outlet show the measurements each day of sampling (Fig. 5-6). The mean, median and maximum inlet and outlet concentrations of nitrate are shown in Tables 7 - 8. The median concentration is relevant because it is robust to the skew that can be caused by outliers, however the mean is more representative of concentration reduction (if any) that may have occurred. In addition the maximum concentrations are shown. Wetlands can be important to water quality not only because they can treat water and remove contaminants, but also because of their ability to disperse water and smooth out peak concentrations. This smoothing effect can be important to the health and viability of aquatic organisms that are sensitive to exposure to concentrations sometimes observed at peaks.

Nitrate concentration reduction (%) was calculated for each VTS on an annual. and seasonal. basis in accord with the equation:

$$Reduction \; = 100 * \frac{\hat{C}o - \hat{C}i}{\hat{C}i}$$

where Co (mg/L) is the mean outlet concentration, Ci (mg/L) is the mean inlet concentration.
Local VTS nitrate removal (-3% to 21% in summer months) is lower than removal 76% removal typically found in the literature (Xiong 2011) for a variety of reasons : low residence times, deep water, young wetland age, insufficient carbon, high inlet concentrations, and inadequate size compared with inlet discharge (Tables 7&8).

Table 7. Comparison of inlet and outlet mean, median and maximum concentration during 2008 and Summer 2009 at the 4 VTSs. Treatment is best estimated by the difference in the mean. VTS systems also disperse pollutants and lower peak concentrations.

Nitrate Concentration (mg/L)							
		2008			Summer 2009		
		Inlet	Outlet	% Reduction	Inlet	Outlet	% Reduction
VTS1	mean	90.39	87.36	3	75.66	74.47	2
	median	85.11	83.13		74.63	75.75	
	maximum	209.00	140.11		93.00	83.75	
VTS2	mean	25.50	22.58	11			
	median	27.20	25.56				
	maximum	49.53	90.25				
VTS3	mean	35.55	31.83	10	36.21	34.49	5
	median	31.36	30.52		33.44	34.81	
	maximum	88.50	65.92		46.93	37.53	
VTS4	mean	75.25	78.37	-4	66.68	62.45	6
	median	67.12	67.67		67.00	64.63	
	maximum	124.35	166.17		77.25	82.50	

Table 8. Comparison of seasonal performance shows higher denitrification in the warmer months. Negative denitrification in the winter could be due to plant die off along with lower denitrification rates. VTS4 had a failed rice crop in 2008, which may have been the cause of its nitrate contribution to water in the summer months.

Nitrate Concentration (mg/L)							
		Summer 2008			Winter/Spring 2008		
		Inlet	Outlet	% Reduction	Inlet	Outlet	% Reduction
VTS1	mean	95.94	85.34	11	83.64	90.02	-8
	median	81.96	82.90		85.24	84.65	
	maximum	209.00	140.11		112.51	127.00	
VTS2	mean	20.47	18.21	11	31.85	29.58	7
	median	20.53	2.71		33.88	30.15	
	maximum	49.53	90.25		46.26	68.75	
VTS3	mean	40.09	31.74	21	29.55	32.03	-8
	median	33.79	30.63		25.95	30.30	
	maximum	88.50	65.92		46.12	44.84	
VTS4	mean	77.57	80.16	-3	70.28	75.02	-7
	median	70.66	67.67		64.23	65.65	
	maximum	124.35	166.17		110.50	132.50	

Figure 5. Boxplot comparisons of inlet and outlet nitrate concentrations in the summers of 2008 and 2009. Local VTS nitrate removal (-3% to 21% in summer months) is lower than reported removal of other studies for a variety of reasons including low residence times, deep water, young wetland age, insufficient carbon and inadequate size compared with inlet concentrations and discharge. These wetlands were incorporated into existing farm drainage systems including drainage ditches and sediment ponds, with the exception of the newly constructed VTS4. Successful denitrification in existing drainages with minor modifications represents the least cost approach to treatment, however proved inadequate to achieving substantial nitrate reduction.

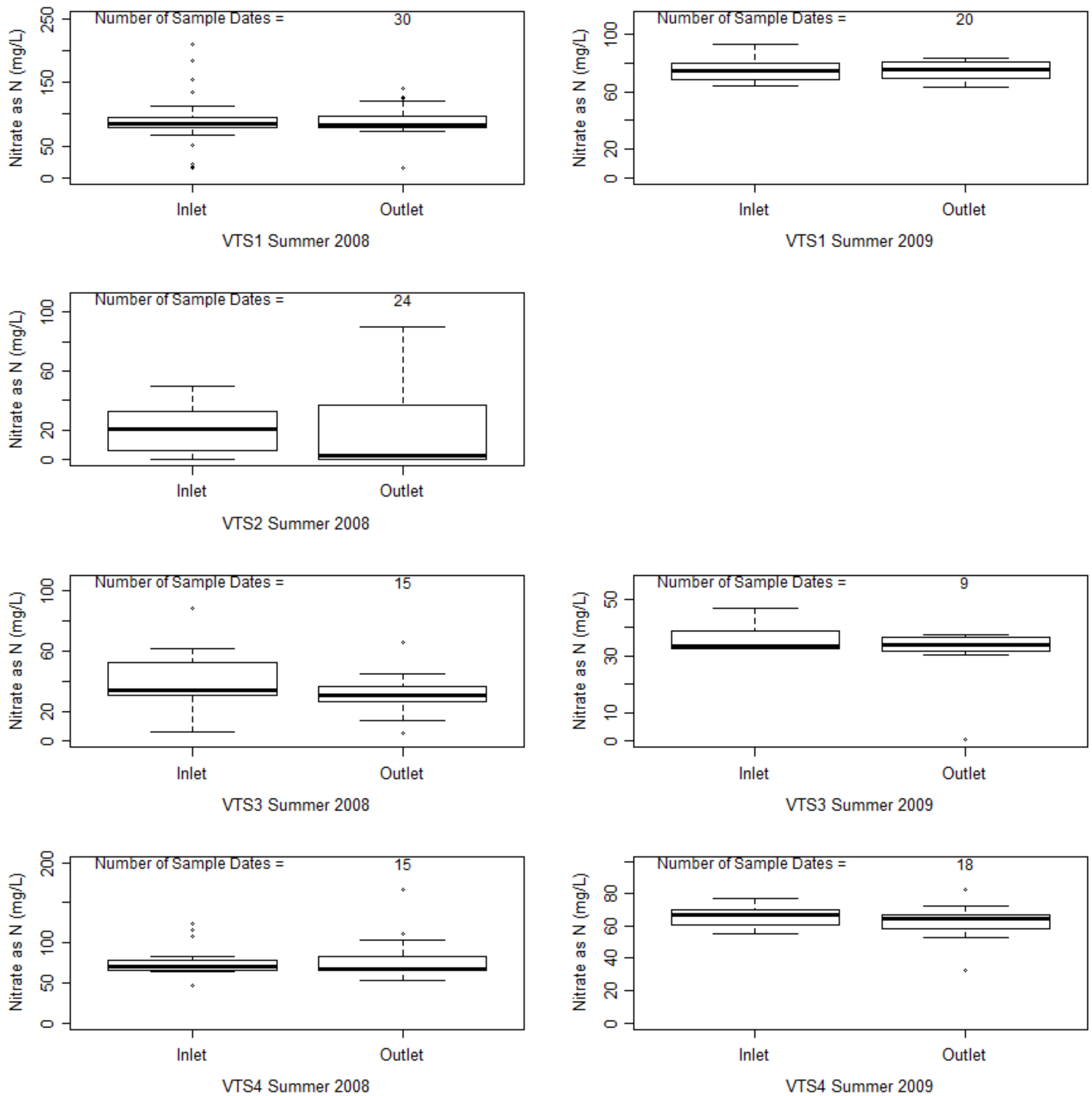


Figure 6. As indicated by the lower median value at the outlet as compared with the inlet, VTS3 shows evidence of higher nitrate removal in the summer period both years, but not in the Winter/Spring months of 2008.

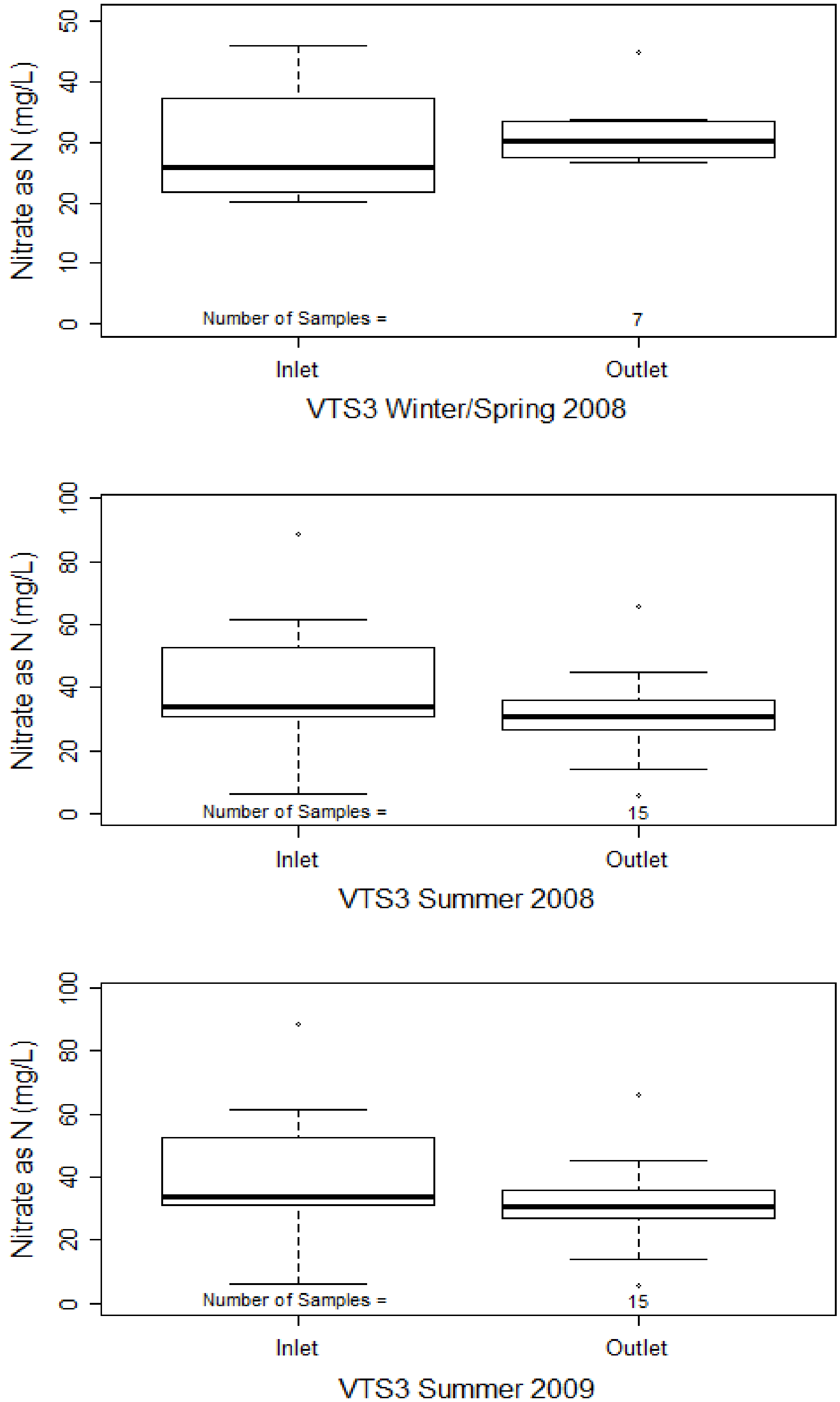


Figure 7A. Plots of inlet and outlet nitrate concentrations (mg/L) for VTS1 Section 1 over 3 Years. Samples were collected on the same day approximately 15 minutes apart, so they do not represent treatment effect as sample collection did not allow for residence time. However comparison of inlet and outlet samples over the long time frame provides insight into VTS performance. The dotted line represents the drinking water standard.

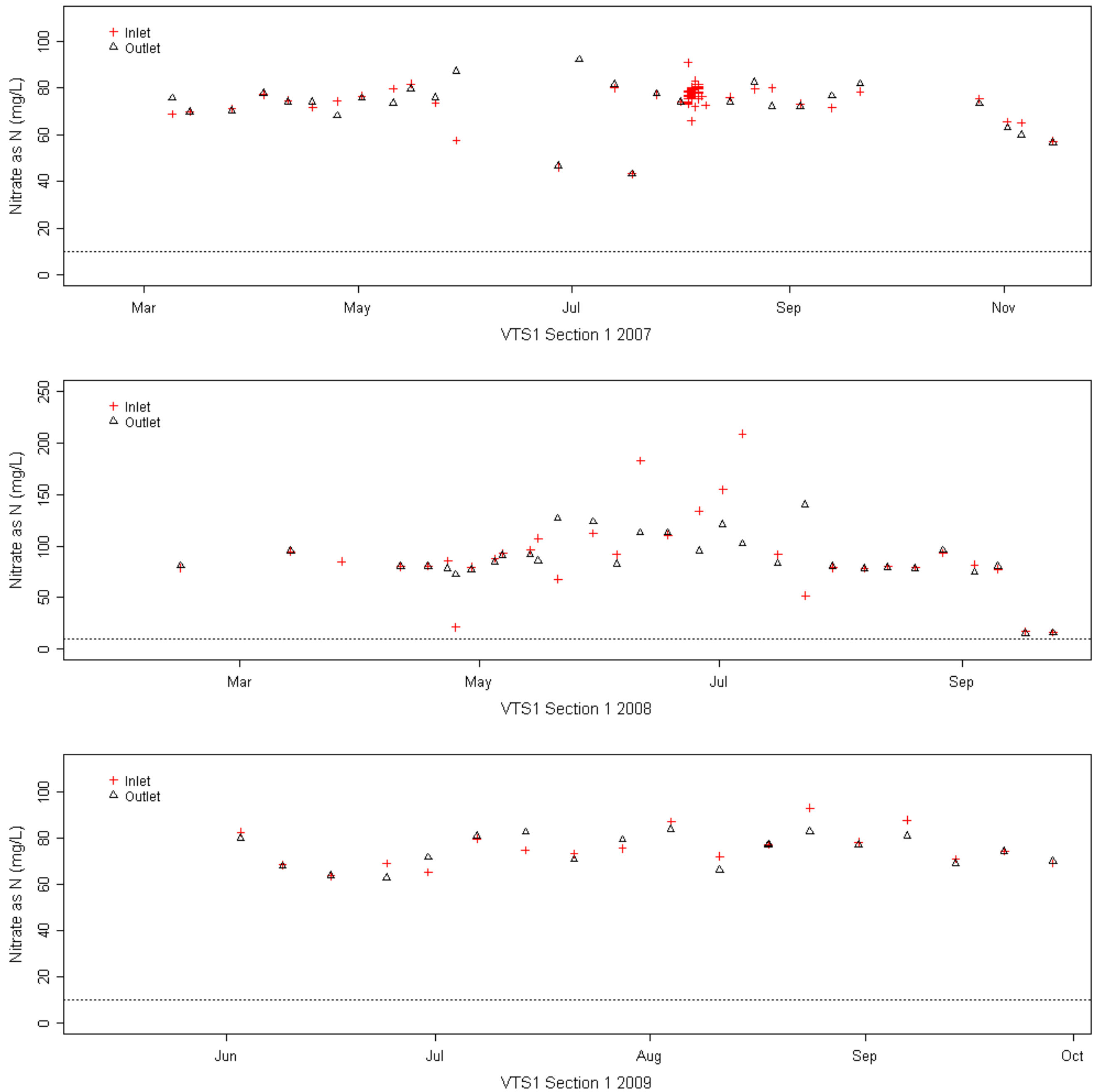


Figure 7B. Plots of inlet and outlet nitrate concentrations (mg/L) for VTS1 Section 2 over 3 Years. Samples were collected on the same day approximately 15 minutes apart, so they do not represent treatment effect as sample collection did not allow for residence time. However comparison of inlet and outlet samples over the long time frame provides insight into VTS performance. The dotted line represents the drinking water standard.

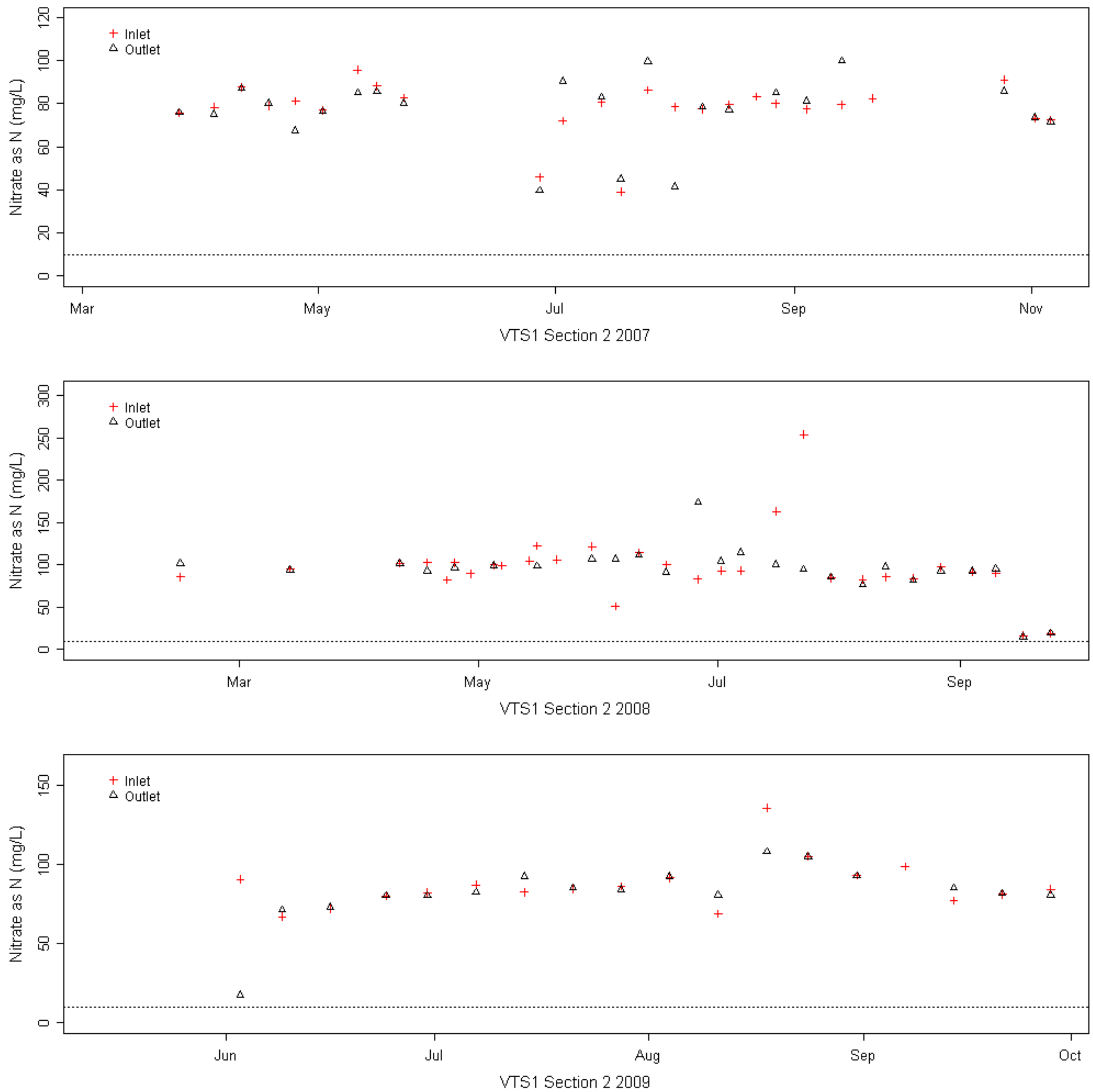


Figure 7C. Plots of inlet and outlet nitrate concentrations for VTS2 and VTS3 sampling results. Note the difference in y-axis scales. Samples are collected on the same day approximately 15 minutes apart, so they do not represent treatment effect as sample collection did not allow for residence time. However comparison of median inlet and outlet samples over the long time frame provides insight into VTS performance. The dotted line represents the drinking water standard.

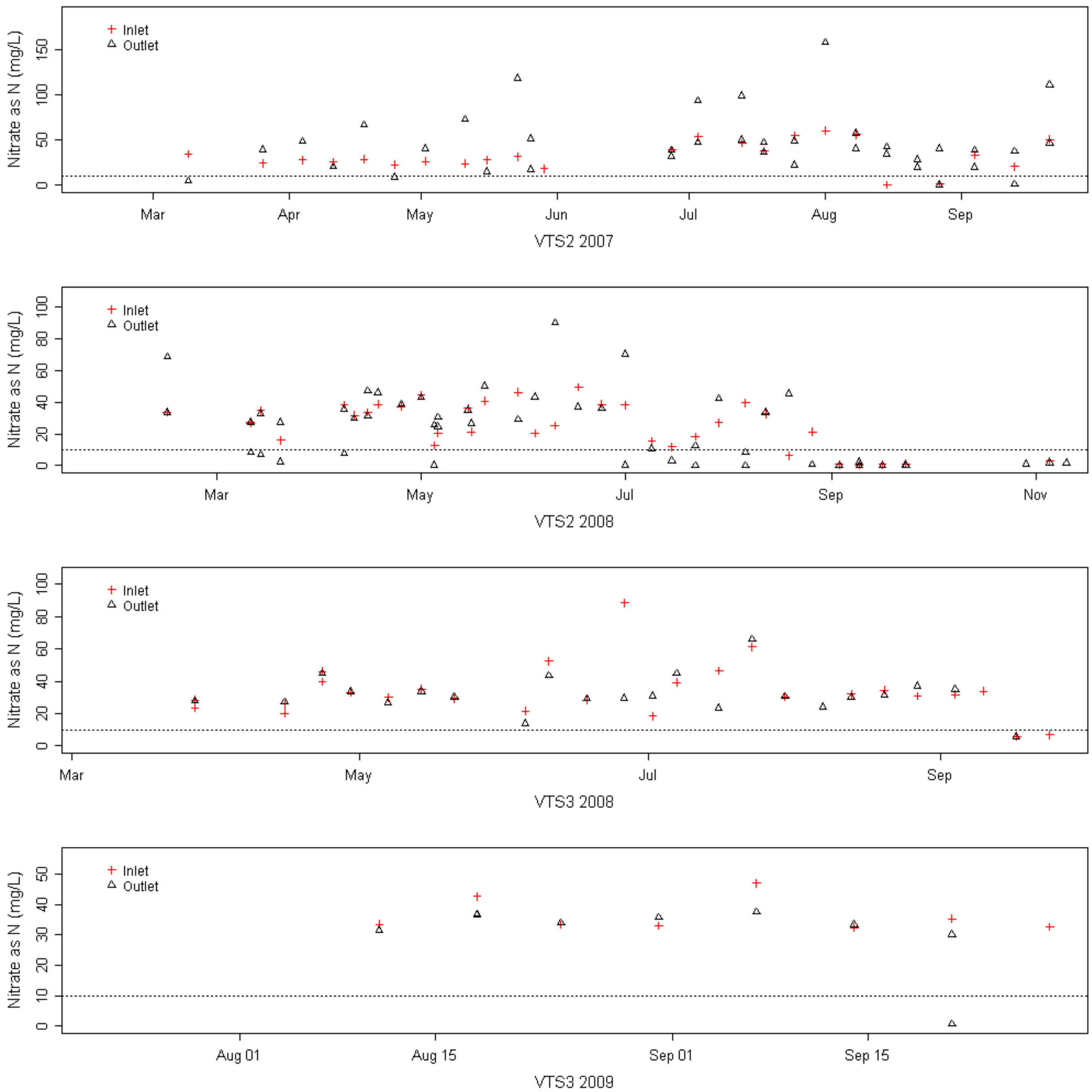
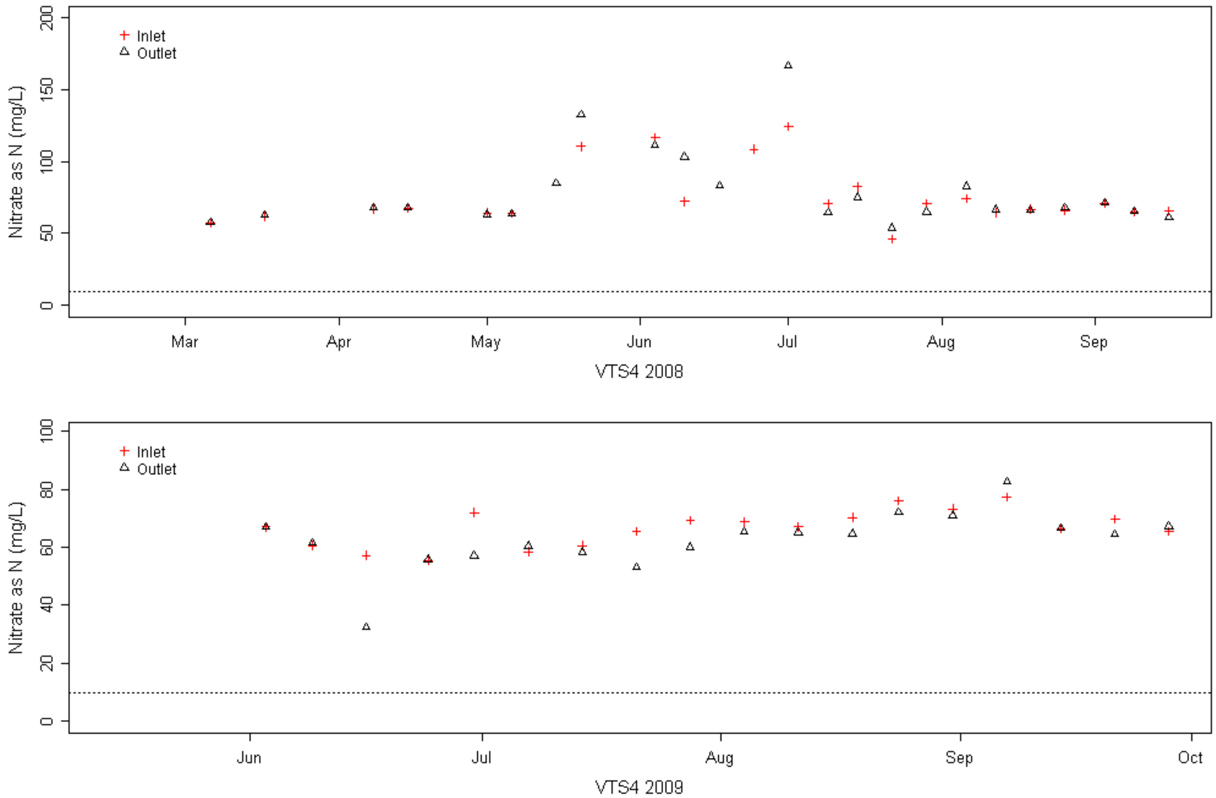


Figure 7D. Plots of inlet and outlet nitrate concentrations for VTS4 sampling results. Note the difference in y-axis scales. Samples are collected on the same day approximately 15 minutes apart, so they do not represent treatment effect as sample collection did not allow for residence time. However comparison of median inlet and outlet samples over the long time frame provides insight into VTS performance.



5.3. Ammonia Concentration

Inlet and outlet concentrations of ammonia were generally very low (Table 9). Neither inlet nor outlet concentrations were higher than the aquatic toxicity objective established by the CCRWQCB for Salinas (0.025 mg/L), except occasionally (Figures 6-8). VTS2 was the only system that occasionally demonstrated outlet concentrations that exceeded this objective , i.e. 5 out of 44 sample dates (11%). The mean and median concentrations are considered estimates as the measured concentrations were frequently below the method detection limit (MDL < 0.005 mg/L).

Table 9. Ammonia mean, median and maximum at the inlet and outlet of each VTS for 2008 and 2009. In most samples, the result was less than the MDL (0.005 mg/L) , such that the mean was less than the MDL. The mean is therefore a rough estimate. Ammonia concentrations were very low in most cases, so the negative reductions observed in VTS3 and VTS4 do not signify concentration levels toxic to aquatic biota (> 0.025 mg/L). As there are living organisms in VTS systems, small amounts of ammonia addition is not unexpected. Percent reduction was not derived (nd) due to the low mean concentrations.

Unionized Ammonia as N Concentration (mg/L)							
		2008			2009		
		Inlet	Outlet	% Reduction	Inlet	Outlet	% Reduction
VTS1	mean	0.002	0.003	nd	0.003	0.001	nd
	median	0.001	0.001		0.002	0.001	
	maximum	0.009	0.011		0.009	0.003	
VTS2	mean	0.003	0.006				
	median	0.002	0.001				
	maximum	0.023	0.060				
VTS3	mean	0.002	0.003	nd	0.002	0.001	nd
	median	0.001	0.001		0.001	0.001	
	maximum	0.010	0.015		0.004	0.002	
VTS4	mean	0.001	0.005	nd	0.000	0.002	nd
	median	0.000	0.004		0.000	0.002	
	maximum	0.006	0.019		0.001	0.004	

Figure 8. Boxplot comparisons of unionized ammonia inlet and outlet concentrations during summers of 2008 and 2009 show mixed results. Ammonia concentrations were generally low in 2008, i.e. <0.025 mg/L - the aquatic toxicity objective established by the RWQCB. In 2009, VTS1 and VTS3 positively influenced peak ammonium, reducing the highest concentrations of ammonium. By contrast, VTS4 received very low input concentrations of ammonia compared with outlet, however the outlet concentrations never exceeded the aquatic toxicity objective.

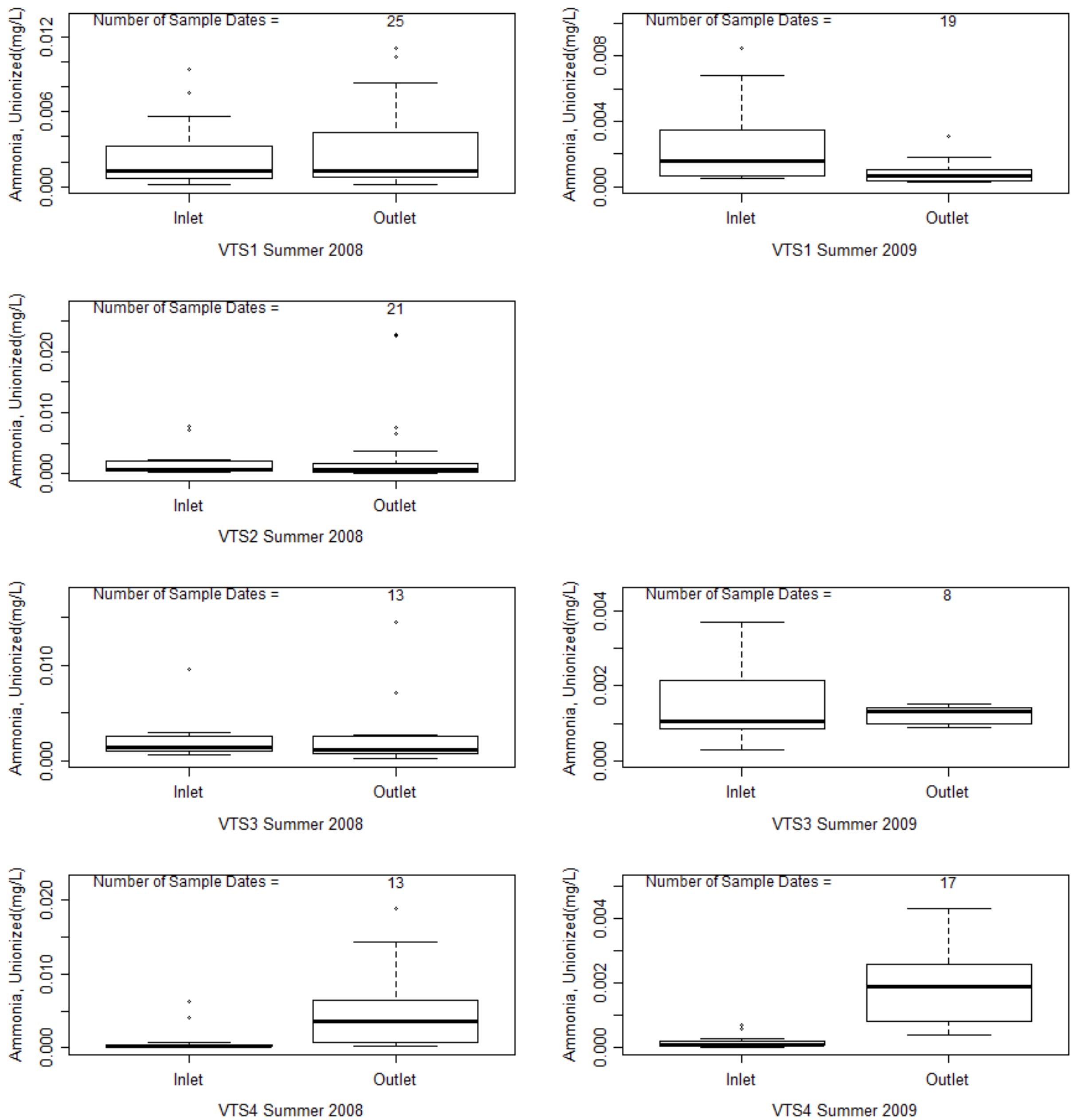


Figure 9A. Plots showing VTS1 Section 1 inlet and outlet concentrations over a 3 year period. The dotted line represents the aquatic toxicity objective established by the CCRWQCB. Note the scale difference of the y axis, such that low concentrations in some systems had maximum concentrations < 0.025 mg/L so there is no dotted line on these plots. There were only two samples that were above the aquatic toxicity objective of 0.025 mg/L, one at the inlet and one at the outlet.

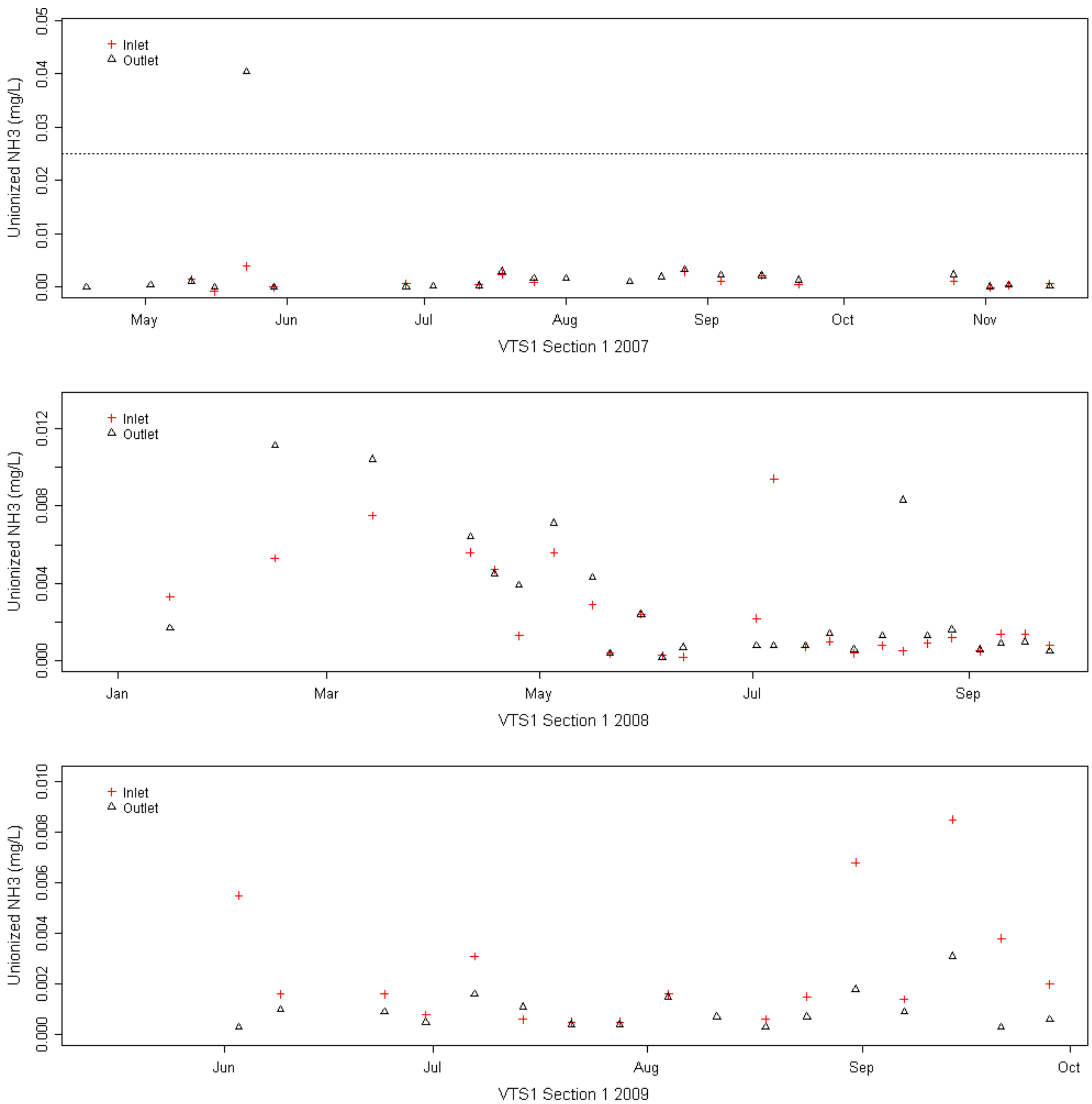


Figure 9B. Plots showing VTS1 Section 2 inlet and outlet unionized ammonia concentrations over a 3 year period. The dotted line represents the aquatic toxicity objective established by the CCRWQCB. Note the scale difference of the y axis, such that low concentrations in some systems had maximum concentrations < 0.025 mg/L so there is no dotted line on these plots. There was one sample that were above the aquatic toxicity objective of 0.025 mg/L, at the inlet.

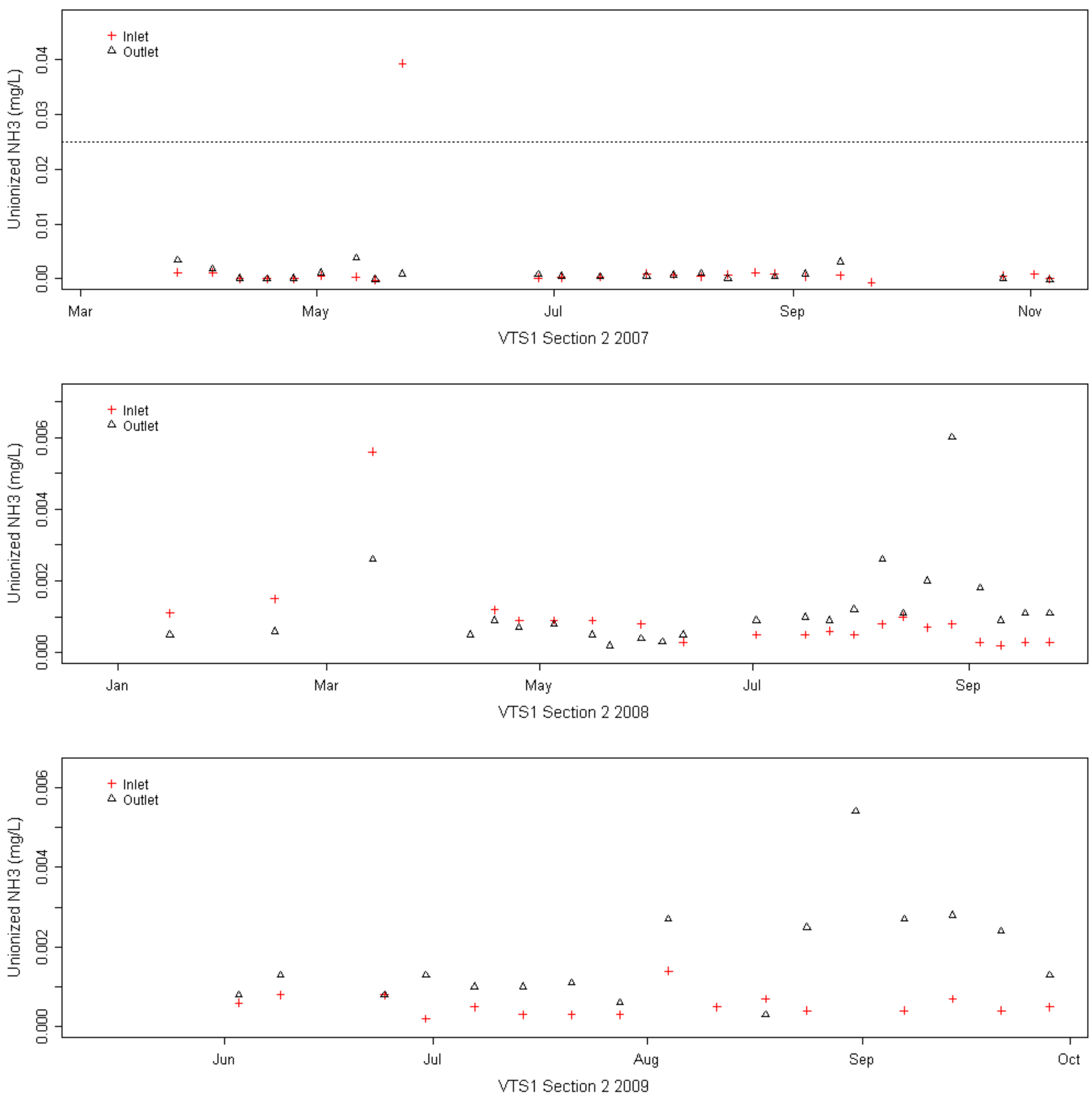


Figure 9C. Plots showing VTS2 and VTS3 inlet and outlet unionized ammonia concentrations over a 2 year period. Note the difference in scales of the y axis. The aquatic toxicity objective of 0.025 mg/L was not exceeded at the inlet or outlet of VTS3. This aquatic toxicity objective was exceeded at the outlet on five sample dates (11%) in VTS2.

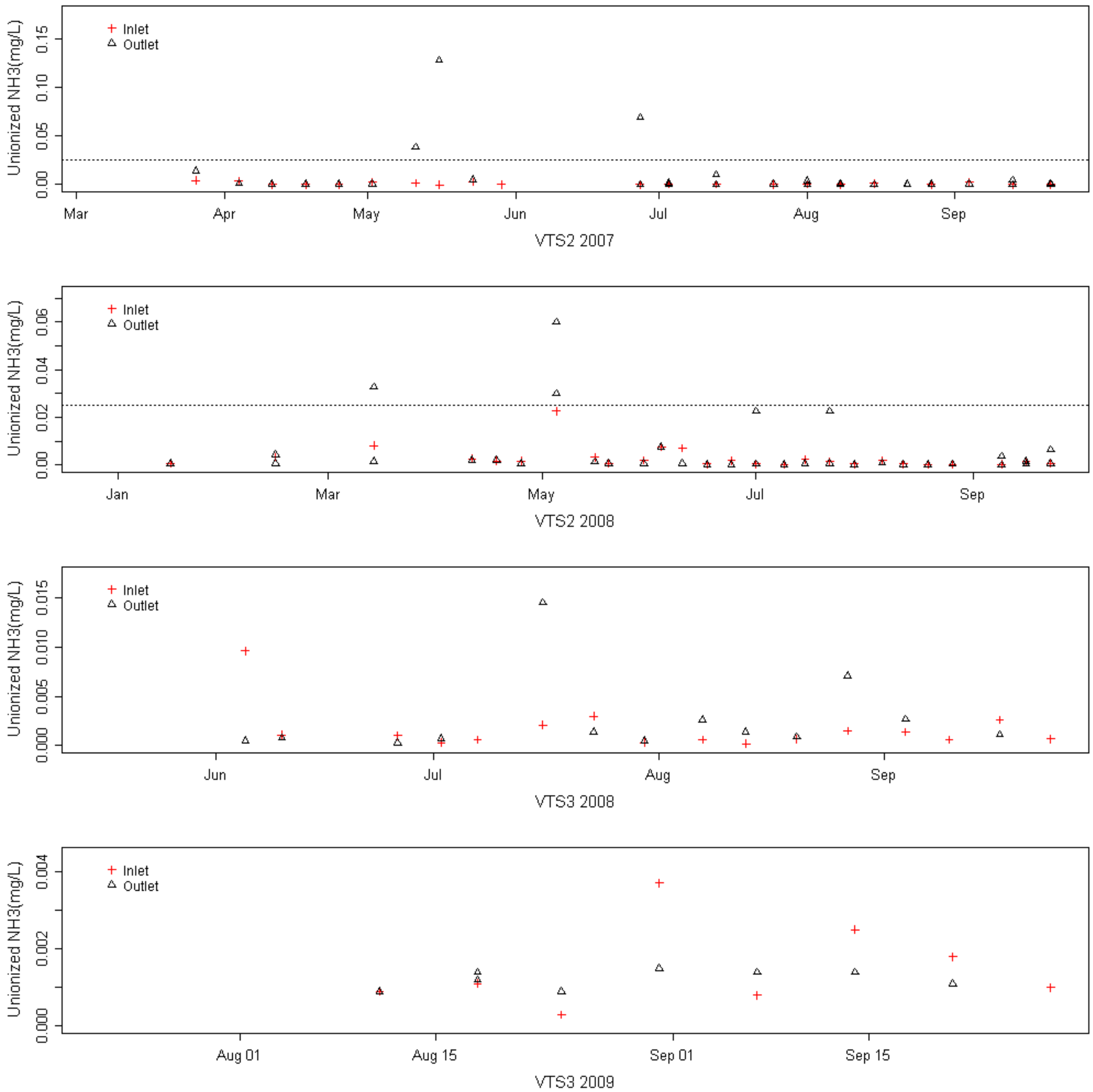
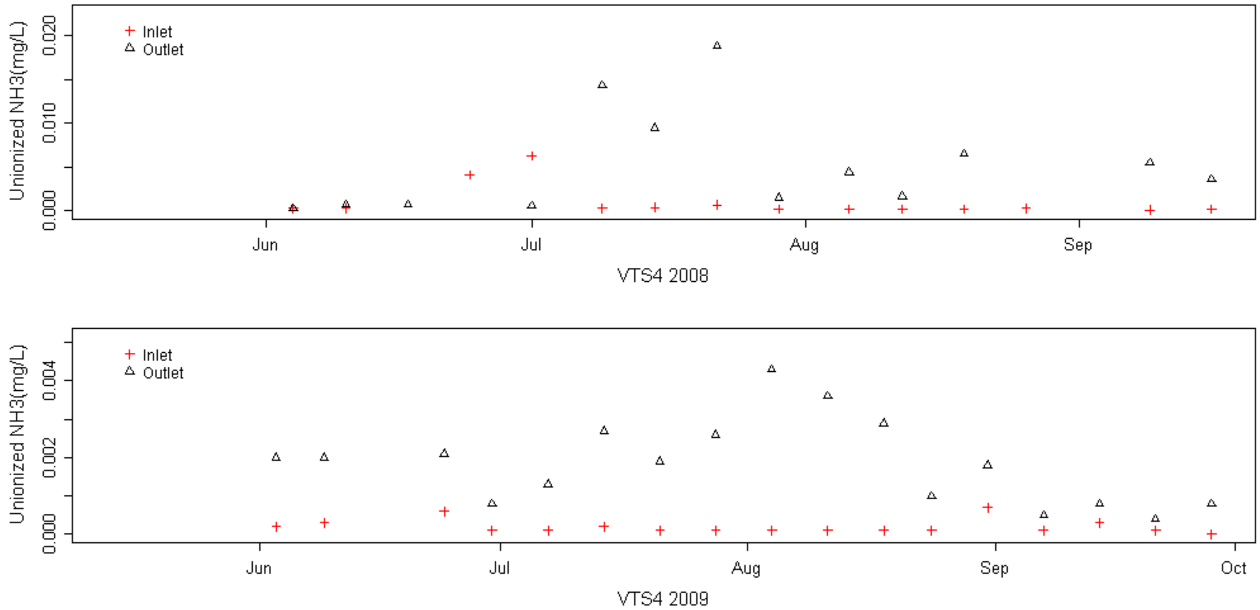


Figure 9D. Plots showing VTS4 inlet and outlet unionized ammonia concentrations over a 2 year period. Note the difference in scales of the y axis. The aquatic toxicity objective of 0.025 mg/L was not exceeded.



5.4. Total Dissolved Solids Concentration

The results of total dissolved solid concentration field measures are displayed in Table 10 and Figures 10-12.

Table 10. Comparison of inlet and outlet concentrations of total dissolved solids in 2008 and 2009.

Total Dissolved Solids Concentration (g/L)							
		2008			2009		
		Inlet	Outlet	% Reduction	Inlet	Outlet	% Reduction
VTS1	mean	1.6650	1.6737	-1	1.6410	1.7044	-4
	median	1.7052	1.6757		1.7261	1.7732	
	maximum	1.8740	1.8848		1.8442	1.8531	
VTS2	mean	1.0835	1.0398	4			
	median	1.0730	1.0950				
	maximum	2.4780	1.9920				
VTS3	mean	0.5865	0.6869	-17	0.6211	0.5933	4
	median	0.6800	0.6926		0.6954	0.7180	
	maximum	0.7690	0.7690		0.8702	0.7403	
VTS4	mean	2.7786	2.7150	2	2.5307	2.0776	18
	median	2.6579	2.5989		2.5216	2.5675	
	maximum	5.3740	5.3150		2.8799	2.7962	

Figure 10. Boxplot comparisons of TDS concentrations at the 4 VTS inlets and outlets in 2008 and 2009. The median concentrations were similar at the inlet as compared with the outlet.

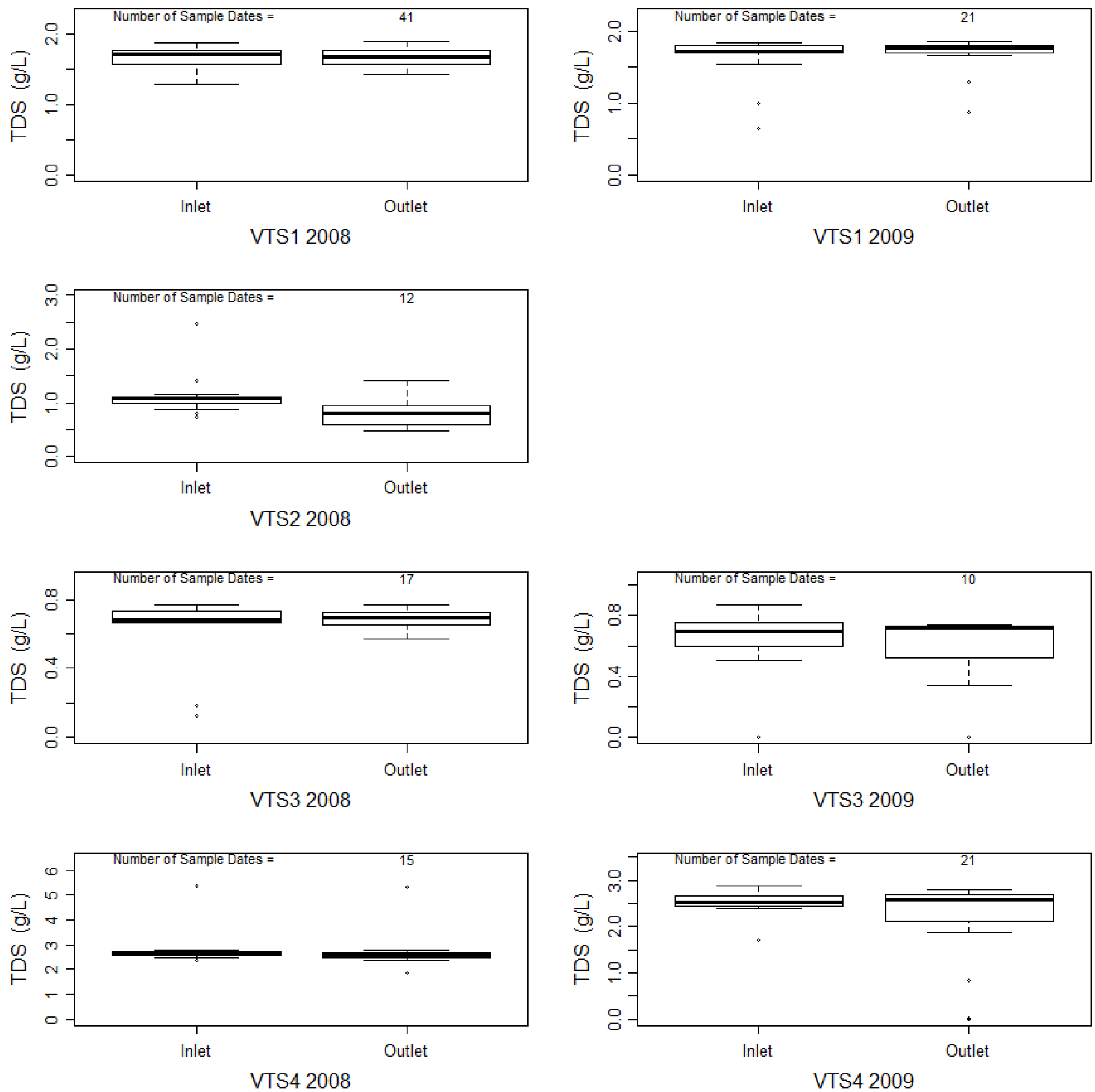


Figure 11A. Plots of total dissolved solid observations at the inlet and outlet of VTS1 Section 1, a vegetated drainage ditch, over three years show similar concentrations.

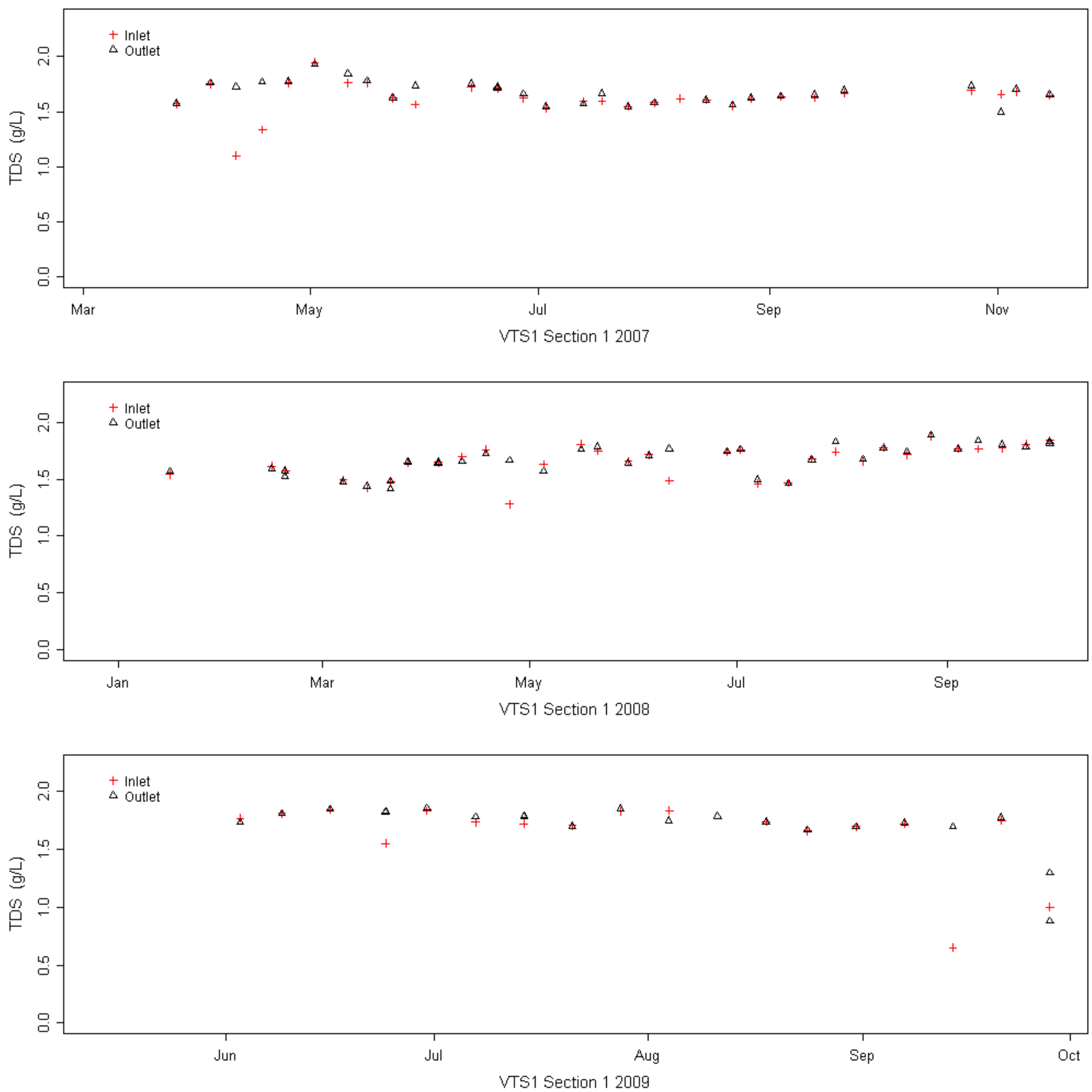


Figure 11B. Plots of total dissolved solid observations at the inlet and outlet of VTS1 Section 2, a vegetated drainage ditch, over three years show similar concentrations.

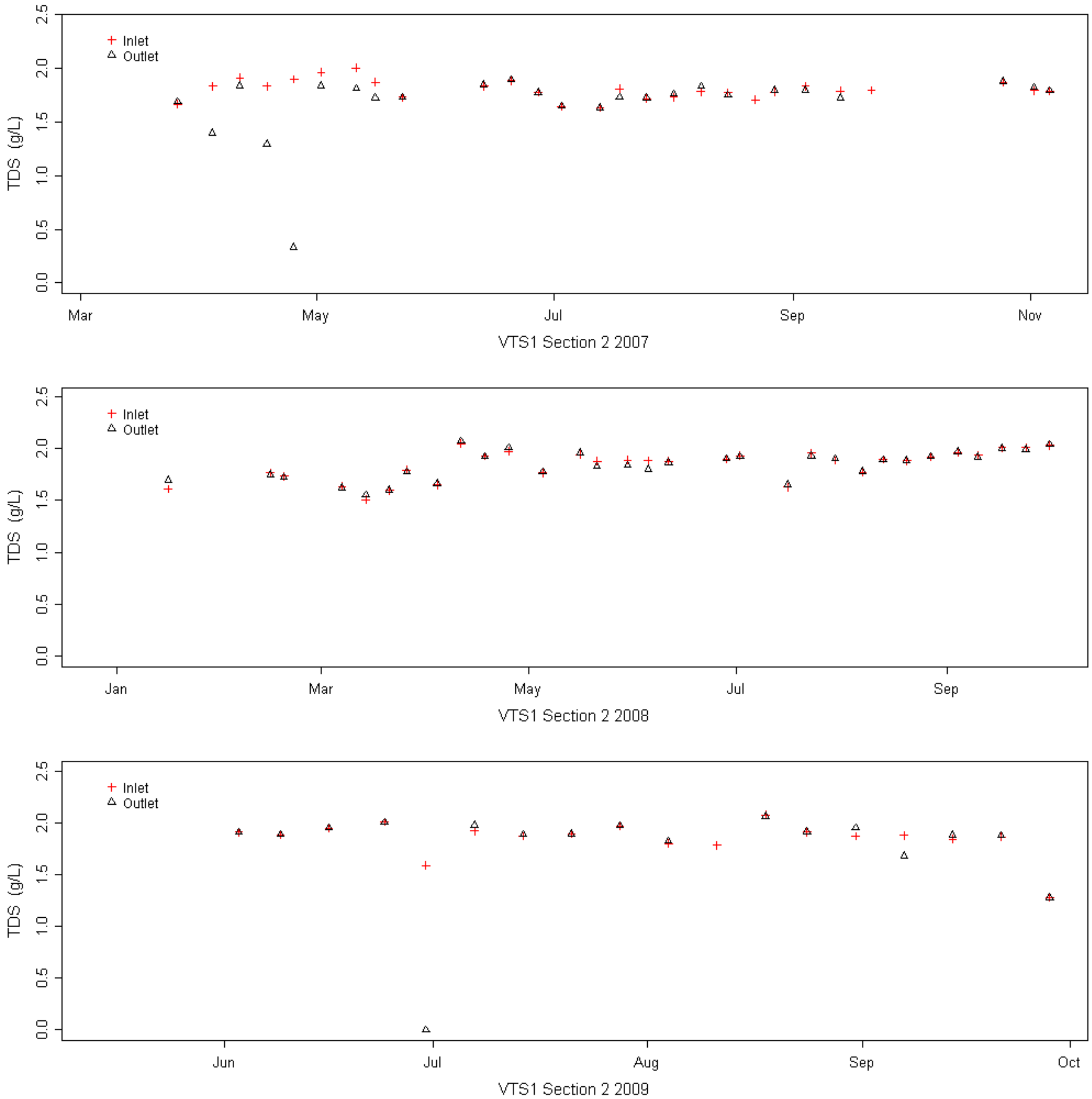


Figure 11C. Plots of total dissolved solid observations at the inlet and outlet of VTS2 (a vegetated drainage ditch) and VTS3 (a vegetated sediment pond) over three years.

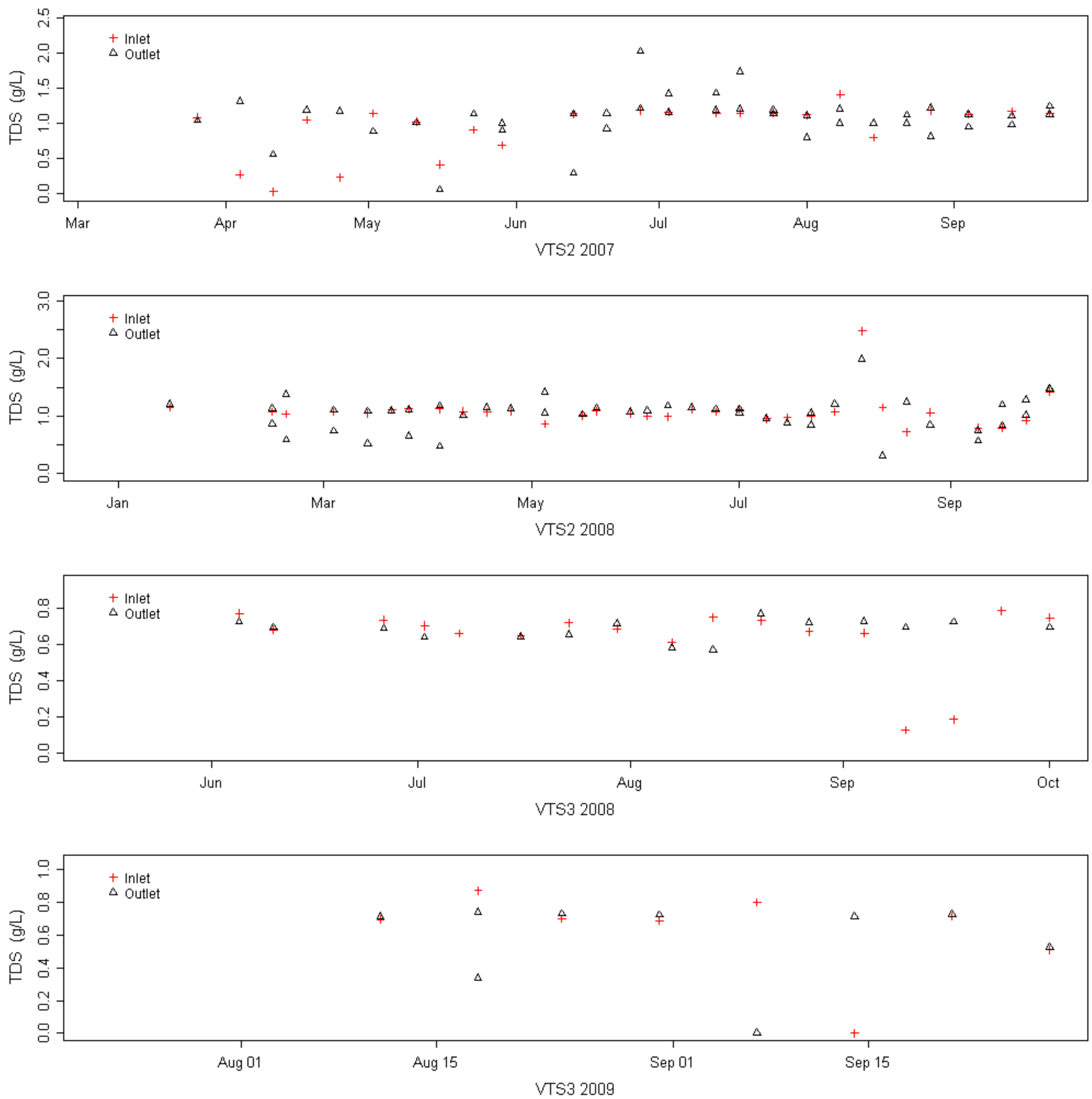
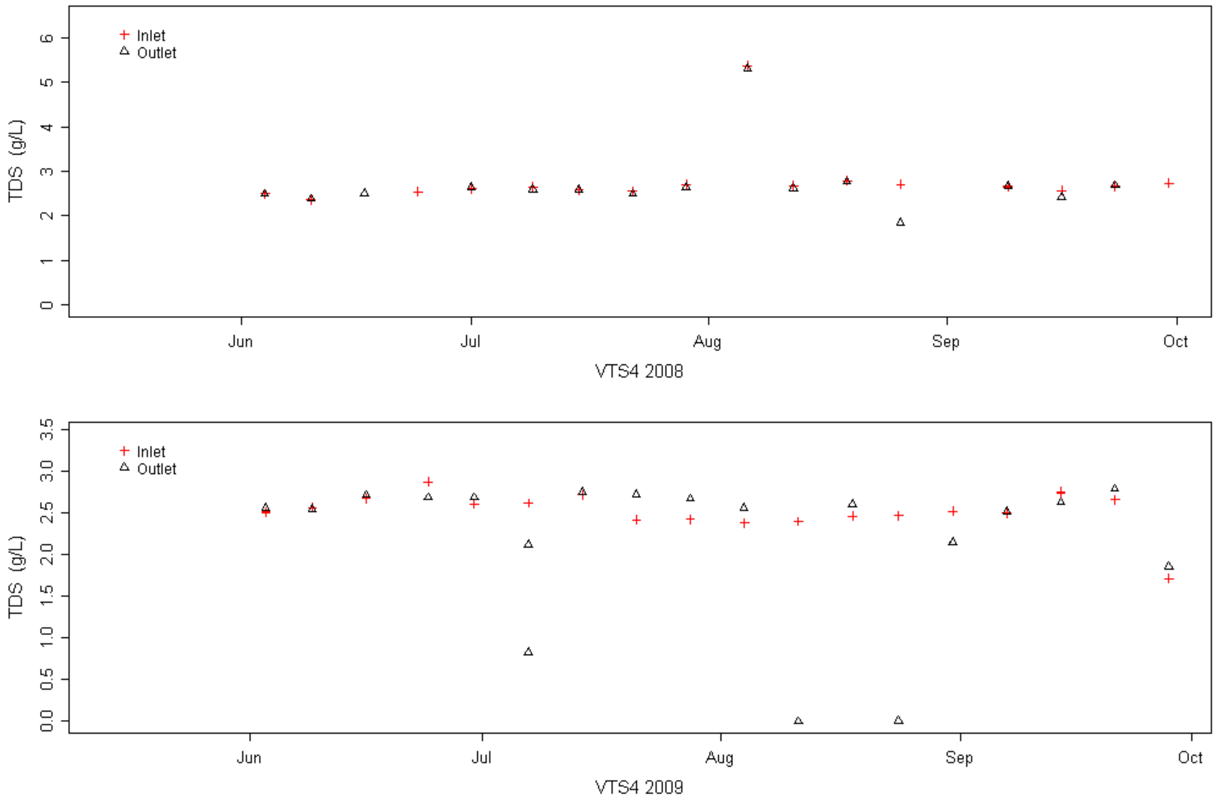


Figure 11D. Plots of total dissolved solid observations at the inlet and outlet of VTS4 (a new wetland) over three years.



5.5. Suspended Sediment Concentration

Suspended sediment at the wetland inlets was generally low with the exception of VTS3 (Fig. 13-14). The primary purpose of the VTS3 vegetated sediment ponds is the removal of sediment, and the two pond system showed high effectiveness at accomplishing this removal: 99% in 2008 and 96% in 2009 (Table 11). VTS2 was second highest, in terms of suspended sediment concentration at the inlet and also demonstrated a substantial reduction of 43% suspended sediment removal Although VTS1 in 2009 and VTS4 in 2008 demonstrated negative sediment removal, they had low inlet mean concentrations (0.003 g/L and 0.015 g/L respectively).

Table 11. Suspended sediment removal was most effective in the VTS systems with high inlet concentrations.

Suspended Sediment Concentration (g/L)							
		2008			2009		
		Inlet	Outlet	% Reduction	Inlet	Outlet	% Reduction
VTS1	mean	0.060	0.004		0.015	0.039	
	median	0.020	0.002	94	0.009	0.011	-155
	maximum	0.293	0.021		0.073	0.192	
VTS2	mean	0.205	0.117				
	median	0.159	0.072	43			
	maximum	0.643	0.349				
VTS3	mean	4.127	0.035		1.864	0.078	
	median	4.275	0.031	99	0.965	0.060	96
	maximum	6.242	0.102		5.333	0.165	
VTS4	mean	0.003	0.012		0.004	0.004	
	median	0.002	0.008	-357	0.002	0.002	0
	maximum	0.007	0.033		0.015	0.010	

Figure 12. Boxplots of suspended sediment concentrations at the 4 VTS inlet and outlets display the greatest removal when sediment concentration at the inlet is high. The median sediment concentrations at VTS3 and VTS2 were substantially lower than outlet concentrations. Although the reverse looks true for VTS4 in 2008, the medians of both the inlet and outlet are quite low at concentrations 0.007 g/L and 0.033 g/L respectively.

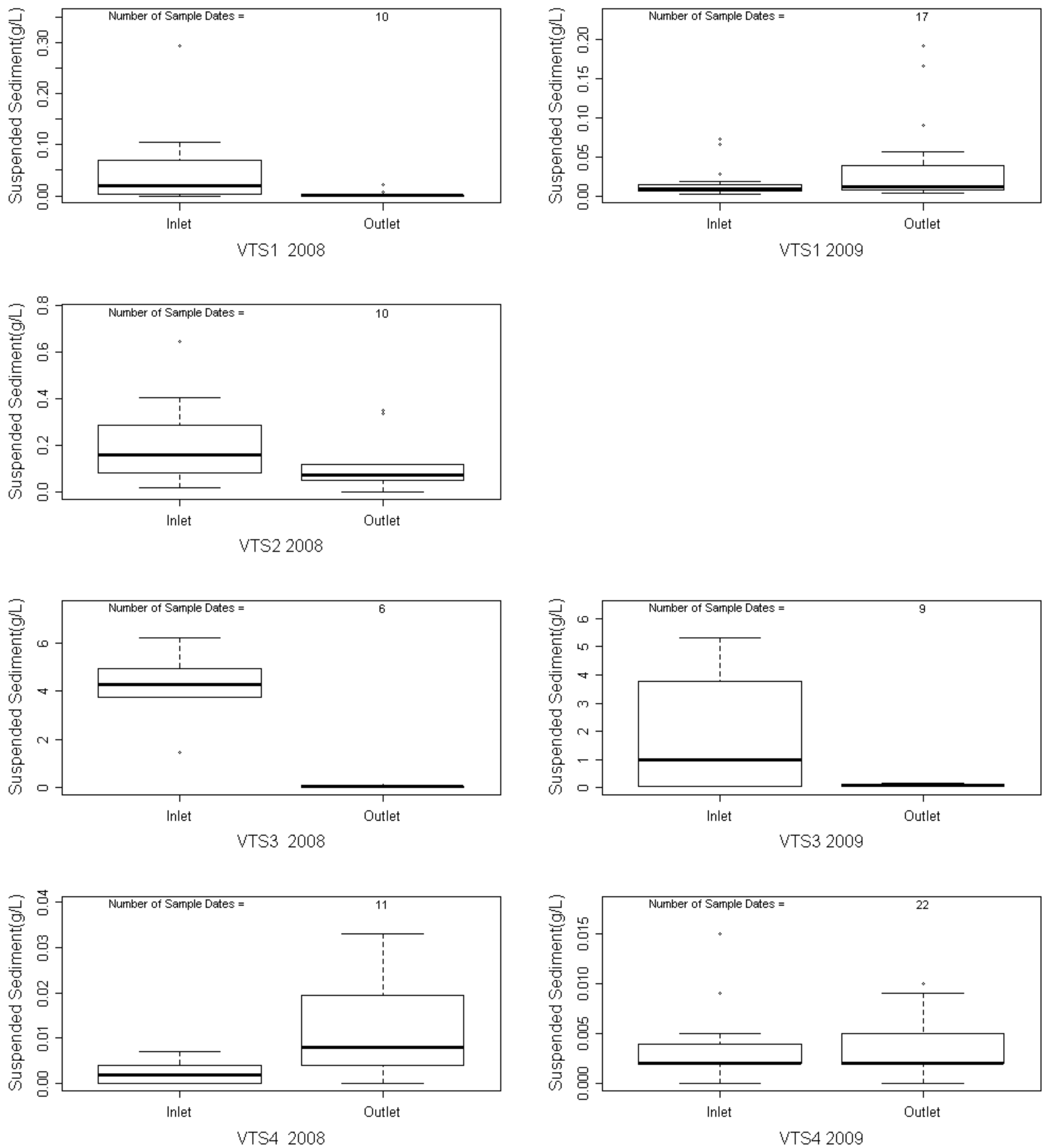
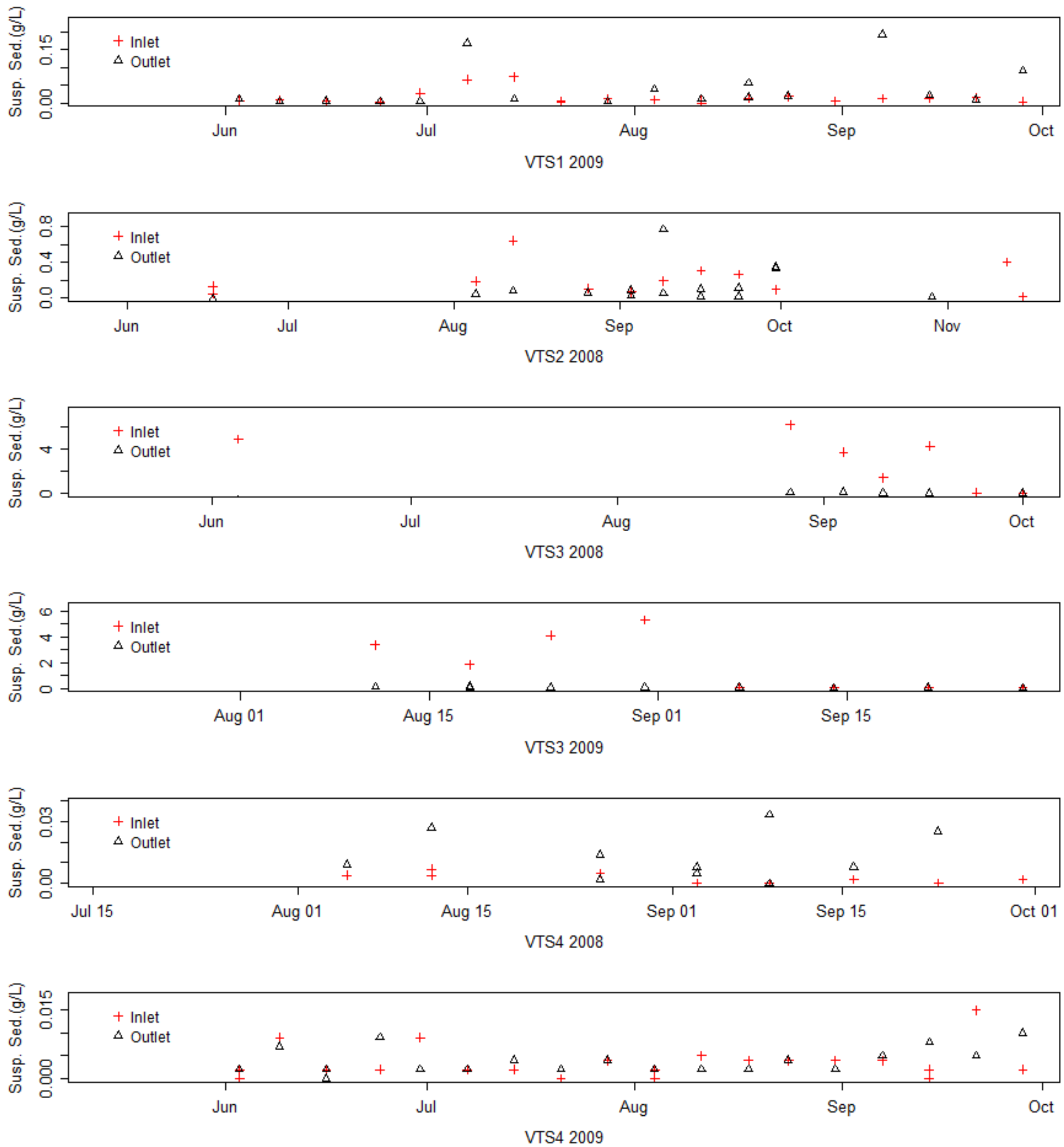


Figure 13. Plots of suspended sediment concentrations at the 4 VTS inlet and outlets display the greatest removal when sediment concentration at the inlet is high.



5.6. Inlet Loading Rates and Mass Balance Reduction

VTS Nutrient Mass Balance / Load Change

Loading rate is a measure of how much chemical is being added to the wetland per unit area and is used for comparisons of wetland performance. The input loading rate is the mass input per area per time calculated by multiplying discharge by concentration, divided by wetland area (Kadlec and Wallace 2009). For example, a 1000 m^2 wetland receiving an inlet concentration of 45 mg/L nitrate concentration at a discharge of 100 m^3/hr (≈18 gpm) would have an inlet loading rate of 4.5 g/ m^2 day. Percent mass removal is calculated by the equation

$$\%\, Mass\ Removal = 100 * \frac{(QiCi - QoCo)}{Qi}$$

Mass balance removal is best computed over a long time span because same day samples, given the long hydraulic retention time of wetlands, are clearly not representative of treatment (Kadlec and Wallace 2009). Kadlec and Wallace (2009) recommend seasonal balances include at least three months of data and annual reduction should ideally span a number of years. In addition to the mass balance it is important to know the water balance of the wetland, as water can be added by precipitation and runoff and removed by groundwater infiltration and evapo-transpiration. We did not consider precipitation addition or runoff in our calculations, since sampling was normally done on days when there was no rain. Evapo-transpiration was small, 1-2% and was not considered. However there was significant mass removal due to infiltration in VTS1 and VTS2; however not for VTS3 and VTS4.

VTS1 & VTS2 Load Calculations

Chemical loading rates and load reductions for VTS1 and VTS2 were computed based on measured concentrations and estimated discharges from field measures. Mean daily discharge for 2007 and 2008 were estimated from the report by Largay et al. (2009). Because discharge was not monitored in 2009 due to the weir removal, discharge was assumed to be the same as in 2008. A time series model of daily inlet and outlet concentration for the duration of each year's sampling period was developed. Measured concentrations from weekly monitoring were linearly interpolated to estimate daily concentration between sample events.

Table 12. VTS1 and VTS2 areal load reduction, inlet and outlet mass and mass load reduction.

VTS1 Load

Year	2007	2008	2009
Sample Period (days)	235	253	118

Analyte	Inlet Mass Loading (g/m^2 d)			Mean Inlet Mass (kg/d)			Mean Outlet Mass (kg/d)			Mass Load Reduction %		
	2007	2008	2009	2007	2008	2009	2007	2008	2009	2007	2008	2009
Nitrate as N	60.17	189.45	156.46	29.06	91.5	75.57	26.44	65.07	54.46	9.0	28.9	27.9
Ammonium as N	0.0007	0.0067	0.0048	0.0003	0.0033	0.0023	0.0008	0.0032	0.0007	-132.6	2.7	69.2
Total Dissolved Solids	1073.26	3394.36	3429.53	518.38	1639.48	1656.46	500.16	1217.37	1275.16	3.5	25.7	23.0
Suspended Sediment	ns	146.15	36.33	ns	70.59	17.55	ns	6.20	30.25	ns	91.2	-72.4
Orthophosphate as P	0.272	0.763	0.749	0.131	0.368	0.362	0.112	0.260	0.226	15.0	29.6	37.4

VTS2 Load

Year	2007	2008	2009
Sample Period (days)	197	265	ns

Analyte	Inlet Mass Loading (g/m^2 d)		Mean Inlet Mass (kg/d)		Mean Outlet Mass (kg/d)		Mass Load Reduction %	
	2007	2008	2007	2008	2007	2008	2007	2008
Nitrate as N	12.5	25.25	4.42	8.92	6.63	3.34	-50.0	62.5
Ammonium as N	0.0004	0.0039	0.0001	0.0014	0.0018	0.0018	-1000.0	-29.2
Total Dissolved Solids	386.87	1200.08	136.72	424.11	156.07	337.60	-14.2	20.4
Suspended Sediment	ns	240.75	ns	85.08	ns	12.91	ns	84.8
Orthophosphate as P	0.122	0.362	0.043	0.128	1.653	0.411	-3000.0	-200.0

VTS3 Load Calculations

Chemical loading rates and load reductions for VTS3 were computed based on measured concentrations and rough estimates of discharge over a range of feasible values. Due to difficulties associated with sporadic inputs from multiple sources at the inlet and outlet piping into a ditch, field measures of discharge could not be obtained. A time series model of daily inlet and outlet concentration for the duration of each year's sampling period was developed using three different flow measures. High, medium and low flow estimates 40 m^3/day, 7.5 m^3/day, and 0.75 m^3/day were computed to cover the range of feasible discharge rates. Measured concentrations from weekly monitoring were linearly interpolated to estimate daily concentration between sample events.

Table 13. VTS3 areal load reduction, inlet and outlet mass, and mass load reduction.

VTS3 Low Flow Load Estimates								
Year	2007	2008	2009					
Sample Period (days)	ns	182	49					
Analyte	**Inlet Mass Loading (g/m^2 d)**		**Mean Inlet Mass (kg/d)**		**Mean Outlet Mass (kg/d)**		**Mass Load Reduction %**	
	2008	**2009**	**2008**	**2009**	**2008**	**2009**	**2008**	**2009**
Nitrate as N	0.050	0.055	0.025	0.027	0.023	0.022	10.7	18.9
Ammonium as N	0.000	0.000	0.000	0.000	0.000	0.000	-101.107	33.08
Total Dissolved Solids	0.956	0.930	0.478	0.465	0.510	0.434	-6.730	6.72
Suspended Sediment	7.018	2.786	3.512	1.394	0.021	0.055	99.393	96.03
Orthophosphate as P	0.001	0.001	0.000	0.000	0.000	0.000	-2.473	-4.28

VTS3 Medium Flow Load Estimates								
Analyte	**Inlet Mass Loading (g/m^2 d)**		**Mean Inlet Mass (kg/d)**		**Mean Outlet Mass (kg/d)**		**Mass Load Reduction %**	
	2008	**2009**	**2008**	**2009**	**2008**	**2009**	**2008**	**2009**
Nitrate as N	0.500	0.548	0.250	0.274	0.230	0.222	10.7	18.9
Ammonium as N	0.000	0.000	0.000	0.000	0.000	0.000	-101.1	33.1
Total Dissolved Solids	9.556	9.298	4.782	4.652	5.103	4.340	-6.7	6.7
Suspended Sediment	70.179	27.861	35.118	13.942	0.213	0.553	99.4	96.0
Orthophosphate as P	0.008	0.008	0.004	0.004	0.004	0.004	-2.5	-4.3

VTS3 High Flow Load Estimates								
Analyte	**Inlet Mass Loading (g/m^2 d)**		**Mean Inlet Mass (kg/d)**		**Mean Outlet Mass (kg/d)**		**Mass Load Reduction %**	
	2008	**2009**	**2008**	**2009**	**2008**	**2009**	**2008**	**2009**
Nitrate as N	2.67	2.92	1.33	1.46	1.23	1.19	10.7	18.9
Ammonium as N	0.0001	0.0001	0.0001	0.0001	0.0001	0.0000	-101.1	33.1
Total Dissolved Solids	50.964	49.587	25.502	24.813	27.219	23.145	-6.7	6.7
Suspended Sediment	374.29	148.59	187.29	74.36	1.14	2.95	99.4	96.0
Orthophosphate as P	0.04	0.04	0.02	0.02	0.02	0.02	-2.47	-4.28

VTS4 Load Calculations

Load reduction was estimated from measured concentrations as well as from both measured and estimated discharges for VTS4. Discharge calculations for 2008 are included in the report by the Center for Agroecology and Sustainable Food Systems (2009) on "Water Quality Monitoring in Vegetative Treatment Systems in the Pajaro Watershed," in more detail under the heading "San Juan VTS." The median 2008 discharge was calculated as 75 m^3/day (CASFS 2009), and this rate was used in our load calculations for 2008. There was no discernible infiltration to groundwater in 2008. In 2009, outflow discharge was measured in the field on a weekly basis starting at the end of June. Discharge was estimated using a daily time series model with linear interpolation between measured weekly concentrations to estimate loading rate and mass removal. Based on field observations that flow was about the same before beginning flow sampling as during the first flow measures, prior flow was estimated to be the mean of the first two flow measurements in 2009. Inlet discharge was not measured because inlet flow was not constant, as pumps discharging water to the wetland were shut-off during periods of peak electricity costs. Inlet flows were assumed to be the same as outlet flow rates.

Table 14. VTS4 areal load reduction, inlet and outlet mass, and mass load reduction.

VTS4 Load								
Year	2007	2008	2009					
Sample Period (days)	ns	195	118					
Analyte	**Inlet Mass Loading (g/m^2 d)**		**Mean Inlet Mass (kg/d)**		**Mean Outlet Mass (kg/d)**		**Mass Load Reduction %**	
	2008	**2009**	**2008**	**2009**	**2008**	**2009**	**2008**	**2009**
Nitrate as N	3.39	6.95	5.76	11.81	6.02	10.96	-4.6	7.3
Ammonium as N	0.0000	0.0000	0.0001	0.0000	0.0004	0.0003	-300.0	-500.0
Total Dissolved Solids	122.57	269.37	208.36	457.93	189.69	401.56	8.960	12.3
Suspended Sediment	0.12	0.48	0.20	0.81	1.33	0.73	-500.0	10.0
Orthophosphate as P	0.002	0.004	0.003	0.007	0.001	0.001	63.2	89.5

6. Summary of Model Performance, Meetings, and Number of CDs Distributed

The model provides a robust and relatively easy to use tool for employment by growers, RCDs and other parties interested in estimating the size of vegetated treatment system needed for nitrate reduction. The VTS sizing model shows a current realistic range of VTS area needed for nitrate removal based on a literature review of three wetland types and the decay rate constants for each in addition to field research of local vegetated agricultural drainage ditches. Sediment can be removed by VTSs, however the substrate in HSSF type systems can be clogged by sediment, therefor sediment removal in advance of the wetland is recommended to prolong the life of the wetland. Ammonia was not included in the model due to the complexity of conversion processes between different forms of nitrogen in wetland and the desired focus on nitrate.

In November and December CSUMB personnel participated in 8 presentations of the use of the VTS Sizing model and findings regarding VTS design for nitrate reduction. A total of about 120 CDs have been distributed to meeting attendees and additional copies are available upon request. The Santa Cruz Resource Conservation District had model information translated into Spanish for the presentation on 12/2/11 and a translator was available to 7 Spanish-speaking growers from the Pajaro watershed present at this meeting. There were 4 hill slope growers who also attended this same meeting and will benefit from the use of the model. Translation to Spanish was also available at the meeting held in San Luis Obispo on 12/16/11 and 3 growers received assistance. There may be additional under-served growers who benefit from the model in the future.

Table 15. Events where the model was presented and VTS design discussed.

Event	Date	Location	Summary	Description	CDs Dist.
Farm Bureau sponsored event	11/4/2011	Monterey County Farm Bureau	Presented the research in algal community response to higher levels of nutrients in streams. Presented the VTS sizing Model and Findings	Attended by Monterey Co Farm Bureau Exec Director, 2 growers, Grower Shipper OrgN, NH3	7
Email to Grower	11/7/2011	Salinas Valley	Ross Jensen heard about the model and wanted a copy, which was emailed to him	Sent model CD, powerpoint pdf, a	1
Central Coast Water Quality Coalition Event	11/14/2011	Santa Clara County Farm Bureau	Presented the research in algal community response to higher levels of nutrients in streams. Presented the VTS sizing Model and Findings	17 attendees including Water Quality Coalition, Preservation Inc, NRCS and growers.	17
Monterey Bay CAPCACE	11/17/2011	Monterey Ag Commissioner	Marc Los Huertos presented biomonitoring field work and wetland model	39 Attendees	3
Central Coast Water Quality Coalition Event	11/18/2011	Santa Barbara Farm Bureau	Preservation Inc presented nitrate levels at monitored sites. WQ coalition presented farm plan template. CSUMB presented wetland sizing model.	19 attendees including grower shipper organization, growers and consultants	19
Central Coast Water Quality Coalition Event	11/21/2011	Monterey County Farm Bureau	Preservation Inc presented nitrate levels at monitored sites. WQ coalition presented farm plan template. CSUMB presented wetland sizing model.	Attendees included growers, RCDs, Preservation Inc., Consultants and other agribusiness (Dow).	20
Central Coast Water Quality Coalition Event	12/2/2011	Santa Cruz Farm Bureau	Preservation Inc presented nitrate levels at monitored sites. WQ coalition presented farm plan template. CSUMB presented periphyton and wetland sizing model.	25 attendees	22
Central Coast Water Quality Coalition Event	12/16/2011	San Luis Obispo Farm Bureau	Preservation Inc presented nitrate levels at monitored sites. WQ coalition presented farm plan template. CSUMB presented wetland sizing model.	27 attendees	30
Central Coast Water Quality Coalition Event	12/16/2011	CCRWQCB	Preservation Inc presented nitrate levels at monitored sites. WQ coalition presented farm plan template. CSUMB presented wetland sizing model. RCDs and NRCS discussed roles.	28 attendees	1

7. Future Work

The value of this model is the explicit inclusion of uncertainly. As more information is gathered in the region, this uncertainly can be reduced to ensure better designs. With the inclusion of locally collected data the model can be validated relative to the response of VTS systems to the nitrate concentrations, loading rates and climatic conditions found on California's Central Coast. Confidence in model output would increase if decay rates were parameterized through using the performance of local VTSs to determine the range of decay rates given the Central Coast's Mediterranean climate and higher nitrate concentrations than found in most regions where wetlands have been researched. Although we reviewed the performance of 4 local VTS systems for this project, none of these systems were woodchip bioreactors, which require the least area for nitrate removal. Furthermore, of the VTS systems we reviewed, only VTS4 was designed as a shallow constructed wetland with a hydraulic loading rate common to wetland systems; however it had not reached maturity. The three other systems were integrated with existing systems with a different primary purpose (e.g. sediment removal or drainage ditch), and were not designed primarily for nitrate removal. Vegetated treatment systems designed specifically for nitrate removal and allowed to mature to full performance would better indicate the anticipated performance of local systems in our region.

In determining regulatory procedure, it should be noted that VTS system performance changes for the first one to two years as plants grow, bacteria colonizes, and flow rates are optimized. Following the initial installation of a new system, a period of time may be necessary for system maturation, for dissolution of easily soluble DOC (in the case of woodchip bioreactors), and for modulation of flow rates to achieve proper operation. During this time period, government regulators should consider that performance is subpar and accept some excursions.

While VTS systems improve water quality, they also influence air quality. One potential issue with woodchip bioreactors and other wetland types is the release of nitrous oxide. Further study of VTSs for green house gas emissions would enable the design of systems that release the least proportion of nitrous oxide compared with N_2 during nitrate transformation.

Finally, the use of VTSs for nitrate removal is one BMP out of a number of potential options growers may want to consider, either singly or in combination. The ability to define the cost and effectiveness of multiple BMPs in particular settings will better allow growers to determine the most effective BMP choices for use on their farms. Also the ability to design a system of BMPs to particular crop needs, discharges and nitrate concentrations could enable growers to meet regulatory standards at the least cost. However, the costs of the design and construction and potential regulatory oversight is an important disincentive. Turning vegetative treatment systems into an incentive program will take creative and thoughtful work in the region.

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Appendix A: Wetland Sizing Model R Code

```
#  Creating a GUI for the Wetland Design Tools Grant
#  Pam Krone-Davis & Marc LosHuertos
#  v. 0.88 Jan. 1, 2012

require(gWidgets)
options("guiToolkit"="RGtk2")
WetlandTypeNames = c("Free Water Surface", "Horizontal Sub-Surface Flow", "Horizontal
      Sub-Surface Flow WoodChip", "Vegetated Ditch")
#***************************************************************************************
# Wetland Sizing Function
#***************************************************************************************
WetArea <- function( ConcIn.ppm, ConcOut.ppm, Discharge.gpm , H2O_Temp.degC, wet_type)
      {
      Q_m3pyr     = Discharge.gpm * 1991  # convert gpm to m3/year
      theta       = 1.088 ## temp dependent removal Arhenius Kadlec(2005)
      SF          = 1.8   # stochasticity factor recommended by Kadlec (2005)
      daysperyear = 365
      meterperfoot = 0.3048
      n = 10000
      ## Free Water Surface (FWS) Wetlands
      if ( wet_type == WetlandTypeNames[1])  {
            N = 3        ## median N value for FWS wetland from Kadlec 2009 is 4.5
       and median P = 3
            shape = 3.2 #.12   ## shape derived from least sse gamma fit to
       Kadlec(2009) fws wetlands
            scale = 9.045   ## scale derived from least sse gamma fit to Kadlec(2009)
       fws wetlands
            ka20   = rgamma( n, shape = shape, scale = scale)
            ka  = ka20 * (theta ^ (H2O_Temp.degC - 20))
            A.m2    = -N  * ((ConcOut.ppm/ConcIn.ppm)^(1/N)  - 1) *
       ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N))  * (Q_m3pyr / ka)
            ## convert to acres and multiply by stochastic factor, ie margin of safety
            A.acre  <<- round(A.m2 *    0.0002471 * SF, digits = 3)
          }
       ## Horizontal sub-surface flow  (HSSF) wetlands  with rock media
      if (wet_type == WetlandTypeNames[2])  {
             ## based on plot for denitrification k Kadlec (2009) of 216 wetlands
             ## a bimodal distribution is produced, prob exists due to some difference
       in VTS properties
            hssf =  data.frame(  k = c(2,7,26,35,40,42,47,75,85,95,105), percent =
       seq(0,1,by = 0.1))
            N <-  6     ## median P value for weathering considerations from Kadlec
       2009
            n = 10000
            ka20 = c( runif(n*0.1, min = hssf$k[1], max = hssf$k[2]),runif(n*0.1, min
       = hssf$k[2], max = hssf$k[3]),
                  runif(n*0.1, min = hssf$k[3], max = hssf$k[4]),runif(n*0.1, min =
       hssf$k[4], max = hssf$k[5]),
                  runif(n*0.1, min = hssf$k[5], max = hssf$k[6]),runif(n*0.1, min =
       hssf$k[6], max = hssf$k[7]),
                  runif(n*0.1, min = hssf$k[7], max = hssf$k[8]),runif(n*0.1, min =
       hssf$k[8], max = hssf$k[9]),
                  runif(n*0.1, min = hssf$k[9], max = hssf$k[10]),runif(n*0.1, min =
       hssf$k[10], max = hssf$k[11]))
            ka  = ka20 * theta ^ (H2O_Temp.degC - 20)     # units m/day
            A.m2    = -N  * ((ConcOut.ppm/ConcIn.ppm)^(1/N)  - 1) *
       ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N))  * (Q_m3pyr / ka)
            A.acre  <<- round(A.m2  *    0.0002471 * SF, digits = 3)
        }
      ### Horizontal Subsurface Flow with woodchip substrate (HSSF Chip)
      if (wet_type == WetlandTypeNames[3])   {
```

```
        N <-  6   ## assume similar hydrology to other HSSF reactors
        k20 = c(0.25,0.86,1.2,1.4, 2.2)  ## volumetric decay constants from
   Leverentz (2010), Roberston,  units hr^-1
        thetachip = 1.09
        dep1 = 4  *  meterperfoot  ## 2 feet depth in feet converted to meters
        dep2 = 8  *  meterperfoot
        kv  = k20 * thetachip ^ (H2O_Temp.degC - 20)
        porosity = 0.6  # void space in the reactor for water flow
        eff.dep1 = dep1 * porosity     ## depth corrected for porosity
        eff.dep2 = dep2 * porosity
        A.m2.dep1    = -N  * ((ConcOut.ppm/ConcIn.ppm)^(1/N)  - 1) *
   ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N))  * (Q_m3pyr /( kv * daysperyear *
   eff.dep1))
        A.m2.dep2    = -N  * ((ConcOut.ppm/ConcIn.ppm)^(1/N)  - 1) *
   ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N))  * (Q_m3pyr /( kv * daysperyear *
   eff.dep2))
        A.acre.dep1  = round(A.m2.dep1  *    0.0002471 * SF, digits = 5)
        A.acre.dep2  = round(A.m2.dep2  *    0.0002471 * SF, digits = 5)
        A.acre <<-  list(A.acre.dep1,A.acre.dep2)
      }

  if   (wet_type == WetlandTypeNames[4]){  ### vegetated ditch
        N = 11
        ka20 = c( 11.1, 13.9, 20.3)
        median(ka20)
        ka  = ka20 * theta ^ (H2O_Temp.degC - 20)
        A.m2    = -N  * ((ConcOut.ppm/ConcIn.ppm)^(1/N)  - 1) *
   ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N))  * (Q_m3pyr / ka)
        A.acre  <<- round(A.m2  *    0.0002471 * SF, digits = 3)
      }
  }

*********************************************************************************
#####  Display Median Size for All Wetland Types on R Console AND Size with Safety
       Factor Recommended by Kadlec (2005)
*********************************************************************************
WetAreaDF =   function( ConcIn.ppm, ConcOut.ppm, Discharge.gpm , H2O_Temp.degC,
   wet_type){
  NAs = rep( NA, 11 )
  est = data.frame( WetlandType=c(WetlandTypeNames, "",  "", "",WetlandTypeNames),
   MedianArea_acres = NAs, MedianArea_With_SafetyFactor = NAs)
  N_FWS=3; N_HSSF = 6; N_Ditch = 11 ; meterperfoot = 0.3048; daysperyear = 365 ;
   gpm_m3pday = 5.451; theta = 1.088; theta_chip =1.09
  kv.chip = 1.2  ; eff.dep2 =  4  *  meterperfoot * 0.6; ka.vegditch.median = 13.9
   ; minperday = 60 * 24; meter2peracre = 4047
  SF = 1.8  # Kadlec 2005 Stochastic safety factor  ;
  dep = 0.3 #  meters
  m3pergallon = 0.003785
  ka.hssf.median = 42 ; ka.fws.median = 27 # Kadlec 2009    m/yr
  acreperm2 =   0.0002471 ; Q_m3pyr     = Discharge.gpm * 1991

  est[1,2]    =  round(((- N_FWS)  * (((ConcOut.ppm/ConcIn.ppm)^(1/N_FWS))  - 1) *
   ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N_FWS))   *
                             (Q_m3pyr / (ka.fws.median * (theta ^ (H2O_Temp.degC-
   20))  )* acreperm2) ) , digits =3)
  est[2,2]    =  round(((- N_HSSF)  * (((ConcOut.ppm/ConcIn.ppm)^(1/N_HSSF))  - 1)
   * ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N_HSSF))   *
                             (Q_m3pyr / (ka.hssf.median * (theta ^ (H2O_Temp.degC-
   20))  )* acreperm2) ) , digits =3)
  est[3,2]    =  round((- N_HSSF  * ((ConcOut.ppm/ConcIn.ppm)^(1/N_HSSF)  - 1) *
   ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N_HSSF))   *
               (Q_m3pyr /( kv.chip * (theta ^ (H2O_Temp.degC-20)) * daysperyear
   * eff.dep2))  * acreperm2) , digits = 3)
```

```
      est[4,2]    =  round(((- N_Ditch)  * (((ConcOut.ppm/ConcIn.ppm)^(1/N_Ditch))  -
       1) * ((ConcOut.ppm/ConcIn.ppm) ^ (-1/N_Ditch))  *
                                 (Q_m3pyr / (ka.vegditch.median * (theta_chip ^
       (H2O_Temp.degC-20))  )* acreperm2) ) , digits =3)
      est[1,3]    =  est[1,2]  * SF  ; est[2,3] = est[2,2]  * SF  ;est[3,3] = est[3,2]
       * SF
      est[4,3]    =  est[4,2]  * SF
      est[8:11,2] =  round((100*(ConcIn.ppm - ConcOut.ppm)/ConcIn.ppm ), digits = 1)
      est[8,2]    =  round(  est[1,3] * meter2peracre * dep /(Discharge.gpm *
       minperday * m3pergallon) , digits = 2)
      est[9,2]    =  round(  est[2,3] * meter2peracre * dep /(Discharge.gpm *
       minperday * m3pergallon) , digits = 2)
      est[10,2]   =  round(  est[3,3]* meter2peracre * eff.dep2  /(Discharge.gpm *
       minperday * m3pergallon ) , digits = 2)
      est[11,2]   =  round(  est[4,3] * meter2peracre * dep /(Discharge.gpm *
       minperday * m3pergallon ) , digits = 2)
      est[8,3]    =  round((ConcIn.ppm - ConcOut.ppm) * ( Q_m3pyr / (est[1,2]/
       acreperm2 )  ) , digits = 2)
      est[9,3]    =  round((ConcIn.ppm - ConcOut.ppm) * ( Q_m3pyr / (est[2,2]/
       acreperm2 )  ) , digits = 2)
      est[10,3]   =  round( (ConcIn.ppm - ConcOut.ppm) * ( Q_m3pyr / (est[3,2]/
       acreperm2 )  ) , digits = 2)
      est[11,3]   =  round((ConcIn.ppm - ConcOut.ppm) * ( Q_m3pyr / (est[4,2]/
       acreperm2 )  ) , digits = 2)
      est[5,2]    =  c(" ")  ; est[5,3] = c(" ") ;est[6,2] = c("Nominal Hydraulic")
       ;est[6,3] = c("Nitrate_Load_Reduc")
      est[7,2]    =  c("Retention Time (days)") ;est[7,3] = c("g/m2-yr")
      print(est)
      #return(est)
      }
#**************************************************************************************
#####  Graphics Function

#***************************************************************************************

graphics.fun = function( ConcIn.ppm, ConcOut.ppm, Discharge.gpm , H2O_Temp.degC,
       wet_type, A.acre){

            plot_width                  = 10
            plot_height                 = 10
            windows(plot_width,plot_height, rescale = "fit")
            #win.graph(plot_width,plot_height)
            n = 10000

          if (wet_type == WetlandTypeNames[1]| wet_type == WetlandTypeNames[2]){
            A.acre = sort(A.acre)
            mean.acre = round(mean(A.acre), digits = 2)
            median.acre = round(median(A.acre), digits = 2)
            CI.acre = A.acre[(1*n*0.05) :(1*n*0.95 ) ]
            CI.upper95 =  max(CI.acre)
            CI.lower95 =  min(CI.acre)
            n =   10000
            ### some of the acres go to very large sizes, so lets limit the size of
       the graph to eliminate 2% worst
            ##  this gives a better display of the expected size needs
            per = A.acre[1:(n*0.98)]
            maxi = max(per)
            #maxi = round(maxi, digits = 0 )
            # set lengths and locations of windows for plots:

            par(mfrow=c(2,1))
            Hist = hist(A.acre, breaks = n*0.02, plot = F)
            Ymax = c(max(Hist$density)+ 0.1)
```

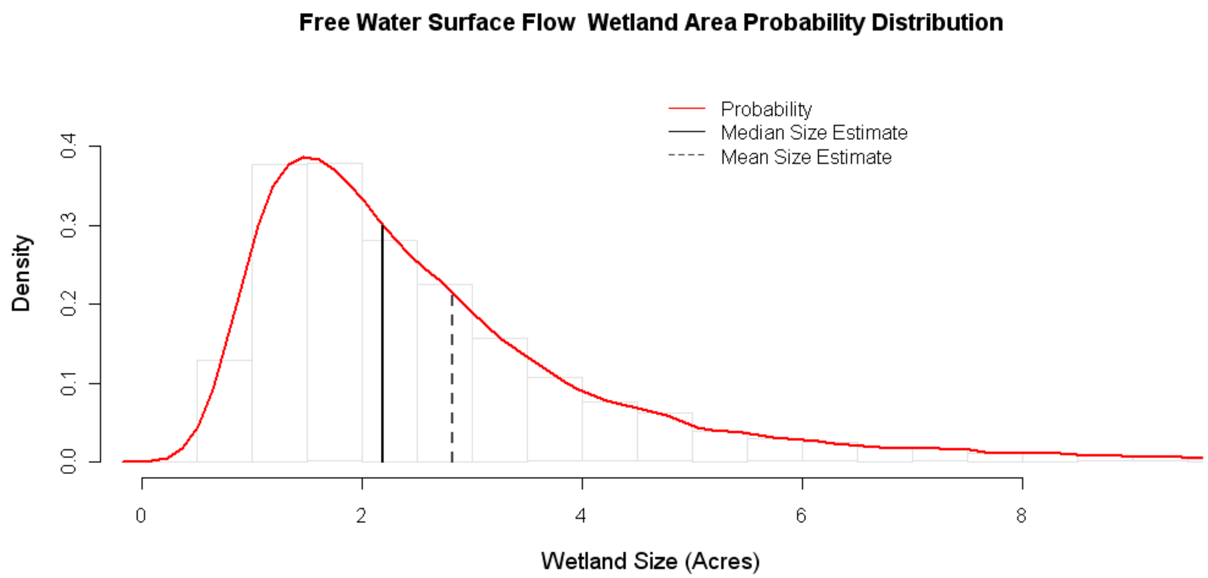
```
     Hist = hist(A.acre, xlim = c(0,maxi),ylim = c(0,Ymax),breaks = n*0.02,
freq = F,xlab = "VTS Size (Acres)",
          main = paste(wet_type," VTS Area Probability Distribution"), border =
"grey88",cex.lab = 1.2  )
     den0 = density( A.acre, adjust=1)
     lines(den0$x,den0$y, col = "red", lwd = 2)
     y.mean =  approx(den0$x,den0$y, xout = c(mean.acre))$y
     lines(c(mean.acre,mean.acre),c(0,y.mean),col="grey28",lwd=2, lty = 2)
     y.med =  approx(den0$x,den0$y, xout = c(median.acre))$y
     lines(c(median.acre,median.acre),c(0,y.med),col="black",lwd=2, lty =1)
     legend( maxi * 0.5, Ymax, leg=c("Probability","Median Size Estimate",
"Mean Size Estimate"),
          col = c("red","black","grey28"),lty=c(1,1,2), bty= 'n')
     ## Print the model input and results in a plot on the same graphics
console
     par(mar=c(2,1,2,1)+.4)
     plot(c(-5,5), c(-1,5), type = "n", xlab=" ", ylab=" ", xaxt = "n", yaxt =
"n", asp = 1)
     text(0,4.8, paste("VTS Type: ",wet_type), cex = 1.2, adj=0.5)
     text(-2,4.0, paste("Model Inputs"), pos=2, cex = 1.2, adj=0)
     text(-1, 3, paste("Discharge (gpm) = ",Discharge.gpm), pos=2, adj=3, col =
"blue", cex = 1.1)
     text(-1, 2.0, paste("Inlet Concentration (ppm)= ",ConcIn.ppm), pos=2,
adj=0, col = "blue", cex = 1.1)
     text(-1, 1, paste("Outlet Concentration (ppm) Target = ",ConcOut.ppm),
pos=2, adj=0, col = "blue", cex = 1.1)
     text(-1,0, paste("Water Temperature (deg. C) = ",H2O_Temp.degC), pos=2,
adj=0, col = "blue", cex = 1.1)
     #Outputs
     text(2,4, paste("Model Outputs"), pos=4, adj=0, cex=1.2)
     text(5.5,3, paste("Median VTS Area (acres) = ",median.acre), pos=2, adj=0,
col = "red", cex = 1.1)
     text(5.4,2,  paste("Mean VTS Area (acres) = ",mean.acre), col = "red",
pos=2, adj=0, cex = 1.1)
     text(6,1,  paste("Percent Concentration Reduction =",
round(100*((ConcIn.ppm - ConcOut.ppm)/ConcIn.ppm),0), "%"),
         col="red", adj=0, cex = 1.1, pos=2)
     text(5.5,0,  paste("95% Confidence Intervals (acres) = "), col="red",
adj=0,  pos=2, cex = 1.1)
     text(4.5,-0.5, paste(CI.lower95, " to ",CI.upper95), col="red", adj=0, cex
= 1.1)
      }
 if (wet_type == WetlandTypeNames[3]) {
     par(mfrow=c(3,1))
     par(mai = c(0.8,1.2,0.2,0.2))
     #hist(A.acre, xlim = c(0,maxi),breaks = n*0.002, freq = F,xlab = "Wetland
Size (Acres)",
         # main = paste(wet_type," Wetland Area Probability Distribution"),
border = "grey88",cex.lab = 1.2  )

     sqftperacre = 43560
     width = 10 # feet wide
     length_ft = 0.006 * sqftperacre/width
     length_ft_4 = A.acre[[1]] * sqftperacre/width
     length_ft_8 = A.acre[[2]] * sqftperacre/width
      ymax = max(length_ft_4 )
     ## Plot the wetland acreage over the range of kv decay rates found in the
literature at 2 depths
     barplot (length_ft_4, main = c("Estimated HSSF Woodchip Length Range,
Depth = 4 ft, Width = 10 ft"),
               col = c("lightblue"), names.arg =  c( "k = 0.25", "k = 0.86","k
= 1.2","k = 1.4","k = 2.2")  ,
                bty="n" , las = 1, cex.names = 1.3,cex.main = 1.5,
```

```
                cex.axis= 1.2,  cex.lab = 1.4, xlab = "Volumetric Nitrate Decay
Rate (/day)", ylim = c(0,ymax))
     op <- par(mar = c(5,7,4,2) + 0.1)
     title( ylab = "Length (feet)", cex.lab = 1.6)
     #text(length_ft_4, signif(length_ft_4,0), pos=4)
     lar = 1.3
     text(0.7, (length_ft_4[1]-700),round(length_ft_4[1],0), cex = lar )
     text(1.9, (length_ft_4[2]+500),round(length_ft_4[2],0), cex = lar )
     text(3.1, (length_ft_4[3]+400),round(length_ft_4[3],0), cex = lar )
     text(4.3, (length_ft_4[4]+400),round(length_ft_4[4],0), cex = lar )
     text(5.5, (length_ft_4[5]+400),round(length_ft_4[5],0), cex = lar )


#text(x=barplot(length_ft_4),y=length_ft_4,label=format(length_ft_4),pos=3)
     #text(x=length_ft_4,label=format(round(length_ft_4,0)),pos=3)
     par(mai = c(0.8,1.2,0.2,0.2))
     barplot (length_ft_8, main = c("Estimated HSSF Woodchip Length Range,
Depth = 8 ft, Width = 10 ft"),
                col = "lightcyan2", names.arg =  c( "k = 0.25", "k = 0.86","k =
1.2","k = 1.4","k = 2.2") ,
                 bty="n" , ylim = c(0,ymax), las = 1, xlab = "Volumetric Nitrate
Decay Rate (/day)",
                cex.axis= 1.2, cex.lab = 1.4, cex.names = 1.3, cex.main = 1.5)
     op <- par(mar = c(5,7,4,2) + 0.1)
     title( ylab = "Length (feet)", cex.lab = 1.6)
     text(0.7, (length_ft_8[1]+400),round(length_ft_8[1],0), cex = lar )
     text(1.9, (length_ft_8[2]+400),round(length_ft_8[2],0), cex = lar )
     text(3.1, (length_ft_8[3]+400),round(length_ft_8[3],0), cex = lar )
     text(4.3, (length_ft_8[4]+400),round(length_ft_8[4],0), cex = lar )
     text(5.5, (length_ft_8[5]+400),round(length_ft_8[5],0), cex = lar )
     ### displAY model output
     par(mar=c(2,1,2,1)+.4)
     plot(c(-5,5), c(-1,5), type = "n", xlab=" ", ylab=" ", xaxt = "n", yaxt =
"n", asp = 1)
     text(0,4.8, paste("VTS Type: ",wet_type), cex = 1.2, adj=.5)
     text(-3,4.0, paste("Model Inputs"), pos=2, cex = 1.2, adj=0)
     text(-2, 3, paste("Discharge (gpm) = ",Discharge.gpm), pos=2, adj=0, col =
"blue", cex = 1.2)
     text(-2, 2.0, paste("Inlet Concentration (ppm)= ",ConcIn.ppm), pos=2,
adj=0, col = "blue", cex = 1.2)
     text(-2, 1, paste("Outlet Concentration (ppm) Target = ",ConcOut.ppm),
pos=2, adj=0, col = "blue", cex=1.2)
     text(-2,0, paste("Water Temperature (deg. C) = ",H2O_Temp.degC), pos=2,
adj=0, col = "blue", cex=1.2)
     #Outputs
     text(1,4, paste("Model Outputs"), pos=4, adj=0, cex=1.2)
     text(0,3,  paste("Length (ft) range of 10 x 4 ft Wetland   = ",
          round(min(length_ft_4),0)," to ",round(max(length_ft_4),0)), adj=0,
col = "red", cex=1.2)
     text(0,2,  paste("Length (ft) range of 10 x 8 ft Wetland   = ",
          round(min(length_ft_8),0)," to ",round(max(length_ft_8),0)), col =
"red", adj = 0, cex=1.2)
     text(6,1,  paste("Percent Concentration Reduction =",
round(100*((ConcIn.ppm - ConcOut.ppm)/ConcIn.ppm),0), "%"),
          cex = 1.2, col="red", adj=0,  pos=2)
     text(0,0,"HSSF Woodchip VTS are recent and insufficient",adj = 0, col =
"red", cex=1.2)
     text(0,-0.5,"research exists to develop 95% Confidence Intervals",adj = 0,
col = "red", cex=1.2)
  }

  if (wet_type == WetlandTypeNames[4]) {
     par(mfrow=c(2,1))
     par(mai=c(1,1,0.5,0.5))
```

```
        mean.acre = round(mean(A.acre), digits = 2)
        median.acre = round(median(A.acre), digits = 2)
        Yupper =  ceiling(max(A.acre))
        barplot (A.acre, main = c("Estimated Vegetated Ditch Area (acres)"), ylim
  = c(0,Yupper),
                    col = c("lightblue"), names.arg =  c( "k = 0.10", "k = 0.13","k
  = 0.19") ,
                    ylab = "Acres", bty="n", las = 1, xlab = "Volumetric Nitrate
  Decay Rate (/day)" )
        ## Display model input and output on the graphics console
        par(mar=c(2,1,2,1)+.4)
        plot(c(-5,5), c(-1,5), type = "n", xlab=" ", ylab=" ", xaxt = "n", yaxt =
  "n", asp = 1)
        text(0,4.8, paste("VTS Type: ",wet_type), cex = 1.2, adj=.5)
        text(-1,4.0, paste("Model Inputs"), pos=2, cex = 1.2, adj=0)
        text(-1, 3, paste("Discharge (gpm) = ",Discharge.gpm), pos=2, adj=0, col =
  "blue")
        text(-1, 2.0, paste("Inlet Concentration (ppm)= ",ConcIn.ppm), pos=2,
  adj=0, col = "blue")
        text(-1, 1, paste("Outlet Concentration (ppm) Target = ",ConcOut.ppm),
  pos=2, adj=0, col = "blue")
        text(-1,0, paste("Water Temperature (deg. C) = ",H2O_Temp.degC), pos=2,
  adj=0, col = "blue")
        #Outputs
        text(2,4, paste("Model Outputs"), pos=4, adj=0, cex=1.2)
        text(6,3, paste("Median VTS Area (acres) = ",median.acre), pos=2, adj=0,
  col = "red")
        text(5.8,2,  paste("Mean VTS Area (acres) = ",mean.acre), col = "red",
  pos=2, adj=0)
        text(6.4 ,1,  paste("Percent Concentration Reduction =",
  round(100*((ConcIn.ppm - ConcOut.ppm)/ConcIn.ppm),0), "%"),
            col="red", adj=0,  pos=2)
        text(5.3 ,0.0,  paste("Insufficient research exists to "), col="red",
  adj=0,  pos=2)
        text(5.6,-0.5,  paste("develop 95% Confidence Intervals "), col="red",
  adj=0,  pos=2)
        }
    }
#************************************************************************************
#####   User Interface and Model Run Functions
#************************************************************************************
  input.fun <- function(ConcIn.ppm=100, ConcOut.ppm = 10, Discharge.gpm = 20,
  H2O_Temp.degC = 15,
        wet_type = c("Free Water Surface", "Horizontal Sub-Surface Flow",
  "Horizontal Sub-Surface Flow WoodChip",
                    "Vegetated Ditch"),
        Graphic = TRUE)
   {
        WetArea(ConcIn.ppm, ConcOut.ppm, Discharge.gpm , H2O_Temp.degC, wet_type)
        WetAreaDF(ConcIn.ppm, ConcOut.ppm, Discharge.gpm, H2O_Temp.degC,wet_type)
        assignto = F
        if(Graphic=="TRUE")
        {
       graphics.fun(ConcIn.ppm, ConcOut.ppm, Discharge.gpm, H2O_Temp.degC,
  wet_type, A.acre)
  # test graphics.fun(100, 10, 5, 8, WetlandTypeNames[2], A.acre=A.acre)
        }}
 # output_path=(
  "C:\\Users\\pam\\Documents\\pesticide\\DNTIS_Module_Decay110523\\plots\\" )
#************************************************************************************
####  Setting up gWidgets
#************************************************************************************
ggenericwidget(input.fun, container=TRUE)
```

Appendix B: Examples of Wetland Sizing Model Graphic Output

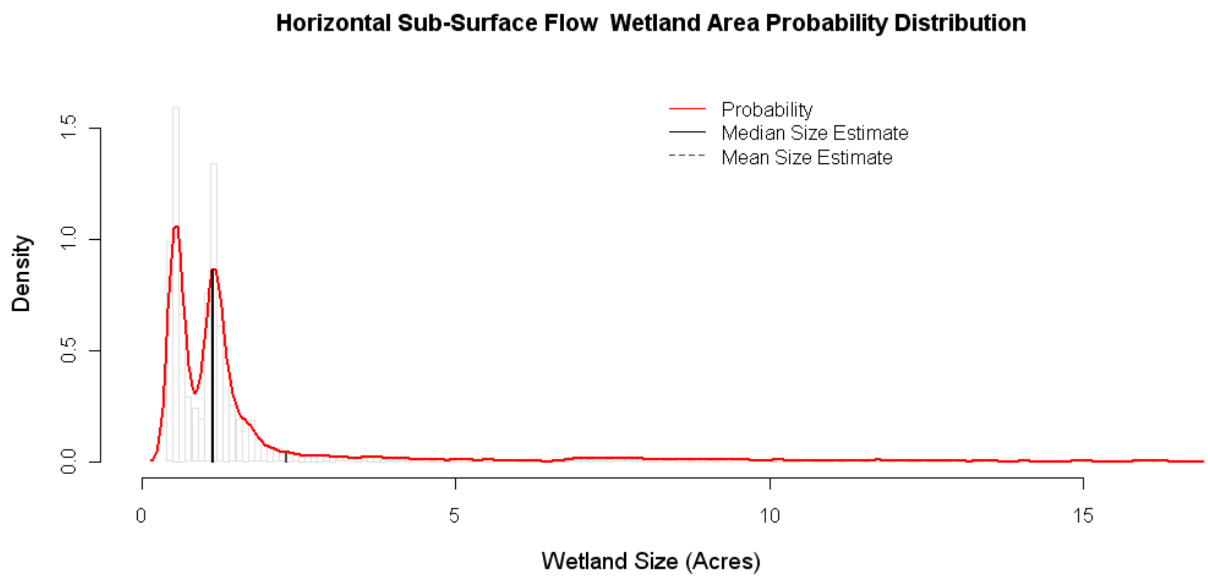
Figure B1. Free water surface wetland size probability distribution for nitrate reduction based on user inputs and the decay rate distribution frequency in Kadlec (2009) plus the safety factor of 1.8 (Kadlec 2005). The largest area (top 2%) wetlands are not included in the model graphic in order to better display the most probable range.



Wetland Type: Free Water Surface Flow

Model Inputs	Model Outputs
Discharge (gpm) = 20	Median Wetland Area (acres) = 2.19
Inlet Concentration (ppm)= 60	Mean Wetland Area (acres) = 2.83
Outlet Concentration (ppm) Target = 10	95% Confidence Intervals (acres) =
Average Water Temperature (C) = 17	0.946 to 6.743

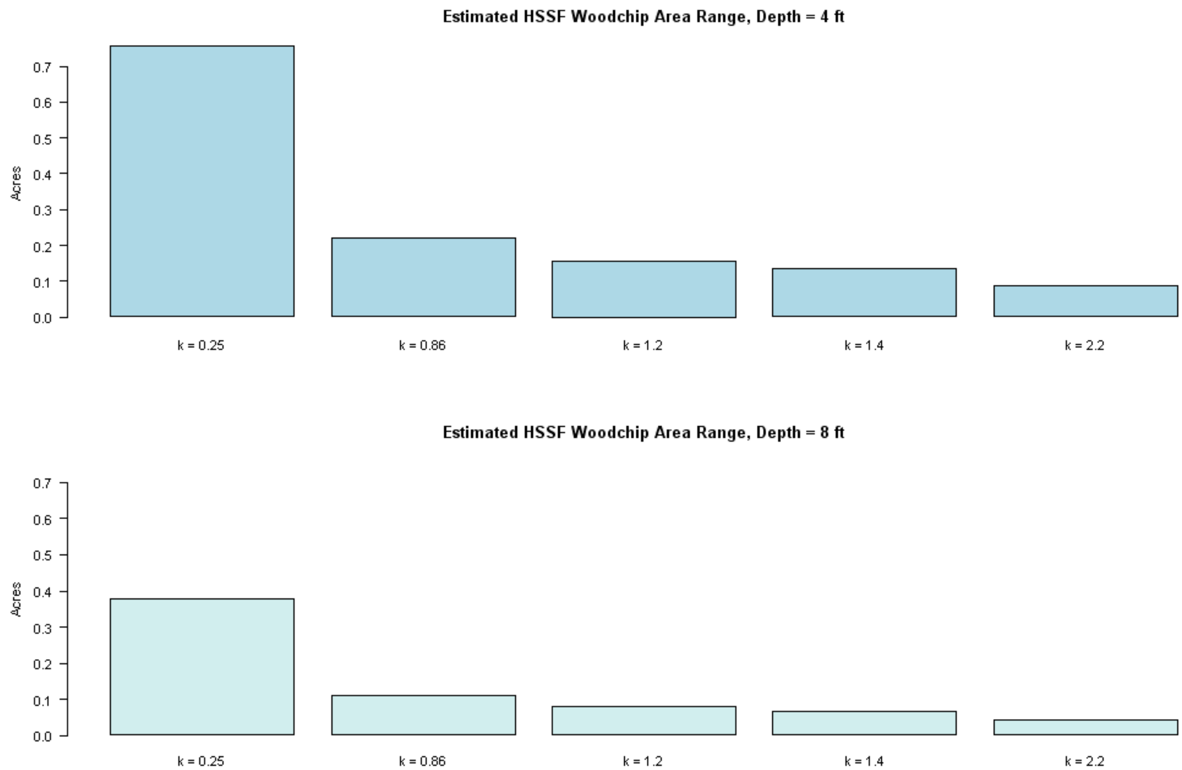
Figure B2. Horizontal sub-surface flow wetland size probability distribution for nitrate reduction based on user inputs and the decay rate distribution frequency in Kadlec (2009) plus the safety factor of 1.8 (Kadlec 2005). The reason for a bimodal distribution is the relatively large proportion of wetlands with nitrate decay rates in two ranges, from 7 – 26 m/yr and from 42 – 75 m/yr. The largest area (top 2%) wetlands are not included in the model graphic in order to better display the most probable range. The very long tail displays the poor nitrate reduction performance and high area requirement of a fraction of HSSF wetlands.



Wetland Type: Horizontal Sub-Surface Flow

Model Inputs	Model Outputs
Discharge (gpm) = 20	Median Wetland Area (acres) = 1.13
Inlet Concentration (ppm)= 60	Mean Wetland Area (acres) = 2.3
Outlet Concentration (ppm) Target = 10	95% Confidence Intervals (acres) =
Average Water Temperature (C) = 17	0.477 to 10.826

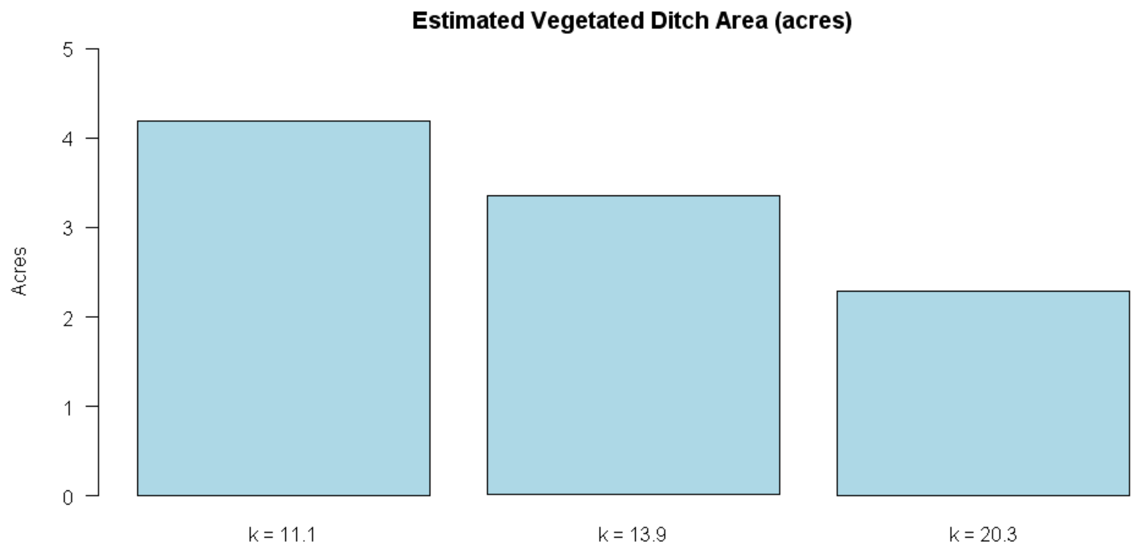
Figure B3. Horizontal sub-surface flow wetland size based on decay rates from the research of Leverentz et al. (2010), Robertson (2010), Moorman et al. (2010) plus the safety factor of 1.8 (Kadlec 2005). Two different depths are displayed, and it is possible to interpolate additional depths.



Wetland Type: Horizontal Sub-Surface Flow WoodChip

Model Inputs	Model Outputs
Discharge (gpm) = 20	Range of Wetland Area (acres) at 4 ft depth = 0.086 to 0.755
Inlet Concentration (ppm)= 65	Range of Wetland Area (acres) at 8 ft depth = 0.043 to 0.377
Outlet Concentration (ppm) Target = 10	HSSF Woodchip Wetlands are recent and insufficient research exists to develop 95% Confidence Intervals
Average Water Temperature (C) = 17	

Figure B4. Vegetated ditch size estimates based on derived decay rates from 2007, 2008 and 2009 for a vegetated ditch in Salinas plus the safety factor of 1.8 (Kadlec 2005).



Wetland Type: Vegetated Ditch

Model Inputs	Model Outputs
Discharge (gpm) = 20	Median Wetland Area (acres) = 3.35
Inlet Concentration (ppm)= 65	Mean Wetland Area (acres) = 3.28
Outlet Concentration (ppm) Target = 10	Insufficient research exists to develop 95% Confidence Intervals
Average Water Temperature (C) = 17	

Appendix C: Water Temperatures at Local Vegetated Treatment Systems