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Constructed Treatment Wetlands for Reducing Nutrient Loading in the Lower Malibu Creek Watershed

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Doctor of Environmental Science and Engineering

by

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DEDICATION

This dissertation is dedicated to

my parents Dr. Calistus Nwaigwe and Chief Rosemary Nwaigwe for setting the bar high and steering me towards it

my husband Emmanuel Amah for his unwavering support

my sons Uka and Nachu (a.k.a “the people”), and my daughter Chechi who made this project *just a little harder* and its completion *so much sweeter*.

and

my friend Cathy Chang for providing all the emotional support and encouragement I could ever need.

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ABSTRACT OF THE DISSERTATION

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Doctor of Environmental Science and Engineering

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Nutrient enrichment by point and non-point source discharges is a major concern in the Malibu Creek Watershed located in Los Angeles County California. The basis for this concern is the persistent and extensive "nuisance" algae growth that plague Malibu Creek and Lagoon in the late spring and summertime. These algae blooms interfere with the recreational enjoyment of these waters and have the potential to cause low-oxygen conditions in aquatic habitat. While nitrogen and phosphorus loading have been identified as major contributors to the problem, studies have indicated that other physical and environmental conditions, including flow velocity and light intensity, play a role in the occurrence of algae blooms.

Constructed wetlands are a low-cost, low technology means of improving water quality with the added benefit of habitat enhancement and/or creation. They are effective to varying degrees in the removal of pollutants such as nutrients, suspended solids pathogens and biochemical oxygen demand from source waters; and would be useful in addressing water quality problems on a watershed scale.

This dissertation provides a comprehensive overview of the algae problem in the Malibu Creek Watershed and efforts taken to curb it to date. A pilot-scale constructed wetland study is conducted to determine the extent of nitrogen reduction attainable when treating effluent from the single point source discharge in the watershed. This discharge is a tertiary-treated effluent that is high in nitrate and low in organic carbon. Results from this study indicate that significant nutrient reduction is achievable even with this type of influent. In the absence of readily available carbon sources, plant uptake was the primary removal mechanism of nitrate – as opposed to denitrification.

Malibu Creek and Lagoon will also benefit from the use of constructed wetlands to treat septic tank effluent and urban run-off in creek flows. However the lack of readily available land in the watershed limits opportunities for wetland creation. Thus, in the Malibu Creek Watershed, it is necessary to involve public agencies, with significant land holdings, in plans to create and operate constructed wetlands for the purpose of water quality enhancement.

CHAPTER 1: INTRODUCTION

1.1 Introduction

The Basin Plan for the Coastal Watersheds of Los Angeles and Ventura Counties (1994) states that "waters shall not contain biostimulatory substances in concentrations that promote aquatic growth to the extent that such growths cause nuisance or adversely affect beneficial uses". This is a water quality objective that Malibu Creek and lagoon have been unable to meet, as they have been plagued with a proliferation of summer algae blooms for years. This is a major issue of concern since the algae interferes with the recreational use of the waters and can also create undesirable conditions of low dissolved oxygen levels which could lead to fish kills and subsequent objectionable odors. The 1996 State Water Resources Control Board (SWRCB) Water Quality Assessment Report listed this creek as an impaired waterbody and cited excess nutrients as a high priority concern. These nutrients come from a variety of sources including a wastewater treatment plant, septic systems, golf courses, and livestock.

Efforts to curb nutrient enrichment to date have included:

(i) Regulatory action geared towards reducing loading from the treatment plant - which is the sole point source discharge to the creek and more recently, septic systems

(ii) Operational changes and the constant investigation of new technology by the owners and operators of the treatment plant

(iii) Efforts by environmental groups to identify sources of pollution

(iv) Promotion of the use of best management practices (BMPs) by watershed groups and environmental agencies.

This dissertation has its basis in the investigation of constructed wetland technology to reduce nutrient loading to the watershed. The Las Virgenes Municipal Water District – co-owners and operators of the treatment plant discharging nitrified tertiary effluent to Malibu Creek - initiated the study. The goal was to create a dual-purpose wetland for treating tertiary effluent from their water reclamation facility, and a portion of Malibu Creek flows.

In many parts of the country, constructed wetlands are considered a cost-effective, low-maintenance way to improve water quality via natural processes. These systems have been used to reduce significant levels of nutrients, metals, and trace organic compounds as well as high levels of biochemical oxygen demand (BOD), suspended solids (SS), and coliform bacteria. This dissertation examines the potential of treatment wetlands to assist in the control of pollutants of concern in the Malibu Creek watershed.

1.2 Research Objectives

The objective of this research was to

(i) Provide a comprehensive overview of the water quality issues in the Malibu Creek watershed as they pertain to the eutrophication problems, including a literature review of nitrate removal in subsurface flow (SSF) constructed wetlands.

(ii) Design and run a pilot-scale wetland study to determine the extent of nitrate and phosphate removal possible in a carbon limited SSF wetland treating nitrified tertiary effluent

(iii) Examine the potential impact of wetland technology in the watershed based on results of the pilot studies; and provide insight to issues raised upon implementation in the Malibu Creek Watershed.

1.3 Organization of Dissertation

This dissertation is organized into three main sections. Chapter 2 provides background information on water quality in Malibu Creek with regard to historical nutrient levels and algae blooms, and the steps taken to improve these conditions to date. It includes an introduction to constructed wetlands and their ability to reduce nutrient and coliform bacteria concentrations in wastewater. A summary of treatment wetland use in California is provided.

Chapter 3 presents the results of a pilot-scale study designed to determine the effectiveness of a subsurface flow constructed wetland in removing nitrate and phosphate from the nitrified effluent of LVMWD's treatment plant. Sampling and analysis conducted during the initial stages of the study were preliminary. Complications regarding water level fluctuations within the cells during this period lead to the decision to exclude results obtained prior to the stabilization of operational conditions. Data collected during the initial stages of the project are provided in Appendix A. A literature review on studies of nitrate removal in other subsurface flow systems is included in this section.

Chapter four examines the potential impact of siting wetlands in the watershed based on the results of the pilot-scale study. It also provides information on the planning and design of a demonstration-scale wetland to be sited in the watershed, along with constraints and potential benefits of the technology; based on the results of the pilot-study and other relevant literature.

The conclusions of the dissertation summarize the key issues addressed in the preceding chapters. The potential of constructed wetland technology as a watershed management tool is highlighted. In addition, future research opportunities for the control of algae in the watershed are presented.

CHAPTER 2: BACKGROUND & LITERATURE REVIEW

2.1 The Malibu Creek Watershed

The Malibu Creek Watershed lies in north-west Los Angeles County, and is located in the greater Santa Monica Bay watershed. It covers an area of approximately 109 square miles (mi^2) and includes the cities of Malibu, Calabasas, Agoura Hills, Hidden Hills, Westlake Village, and Thousand Oaks, and unincorporated areas of Ventura and Los Angeles counties (Figure 2-1). The watershed has a Mediterranean climate characterized by a mild rainy season from April to November and a long warm dry season from May to October.

Much of the land in the watershed is publicly owned and recreational activities on these lands constitute the greatest land use in the area (SMBRP, 1993). The Santa Monica Mountains National Recreation Area, operated by the National Park Service, has an area of 10.5 mi^2, and the holdings of the California State Department of Parks and Recreation totals 13.6 mi^2 (USDA, 1995). Ventura and Los Angeles Counties and various cities within the watershed hold title to land for parks, and other public uses (USDA, 1995).

The Las Virgenes Municipal Water District (LVMWD) and the Triunfo County Sanitation District provide sewer service for close to 80,000 residents, representing most of the urbanized area. Also, approximately 2400 residential and commercial septic systems are used throughout the watershed (Tetra Tech Inc, 2001).

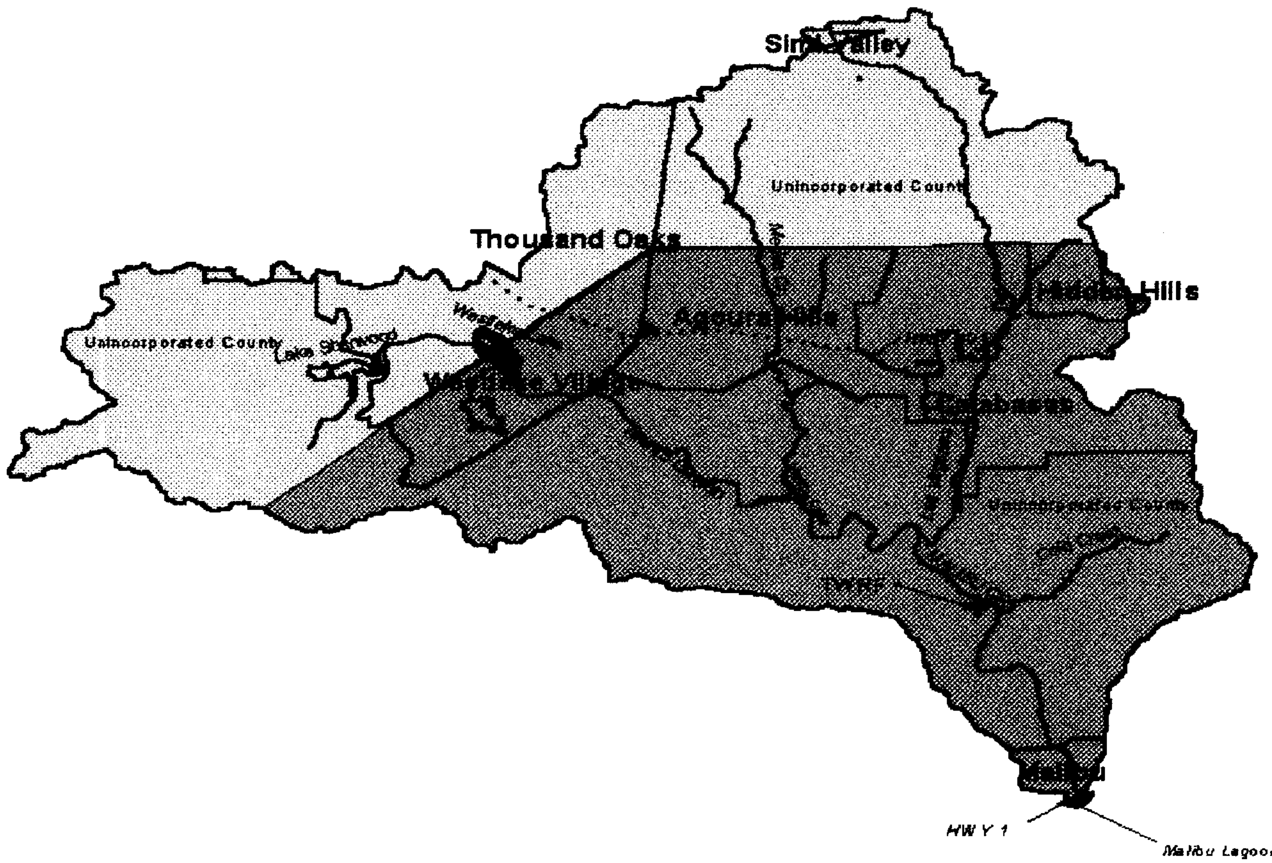


Figure 2-1: The Malibu Creek Watershed.

Ecological Significance

Malibu Lagoon is the only brackish lagoon in the Santa Monica Bay. It provides critical habitat for migrating birds and serves as transition habitat for steelhead trout prior to their spawning runs up Malibu Creek, and for the young steelhead returning to the ocean. It also has a population of an endangered species of fish, the tidewater goby (Eucyclogobius newberyi), that was reintroduced into it. The lagoon is one of the few estuaries on the south coast that drain into the Santa Monica Bay. The lagoon is also adjacent to Surfrider Beach, which is a popular tourist attraction in part due to the prime surfing conditions found there. Due to the ecological importance of the area, there is a strong regulatory presence. Table 2-1 lists the major agencies involved in the oversight of the watershed.

2.1.1 Hydrology of the Watershed

Malibu Creek flows for 8.5 miles through the lower portion of the watershed. It has its headwaters in Malibou Lake, which is fed by Triunfo Canyon and Medea Creeks. Other major tributaries include Las Virgenes Creek and Cold Creek (Figure 2-1). Flow volumes are high during and immediately after storm events that occur in the fall, winter and spring. In summertime, flows are generally lower and some parts of the lower Malibu Creek run dry. The Creek flows into Malibu Lagoon, which empties into the Santa Monica Bay. For most of the summer the lagoon is closed by a sand berm, which builds up gradually during the spring. This berm regulates water levels in the lagoon and is washed out in the fall during rain events. Until recently, the berm was artificially breached by the California Department of Parks and Recreation or the Los Angeles

County Department of Public Works when water levels in the lagoon rose sufficiently to cause water quality problems in the lagoon and surrounding areas. Unauthorized breaching by local citizens is common in late fall.

The major source of water in the watershed is precipitation. The watershed received an annual average of 23 inches of rainfall during the 1997/98 and 1998/99-storm seasons, which produced an average run-off volume of 20060 acre-ft to the creek (LACDPW, 1999). Recycled water discharge from a domestic wastewater treatment facility, operated by LVMWD, is another significant water source- from mid November to mid-April.

The Tapia Wastewater Reclamation Facility (TWRF) is located approximately five miles upstream from the Malibu Lagoon (Figure: 2-1). It is a tertiary treatment plant with a capacity to handle 16 million gallons of wastewater per day - presently it treats about 10 (mgd). A portion of the effluent produced is recycled year round for irrigation or industrial uses throughout Malibu Creek Watershed and the balance is discharged directly into Malibu Creek. Recycling increases significantly during the summer months due to increased use of reclaimed water by irrigation customers. The average daily flow through the plant for 1996/97 was 8.8 mgd and the average amount discharged to the creek within this period was 2.4 mgd. Since 1999, the Los Angeles Regional Water Quality Control Board has prohibited discharges to the creek during the period from April 15th to November 15th.

Table 2-1: Regulatory agencies with Jurisdiction in the Malibu Creek Watershed.

Agency	Purpose	Jurisdiction
National Park Service	The preservation and conservation of natural resources in all national park lands.	Federal Park lands
U.S. Army Corps of Engineers	Protection of existing wetlands from adverse impacts.	Lower Malibu Creek and Lagoon
California Department of Parks and Recreation	Preservation of state lands in their natural condition.	State Parks lands – including Malibu Creek and Lagoon.
Los Angeles Regional Water Quality Control Board	Protection of water quality by reducing or eliminating introduction of pollutants into receiving waters.	Surface- and ground-water quality in the watershed
California Department of Fish and Game	Conservation, protection, restoration, and propagation of selected species of native fish and wildlife that are threatened with extinction.	All surface waters in the watershed
U.S. Fish and Wildlife Service	Conservation, protection, restoration, and propagation of selected species of native fish and wildlife that are threatened with extinction	All surface waters in the watershed
National Marine Fisheries Service	Protection, conservation, and management of marine resources. Protects the anadromous fishes in the watershed.	Malibu Creek and Lagoon
California Coastal Commission	Effective protection and careful development of the coastal zone.	Coastline, Malibu Lagoon

2.1.2 Water Quality in the Watershed

Two major water quality concerns in the Malibu Creek watershed are the proliferation of algae in Malibu Creek and Lagoon during the summer, and high bacterial counts throughout the year. The algae problem is believed to stem from an over enrichment by nutrients from point and non-point sources. The TWRF is the only point source discharger in the watershed and is a significant contributor of nutrients to the creek in the wet season. Non point sources of nutrients in the creek include septic tank discharges, storm-water runoff, natural organic matter and aerial deposition (CRWQCB, 1990). Comprehensive estimates of nutrient loading from different sources in the watershed, derived by Tetra Tech (2001), are presented in Table 2-2. While TWRF is responsible for 28.5% of the nitrogen load and 45.3% of the phosphorus on an annual basis, the balance, which is the majority, comes from non-point sources.

Historical Water Quality in the Creek and Lagoon

Water quality in Malibu Creek has been monitored since 1978 when Tapia began continuous discharge to Malibu Creek. Historical phosphate and nitrate levels in the creek are shown in Figures 2-2 and 2-3 respectively. Nutrient levels in the creek are impacted by effluent discharge as shown by the difference in nutrient levels immediately above and below the discharge. A reducing trend in nutrient concentrations is also evident, particularly in the mid-nineties and in the lower portion of the creek. A similar trend is depicted in Figure 2-4 which shows nitrate and phosphate levels in the lagoon over time. Despite this apparent reduction in nutrient concentration over time, nuisance algae have been increasing throughout the creek and in the lagoon (Figure 2-5).

The Algae Problem

The Los Angeles Regional Water Quality Control Board (LARWQCB) conducted a water quality assessment of Malibu Creek from October 1975 to December 1976 in order to determine if the beneficial uses were being met. The study noted a proliferation of algae in the upper and lower reaches of the creek, and in the lagoon from May to August. Most of the growth was identified as *Cladophora*, a long filamentous type of algae. Though significantly high levels of nitrate were noted downstream of effluent discharge, and in the vicinity of private septic systems along the creek, nitrogen objectives were not set. The basis for this decision was that background concentrations in the creek were high enough to stimulate algae growth (LARWQCB, 1977). This study occurred during the period when TWRF was prohibited from discharging its effluent for five months in the dry season.

After year-round discharge was granted to the district in 1978, downstream residents began to complain that this was the cause of algae in the creek. The California Coastal Commission directed a study to determine the impacts of year round effluent discharge on conditions within the creek. Chapman (1980) studied the algae population of Malibu Creek at locations above and below the effluent discharge point from May 1978 to June 1979. He documented the types of algae in the creek and their favorable growth conditions (Table 2-3). He identified *Cladophora glomerata* as the principal algae throughout the creek; and suggested that high flows, P: N ratios (in conjunction with high phosphorus levels), and light intensity favor the growth of this alga. The summer decline of *Cladophora*, in certain locations along the creek was attributed to slow moving waters in those areas. The winter decline was attributed to the scouring action of floodwater. He concluded that other conditions within the creek encouraged abundant growth of this alga and that the impact of the nutrients and water flow from the effluent discharge was minimal. Other factors such as changes in temperature and flow velocity were credited for the cyclical changes in the nature and abundance of algae throughout the creek.

In a study of Malibu Creek and Lagoon, Ambrose et al. (1995) attributed the summer eutrophic conditions in the lagoon to anthropogenic nutrient loading from effluent discharge, golf courses and septic systems. In contrast, a nutrient study completed by the consulting firm of CH_2MHILL (2000) showed no positive correlation between algae growth and nitrate or phosphorus concentrations in Malibu Creek and Lagoon. The study also reported that natural background nutrient levels in the creek were sufficient to produce saturated growth conditions for the naturally occurring algae. This was supported by observations that algae nuisance conditions had not decreased significantly with the elimination of summer-time effluent discharge.

Table 2-2: Estimates of Annual Loading of Nutrients to the entire Malibu Creek Watershed

	Nitrogen		Phosphorus	
Source	**lbs/yr**	**contribution**	**lbs/yr**	**contribution**
WWTP Discharge	190,989	28.5%	41760	45.3%
Chaparal/Sage Scrub	109,736	16.4%	12771	13.9%
Septic Systems	64,011	9.5%	4912	5.3%
Effluent Irrigation	49,233	7.3%	4275	4.6%
Groundwater	43,123	6.4%	5736	6.2%
Golf Courses (Fertilization)	33,926	5.1%	6021	6.5%
Agricultural/Livestock	33,226	5.0%	2191	2.4%
High/Med. Density Residential	29,399	4.4%	3431	3.7%
Tidal Inflow	24,046	3.6%	1578	1.7%
Commercial/Industrial	19,848	3.0%	2010	2.2%
Imported water	17,944	2.7%	1275	1.4%
Grasslands	10,696	1.6%	1253	1.4%
Low Density Residential	9,057	1.3%	1116	1.2%
Atmospheric deposition	7,470	1.1%	91	0.1%
Sludge Injection	6,140	0.9%	614	0.7%
Vacant Lots	5,489	0.8%	637	0.7%
Rural Residential	5,421	0.8%	616	0.7%
Woodlands	4,796	0.7%	599	0.7%
Sediment Release	4,490	0.7%	621	0.7%
Birds	1,782	0.3%	642	0.7%
Lagoon Drains	103	0.02%	2	0.002%
Total	**670,925**	**100%**	**92,151**	**100%**

Adapted from TetraTech (2001)

Figure 2-2: Historical Monthly Average Phosphate Concentrations in Malibu Creek

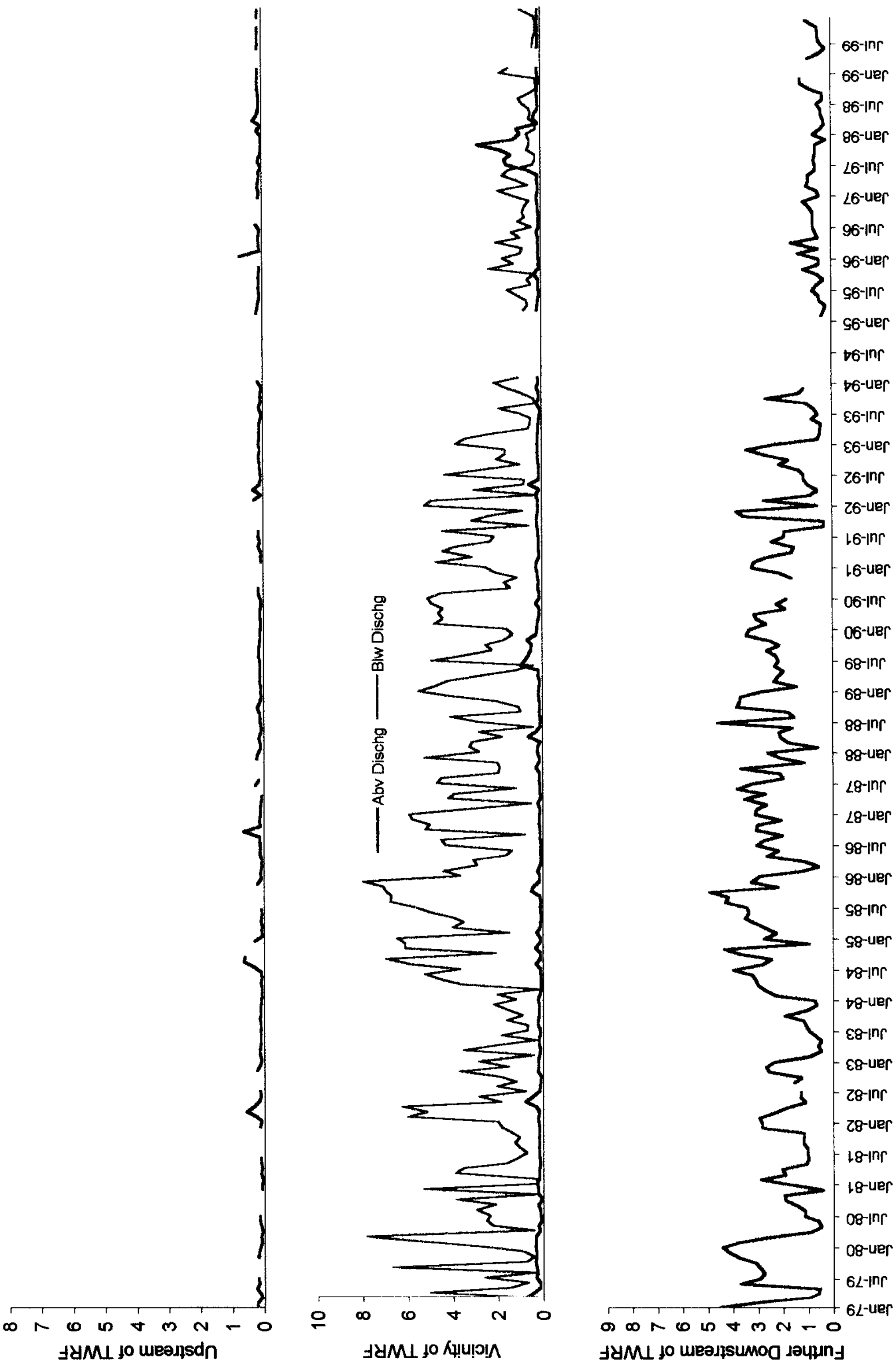


Figure 2-3: Historical Average Monthly Nitrate Conentrations in Malibu Creek

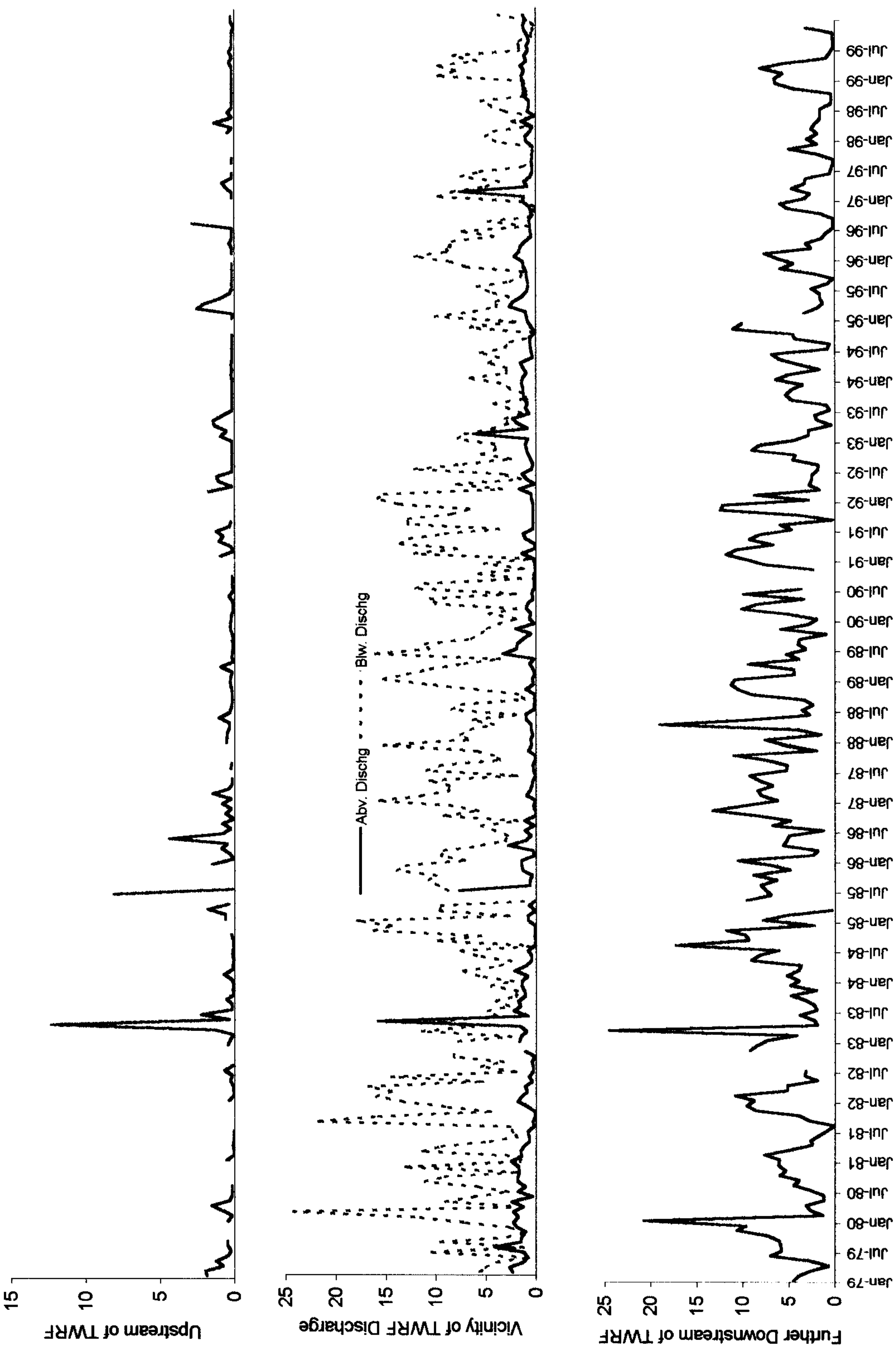


Figure 2-4: Historical Nutrient Levels in Malibu Lagoon

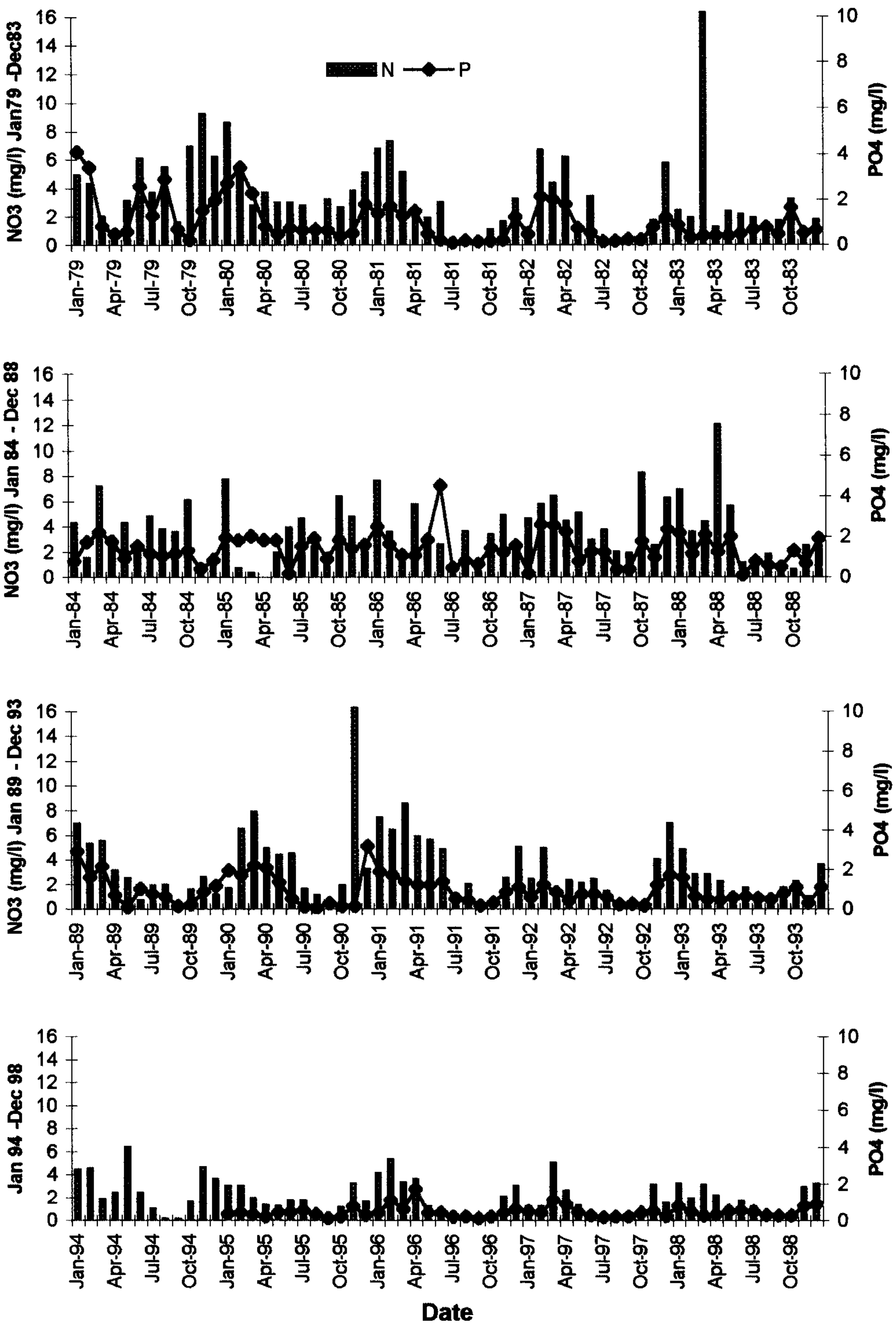


Figure 2-5: Historical Percent Algae Cover in Malibu Creek

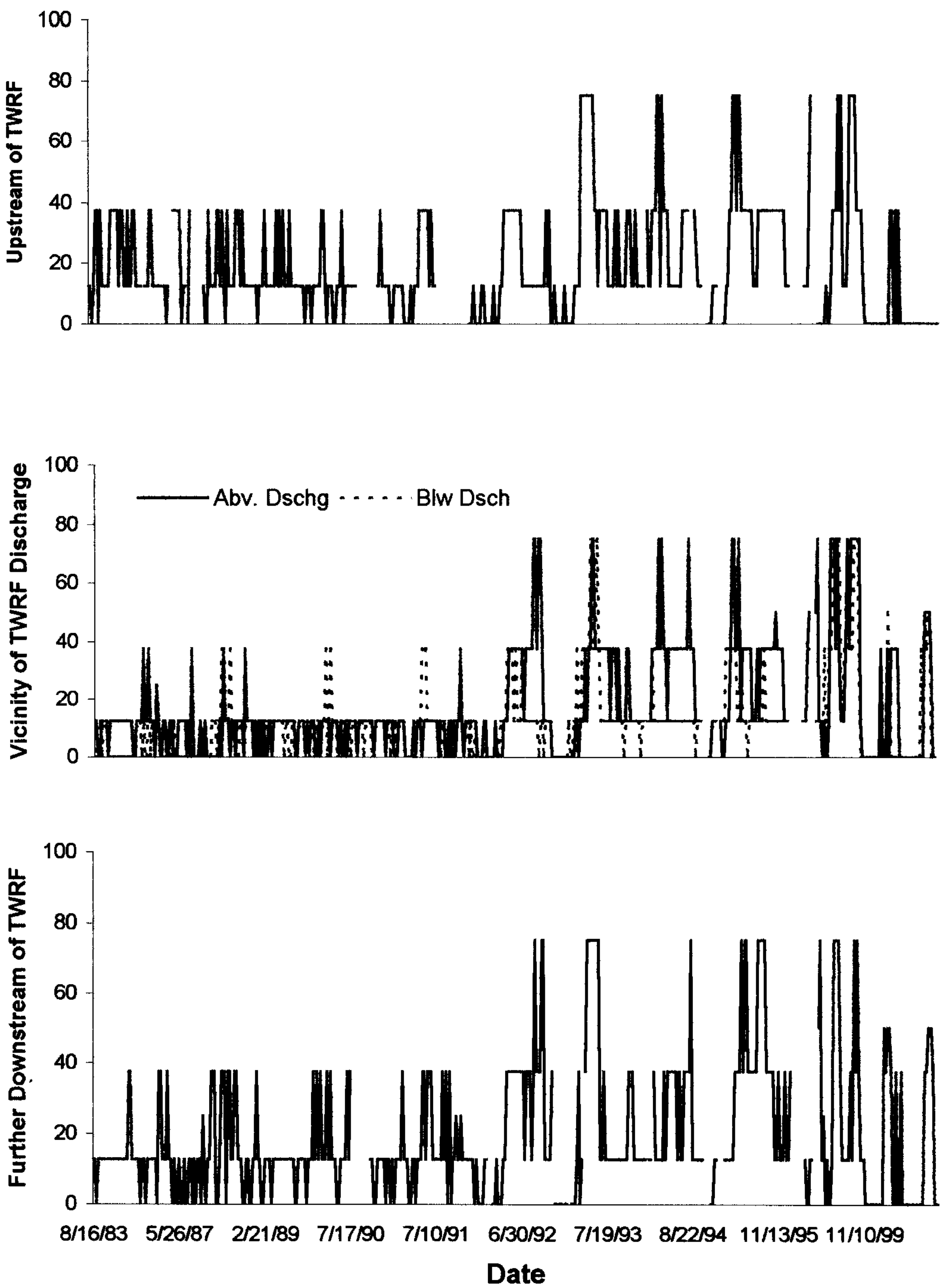


Table 2-3: Types of algae present in Malibu Creek

Specie	Description	Period of Occurrence	Optimal conditions
Cladophora Glomerata	Blanket-weed	winter-spring Summer (throughout creek - in areas with high flows)	High light intensity, High flow rates. Phosphate concentration (1-3ppm)
Tetraspora	Attached algae	Found throughout the creek - absent in winter months	Low flow, high temperature and light intensity, nutrient supply
Phormidium-Nostoc	Blue-green algae	summer	Warm weather and availability of damp sandy areas
Vaucheria	Benthic algae	Summer (in localized quiet waters)	Warm temperatures, Reduced flow rate, decline in chlodophra, and sandy margins
Enteromorpha	Free-floating algae Marine algae adapted to freshwater habitat		Disappeared in winter
Cocconeis and Gomphonema	Planktonic algae	Numbers decrease in summer; occur throughout the creek	
Schizomeres		Fall (localized-limited to area around effluent pipe)	Nutrient-rich waters
Spirogyra Ulothrix		Random occurrence throughout creek	

Compiled from Chapman, 1980

2.1.3 Addressing Water Quality Concerns

The existing body of knowledge indicates that excess algae growth in the watershed is due to other environmental conditions in conjunction with elevated nutrient levels. However, efforts to date have focused on limiting nutrient loading to the creek and lagoon. Historical efforts to protect water quality from wastewater discharges are chronicled in Table 2-4.

Discharger efforts: With the continued expansion of the consumer base for recycled effluent, LVMWD has succeeded in reducing its nutrient load to the creek by discharging less effluent. This program has been in place since 1984 and has reached levels of almost 100% recycle in the summer months. In 1991, LVMWD conducted a Biological Nutrient Reduction (BNR) effort to reduce concentrations of nutrients in their effluent. This resulted in a substantial drop in the nitrate levels in the discharge and subsequent loading to the creek for a number of years. However these losses were soon offset by operational changes made in 1996 (return flows from a new composting facility), which resulted in an increase in effluent nutrient concentrations – though not to pre-BNR levels. From 1998-2000 a study was conducted to investigate removal of nutrients from their effluent via treatment wetlands. In 1999 other means of year-round disposal of their reclaimed water was investigated, with the intent of eliminating all nutrient loads to the creek. Currently, LVMWD is proceeding with plans to construct and operate a treatment wetland for nutrient removal from their effluent, and pollution reduction from Malibu Creek flows.

Regulatory Efforts: In 1996, Malibu Creek and Lagoon were listed as impaired for algae and coliform bacteria on the 303(d) list of the Clean Water Act. This led to the inclusion of nutrient limits in LVMWD's effluent discharge permit in 1997. A six-month (May till October) discharge prohibition was placed on TWRFs effluent in addition to the new limits. In 1999 the discharge prohibition was extended to seven months (April 15, to November 15). The purpose of the prohibition was, in part, to limit the effluent contribution to the algae problem during the summer months when recreational use is at its highest and the algae problem is at its worst.

In the same year, LARWQCB conducted an investigation the contribution of septic systems to nutrient and bacteria loading to the creek and lagoon. The study determined that septic systems where a source of pollution to the groundwater, and subsequently the creek and lagoon. In 2000, waste discharge requirements were issued to owners of commercial and multi-family septic systems requiring them to monitor effluent from their systems, and the groundwater. A total maximum daily load (TMDL) was developed for nutrients and coliform bacteria in 2002. It is expected to reduce nutrient loading by placing stricter limits on septic systems, effluent discharge and irrigation, and fertilization of golf courses within the watershed.

Other Efforts: the Resource Conservation District (RCD) of the Santa Monica Mountains made a significant contribution towards non-point source pollutant reduction by developing a "Stable and Horse Management BMP" manual from 1996 to 1999. This was geared towards educating residents on better handling and disposal of wastes from their livestock. Other efforts include intensive monitoring by environmental groups and public agencies in an effort to fully characterize conditions within the creek and lagoon.

Table 2-4 Efforts geared towards reducing pollutant loading to Malibu Creek and Lagoon.

Date	Wastewater Treatment Plant	Septic Systems
1965	Original Tapia Water Reclamation Facility (TWRF). Effluent disposed of by spray irrigation since discharge was prohibited.	County of Los Angeles began planning for sewers in the Malibu area in response to concerns about septic systems
1970		State Board staff adopted an Area-wide Sewage Plan (ASP) for Malibu after determination that for the most part soil conditions are unfavorable for subsurface disposal and failures occur rather frequently
1972	Installation of aeration basins and sedimentation tanks to improve quality of effluent.	The Regional Board adopted a policy (Resolution No. 72-4) precluding consideration of any applications for waste discharge, unless a developer demonstrated how the discharge would comply with a comprehensive ASP.
1978	Two-year test discharge permit granted with no nutrient limits. Long term monitoring required	
1984	Tertiary filters installed. Recycled water distribution reduces effluent discharge to the creek in summer. No nutrient limits set at permit renewal hearing.	
1985		The County Health Officer declared a Health Hazard. Residents initiated litigation against County over proposed sewer
1991	Designation of Malibu Creek as a drinking water Biological Nutrient Reduction efforts conducted by treatment plant to reduce loading from their discharge	Opposition to sewers and desire for local control led residents to Incorporation of City of Malibu
1993		County settles litigation with City agreeing to cease efforts to install a wastewater collection system
1996	Creek and Lagoon listed as impaired for algae and coliform bacteria	
1997	Nutrient limits placed on effluent discharge: 10mg/l nitrate, Interim limits: 13mg/l nitrate 3mg/l phosphate. 6-month prohibition placed on effluent discharge (May – October)	

Table 2-4 cont'd Efforts geared towards reducing pollutant loading to Malibu Creek and Lagoon.

Date	Wastewater Treatment Plant	Septic Systems
1998	Diversion of summertime effluent to the Los Angeles River. Investigation of the use of treatment wetlands for nutrient reduction in effluent.	Regional board proceeded with a directive requiring Malibu City to undertake a technical investigation of the impact of discharges from septic systems.
1999	Discharge prohibition extended to 7 months (mid April to mid November).	Regional Board issues an Administrative Civil Liability in the amount of $5,700 which was later rescinded upon the City's willingness to undertake the investigation voluntarily
2000	Total Maximum Daily load developed for nutrient and coliform bacteria in the watershed.	Preliminary results showed that there was contamination of waters from septic tank discharges
2002	Approval of waste discharge permit for treatment wetlands. Draft TMDL with more stringent nutrient limits 2.5 mg/l N and 0.4 mg/l P for creek. 0.4mg/l TN and 0.02 mg/l TP for Lagoon.	Draft TMDL places restrictions on discharges from commercial septic systems.

2.2 Use of Wetlands for Wastewater Treatment

Constructed wetlands are defined by USEPA (1993) as "wetlands specifically constructed for the purpose of pollution control and waste management, at locations other than existing wetlands". These wetlands are treatment systems designed to mimic the biological, physical and chemical treatment mechanisms that occur in natural wetlands with the added benefit of improved process control. Like conventional treatment units, they are designed to be compatible with their influent stream and to produce the desired water quality. These systems are not considered waters of the US and are therefore not subject to the regulations that govern use of natural wetlands for treatment purposes (USEPA, 1987).

Treatment wetlands comprise both natural wetlands and constructed ones. However with the trend towards preservation of such systems, the use of natural wetlands for wastewater treatment is not encouraged, so constructing artificial wetlands for this purpose is more common.

2.2.1 Types of Constructed Wetlands

There are two major kinds of wetlands – free water surface (FWS) and subsurface flow (SSF) wetlands (Fig: 2-6).

Free water surface (FWS) wetlands usually consist of large shallow basins, separated from groundwater by a liner or impermeable soil, with a few inches of bottom sediments supporting stands of wetland vegetation - most commonly cattails (*Typha sp.*), bulrushes

(*Scirpus sp.*) and sedges (*Carex sp.*). The wastewater is maintained at a shallow depth and flows over the sediment-plant matrix where it is treated as it comes into contact with the bacterial/microbial population therein. FWS are very similar to natural wetlands and can be used for both restoration and treatment purposes.

Subsurface flow (SSF) wetlands (also known as hydrophonic filter beds and reed beds) are planted media beds (usually soil or gravel) through which the water to be treated flows. The water level is kept a few inches from the bed and treatment occurs as water comes into contact with the microorganisms attached to the plant roots and the media. They range in depth from 30 to 90 cm, with the actual value usually depending on expected root penetration of the bed. Conditions within the bed are largely anaerobic and this facilitates reduction processes such as the conversion of nitrate to gaseous nitrogen.

The advantages of SSF wetlands include greater contact opportunities between water and microorganisms and therefore shorter detention times. Also, the absence of an exposed water surface limits the occurrence of odors and vector nuisances that are associated with FWS wetlands. The flow through such systems is usually horizontal; however SSF systems exist that employ downward flow through the media. Such systems are termed Vertical flow systems (VFS) and they create a more aerobic environment for processes such as ammonia nitrification. Free water surface wetlands are more commonly used in the United States while SSF wetlands are more popular in Europe due to their smaller land requirements.

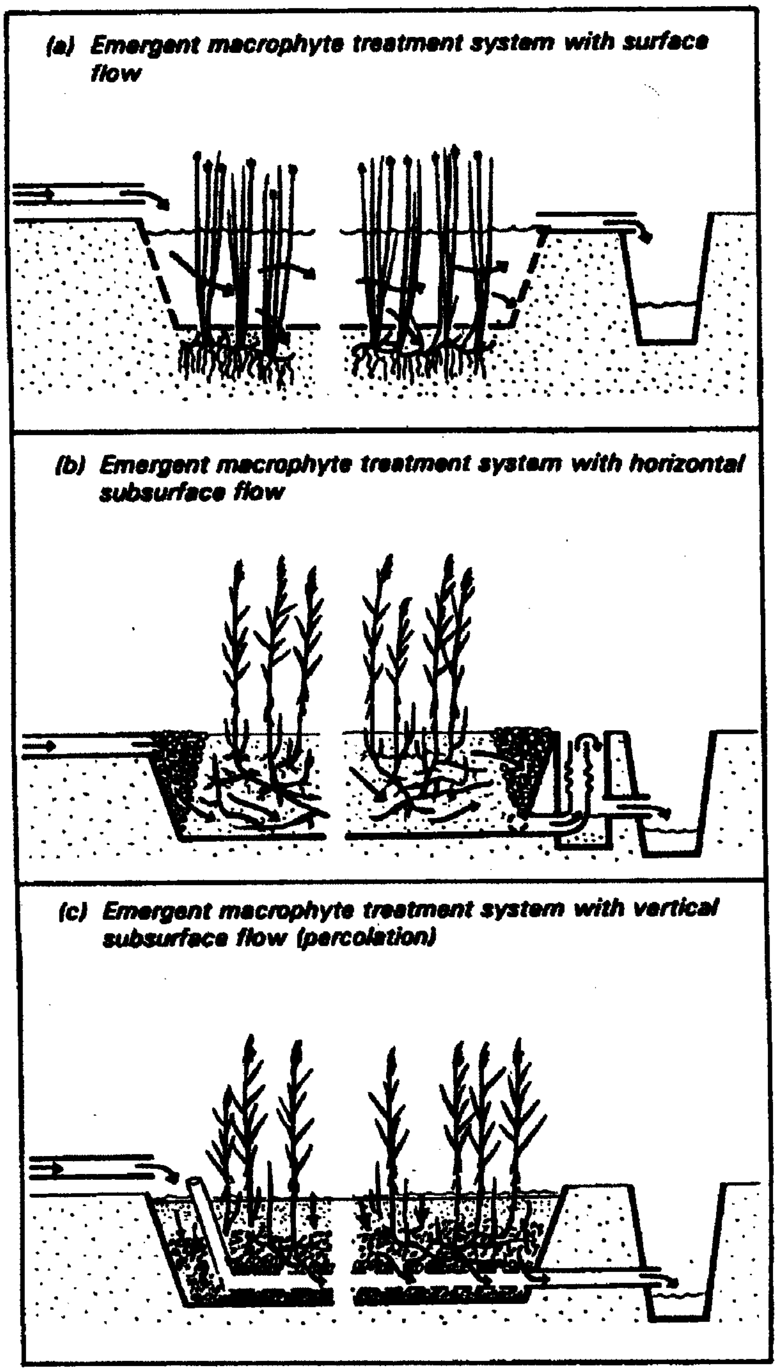


Figure 2-6: Types of Constructed wetlands – source (Brix, 1993)

2.2.2 Function of plants in treatment wetlands

Wetland plants play an important role in these treatment systems by performing a variety of functions. These include:

(i). Increasing surface area available for microbial activity. Hatano et al., 1993 determined microbial populations on the rhizomes of cattails and reeds in gravel bed constructed wetlands to be two to three orders of magnitude greater than those on the gravel media. Availability of attachment surfaces strongly influences denitrification rates within a wetland (Hammer and Knight, 1994).

(ii). Creation of aerobic microsites around roots and rhizomes by oxygen loss through these structures. Oxygen transfer is necessary for increasing aerobic degradation of organic matter within the bed. (Brix, 1994), and nitrification of ammonia.

(iii) Provision of organic carbon from decaying roots and plant litter, and secretion of organic compounds to aid in the decomposition of compounds such as nitrate.

The importance of plants in nutrient removal was demonstrated by Drizo et al., (1997). In their study of laboratory-scale subsurface wetlands, they found that there was a higher nitrate and ammonia removal efficiency in planted beds than in unplanted ones.

2.2.3 Pollutant Removal in Wetlands

Physical, chemical, and biological mechanisms of removal occur within a wetland. These processes include straining and trapping of solids, oxidation-reduction, precipitation, and nitrification-denitrification. Wetlands have been used for years to remove a variety of pollutants, particularly BOD, suspended solids, and coliform bacteria. More recently, the capacity of wetlands to remove nutrients and metals has been demonstrated, and the removal of organic compounds is being studied.

Nitrogen removal in wetlands

Denitrification is the major mechanism for nitrogen loss in wetlands (Oostrom and Russell, 1994; Hammer and Knight, 1994; Brix, 1993; and Cooke, 1994). Hammer and Knight, (1994) conducted a study of fifty-two natural and constructed wetlands that determined nitrogen removal to be varied. The form of nitrogen in the influent is an important factor in removal rates (Hammer and Knight, 1994). For example, the removal of organic nitrogen would require three stages: ammonification, nitrification, and denitrification. Ammonia would require two steps, and nitrate removal only one. In addition to this, conditions have to be favorable for each process to occur. Since SSF beds provide favorable anaerobic conditions, oxidized forms of nitrogen appear to be more easily removed in wetlands than ammonia or dissolved organic nitrogen (Williams and Brown, 1993). Therefore, the rate of denitrification in wetlands is mainly limited by nitrification when the nitrogen to be removed is in the form of ammonia. (Platzer and Netter, 1994; Bowner, 1987; and Geller, 1997).

Denitrification is the biological transformation, by bacteria, of nitrate to nitrogen gas using organic carbon a source of energy. Denitrifying bacteria include *Pseudiomonas, Achromobacter, Aerobacter, Bacillus, Lactobacillus, Proteus and Micrococcus* (Hammer and Knight, 1994; Alexander, 1961; Metcalf and Eddy, 1991). They are facultative and can grow aerobically in the absence of nitrate and anaerobically in its presence (Hammer and Knight, 1994). Factors affecting denitrification include dissolved oxygen, temperature and available carbon.

Plant biomass as a source of organic carbon

While conventional advanced water treatment processes usually use some easily degradable form of carbon such as ethanol, or methanol, for denitrification purposes, most constructed wetlands rely on the BOD loading of the wastewater and/or the organic material within the wetland. In many cases influents to these constructed wetlands are usually secondary or tertiary effluent, and their BOD loading has already been reduced. Primary effluent can be added to the wastewater to provide the carbon source. However, the addition of such material tends to increase the pollutant loading to the system (Gersberg et al., 1984). Therefore, wetlands have relied to a large degree on the organic matter present in the wetland for denitrification. (Oostrom and Russell, 1994; and Barchand, 1996).

Plants on average contain 45-50% (dry weight) carbon and therefore constitute a significant source of carbon in wetland systems. Organic carbon is supplied by material released from dead roots within the media and decomposition of aboveground plant material. The nature of carbon available depends on the stage of decomposition of the plant material. In the first stage, readily decomposable dissolved organic matter is released. This occurs usually within 30 to 60 days of senescence (Mitsch and Gosselink, 1993). Wetzel (1993) estimated that about 30 - 40% of the net primary productivity is released in the form of dissolved organic material within a few hours of senescence. However, the magnitude of the initial loss depends to a large degree on the species of plant and parts being decomposed (Kadlec, 1989). Cattails, for example, only lose about 20% of their weight during this period. In the second stage complex polymers with slow exponential decay rates become available. Table 3-1 lists the half-lives of three common wetland plants (Kadlec, 1989).

Table 2-5: Half-lives of different types of wetland biomass

Type	**Half-life (days)**
Reeds (*Phragmites*)	220 ± 60
Cattails (*Typha*)	400 ± 80
Bulrush (*Scirpus*)	260 ± 50

*Adapted and modified from Kadlec, 1989.

Biodegradation of organic material can be aerobic or anaerobic. Use of plant litter on the bed surface for aerobic respiration results in a loss of carbon from the system. However, fermentation, which is the dominant process in anaerobic environments, produces low molecular weight carbons from more complex compounds. This process occurs when the organic matter itself is the terminal electron acceptor in anaerobic respiration by organisms (Mitsch and Gosselink, 1993). Products include easily degradable acids and alcohols.

Plants also supply organic carbon from compounds exuded by their roots. The quantity of these exudates is estimated at 5-25% of photosynthetically fixed carbon (Brix, 1997). This trait is considered a significant factor in the high denitirification rates observed in rhizome biofilms of wetland plants (Oostrom and Russell, 1994), (Williams et al., 1994). Denitrification in newly constructed wetlands is limited by availability of organic carbon, but as the wetland matures, the build up of detritus usually removes this limitation (Davidsson and Leonardson, 1996).

SSF wetlands are believed to be limited, compared to FWS systems, when relying on carbon as a source for organic matter since the litter falls atop the bed leaving the deeper sections starved of organic matter. Williams et al., 1994 showed that the greatest potential for denitrification was at the top of the bed where the availability of carbon was greatest. With the passage of time, build up of organic matter occurs and SSF beds function similar to FWS systems with respect to internal organic carbon.

Phosphate removal in wetlands

The major removal mechanisms for phosphorus in wetlands include plant uptake, and adsorption and subsequent precipitation (WERF, 2000; White et al., 1994). Particulate phosphorus is removed by sedimentation within the wetland. Other factors such as input rates and detention time affect the (Richardson and Craft, 1993). Phosphorus removal in SSF wetlands is limited since the media does not usually have large adsorptive capacities. In some wetlands, soil is used in place of gravel to increase phosphorus removal; this leads to larger wetlands since the hydraulic capacity reduces with decrease in grain size (USEPA, 1993). Litter accumulated within a wetland could provide additional sorption sites. However, in SSF wetlands the litter is restricted to the top of the bed and barely comes in contact with the water (Vymazal, 1999). The removal attributed to plant uptake is usually offset by the release of phosphorus that occurs when the plants die and decay.

Pathogen removal in wetlands

The main mechanisms for coliform removal in subsurface beds are sedimentation, predation, filtration, adsorption, and natural die-off. (Green et al., (1997), Gersberg et. al., (1989), and Ottova, (1997). (Bavor et al., 1989) describe the removal of coliforms in constructed wetlands as a first-order relationship that is a function of temperature, detention time and influent concentration. (Butler et al., 1993) describes it as a function of wetland length, which is likely to be as a result of increased opportunity for sedimentation, filtration and interaction with predators and toxic chemicals. Anaerobic conditions prolong fecal coliform survival in natural waters Ottova et al., (1997).
In addition, the roots of certain aquatic plants are known to excrete certain compounds that can kill coliform and salmonella bacteria (Ottova et al., 1997).

Gravel beds have proven to be efficient in pathogen removal. Ottova et al., (1997) studied the coliform removal efficiency of five different SSF wetlands and found approximately 99% removal in four of them. The lower performance (95%) of the fifth bed was attributed to poor root penetration of the vegetation, which may have led to anaerobic conditions. Other research has shown similar reductions of 99 % or higher (Green et al, (1997), Rivera et al., (1997) and TVA, (1990).

2.2.4 Constructed Wetland Use in California

Constructed wetlands have been used in California to solve water quality problems from as early as 1977. Table 2-6 is a compilation of some constructed treatment wetlands in this region. While the majority of them are surface flow wetlands, infiltration systems are used when discharge to receiving water bodies is not an option. Most of these treatment wetlands are affiliated with wastewater treatment plants and were created as a means of meeting more stringent waste discharge requirements. In addition to their treatment objectives, these wetlands provide the added ecological benefit of newly created habitats that support a diversity of species.

Table 2-6: Compilation of some constructed treatment wetlands in California

Location/ Type	Size/ Capacity	Life	Motive	Benefits	Previous discharge point
Arcata Marsh (CA) FWS treatment and enhancement marshes	38.5 acres/ 2.3 mgd	1986 - Present	Alternative to costly and complicated treatment facility	Enhancement of biological productivity of wetland environment in area of discharge Rearing of salmon and steelhead. Recreation and education opportunities. Habitat or rest stop for birds	Humboldt Bay
Marine Co. (CA) FWS part of a 385 acre system	20 acres/ 2.9 mgd	1984 - Present	alleviate burden of summer discharge prohibition	Reclamation of up to 350 million gallons of effluent per year for landscape irrigation. Creation of a biologically diverse wetland ecosystem.	San Pablo Bay
Hayward Marsh (CA) FWS	172 acres/ 12.2 mgd	1988 - Present	restoration of marshes and enhancement of the Hayward (CA) shoreline	Treatment of secondary effluent prior to discharge to the bay.	deep waters of San Francisco Bay
Martinez (CA) FWS	20 acres/ 1.3 mgd	1977 - Present	improve quality of effluent thereby preventing annexation to a large regional plant	Creation of diverse aquatic habitats that have attracted numerous species of birds and other animals. Specifically provide protected nesting habitat for waterfowl.	Peyton Slough (tributary to Carquinez Straits)
City of Santa Rosa (CA) FWS demonstration system	4.05 ha. / 2 mgd	1989 - Present	Maximizing reclaimed water re-use by wetland creation and restoration.	Environmental enhancement.	Russian River
Guistine (CA) FWS	23 acres/ 1mgd	1988 - Present	improve quality of oxidation pond effluent		tributary of San Joaquin River

Table 2-6 Cont'd: Compilation of some constructed treatment wetlands

Location/Type	Size/ Capacity	Life	Motive	Benefits	Previous discharge point
Hemet/San Jacinto (CA) FWS	50 acres/ 1mgd	1994 Present	water quality improvement Creation of wildlife habitat	potential to supplement diminishing groundwater resources	reclaimed water distribution system
Riverside (CA) FWS	~48 acres*/ 10mgd	1995 - Present	achieve compliance with more stringent water quality requirements,	Restoration of high quality riparian habitat.	Santa Ana River
Graton (CA) SSI	15 acres/ 0.14mgd**	1999 - Present	alternative to costly tertiary treatment. prevent linkage to a central sewage treatment facility	Keeps control sewer system local. Prevents rapid development. Replenishes local aquifers.	Russian River
Richmond (CA) FWS	3.6 ha/ 0.42 mgd	1989 – n.a.	treatment of industrial wastewater		
Santee (CA) SSF demonstration	1mgd	1983 - 1985	Nitrogen reduction in secondary effluent		

*Estimated from available data; **projected capacity, n.a = not available

2.3 References

Alexander, M. 1961. Introduction to Soil Microbiology. John Wiley & Sons Inc. New York.

Ambrose, R. F., Suffet, I. H., Que Hee, S. S. 1995. Enhanced Environmental Monitoring Program at Malibu Lagoon and Malibu Creek.

Barchand, P.A M. 1996. Effects of managing Vegetative species, Hydraulic Residence Time, Wetland Age and Water Depth on Removing Nitrate from Nitrified Wastewater in Constructed Wetland Macrocosms in the Prado Basin, Riverside County, California. Dissertation - University of California Berkeley.

Bavor, H. J., Roser, D.J., Fisher, P. J., Smalls, I.C. 1989. Performance of Solid-Matrix Wetland Systems Viewed as Fixed-Film Bioreactors. In Constructed Wetlands for Wastewater Treatment, D. A. Hammer (ed), Lewis Publishers, Chelsea, Michigan. 646-656.

Bowner, K. H. 1987. Nutrient Removal from Effluents by an Artificial Wetland: Influence of Rhizosphere Aeration and Preferential Flow Studied Using Bromide and Dye Tracers. Wat. Res. (21) 3: 591-599.

Brix, H. 1993. Wastewater Treatment in Constructed Wetlands: System Design, Removal Processes, and Treatment Performance. In Constructed Wetlands for Water Quality Improvement, G. A. Moshiri (ed), Lewis Publishers, Ann Arbor. 9-22.

Brix, H. 1994. Functions of Macrophytes in Constructed Wetlands. Wat. Sci. Tech. 29(4): 71-78.

Brix, H. 1997. Do Macrophytes Play a Role in Constructed Treatment Wetlands? Wat. Sci. Tech. 35(5): 11-17.

Butler, J. E., Ford, M. G., May , F. E., Ashworth, R. F., Williams, J. B., Dewedar, M. E., Baghat, M. M. M. 1993. Gravel Bed Hydroponic Sewage Treatment: Performance and Potential. In Constructed Wetlands for Water Quality Improvement, G. A. Moshiri (ed), Lewis Publishers, Ann Arbor. 237-247.

California Regional Water Quality Control Board Los Angeles Region (CRWQCB-LA). 1977. Malibu Creek and Lagoon: Water Quality Assessment and Updating of its Designated Uses and Water Quality Objectives. (Draft).

California Regional Water Quality Control Board Los Angeles Region (CRWQCB-LA). 1990. Malibu Lagoon Coliform and Nutrient Study. Working Draft.

CH2MHILL (2000). Evaluation of Nutrient Standards for Malibu Creek and Malibu Lagoon. Prepared for the Las Virgenes Municipal Water District and the Triunfo Sanitation District.

Chapman, D. 1980. Algae of the Malibu Creek and Changes in the Algae Populations in the Creek: May 1978 - July 1979. In Malibu Creek Study 1978-1979. Prepared by James Montgomery Consulting Engineers, Inc.

Cooke, J.G. (1994). Nutrient Transformations in a Natural Wetland Receiving Sewage Effluent and the Implications for Waste Treatment. Wat. Sci. Tech. 29(4): 209-217.

Davidsson, T. E., Leonardson, L. G. 1996. Effects of Organic Carbon and Nitrate Additions on Denitrification in two Artificially Flooded Soils. Ecol. Eng. 7 : 139-149.

Drizo, A., Frost, C.A., Smith, K.A., Grace, J. 1997. Phosphate and Ammonium Removal by Constructed Wetlands with Horizontal Subsurface Flow, Using Shale as a Substrate. Wat. Sci. Tech. 35(5): 95-102.

Geller, G. 1997. Horizontal Subsurface Flow Systems in the German Speaking Countries: Summary of Long Term Scientific and Practical Experiences; Recommendations. Wat. Sci. Tech. 35(5): 157-166.

Gersberg, R. m., Elkins, B. V., Goldman, C. R. 1984. Wastewater Treatment by Artificial Wetlands. Wat. Sci. Tech 17(4/5): 443-450.

Gersberg, R. M.; R. A. Gearheart, and Mike Ives. 1989. Pathogen removal in constructed wetlands. In Constructed Wetlands for Wastewater Treatment; Municipal, Industrial and Agricultural. D.A. Hammer (Ed). Lewis Publishers, Chelsea, Michigan. 431-446.

Green, M. B.; P. Griffin, J. K. Seabridge, and D. Dhobie. 1997. Removal of bacteria in subsurface flow wetlands. Wat. Sci. Tech. 35(5):109-125.

Hammer, D. A., Knight, R. L. 1994. Designing Constructed Wetlands for Nitrogen Removal. Wat. Sc. Tech. 29(4): 15-27.

Hatano, K., Trettin, C. C., House, C. H., Wollum, A. G. 1993. Microbial Populations and Decomposition Activity in Three Subsurface Constructed Wetlands. In Constructed Wetlands for Water Quality Improvement, G. A. Moshiri (ed), Lewis Publishers, Ann Arbor. McGraw-Hill, Inc. New York. 541-547.

Kadlec, R. H. 1989. Dynamics of Inorganic and Organic Materials in Wetland Ecosystems: Decomposition in Wastewater Wetlands. In Constructed Wetlands for Wastewater Treatment; Municipal, Industrial and Agricultural. D.A. Hammer (Ed). Lewis Publishers, Chelsea, Michigan. 459-468.

Los Angeles County Department of Public Works (LACDPW), URS Grenier Woodward Clyde, and Southern California Coastal Water Research Project. 1999. Los Angeles County 1998-99 Storwater Monitoring Report.

Metcalf and Eddy. 1991. Wastewater Engineering: Treatment Disposal and Reuse. McGraw Hill, New York.

Mistsch, W. J., Gosselink, J. G. 1993. Wetlands. Van Norstrand Reinhold. New York.

Oostrom, A. J. and Russell, A. J. 1994. Denitrification in Constructed Wastewater Wetlands Receiving High Concentrations of Nitrate. Wat. Sci. Tech. 29(4):7-14.

Ottova, V., Balcarova, J., Vymazal, J. 1997. Microbial Characteristics of Constructed Wetlands. Wat. Sci. Tech. 35(5): 117-2-123.

Platzer, C., Netter, R. 1994. Factors Affecting Nitrogen Removal in Horizontal Flow Reed Beds. Wat. Sci. Tech. 29(4): 319-324

Richardson, C. J. and Craft, C.B. 1993. Effective Phosphorus Retention in Wetlands: Fact or Fiction? In Constructed Wetlands for Water Quality Improvement, G. A. Moshiri (ed), Lewis Publishers, Ann Arbor. McGraw-Hill, Inc. New York. 271-282.

Rivera, F., Warren, A., Curds, C. R. 1997. The Application of the Root Zone Method for the Treatment and Re-use of High-Strength Abattoir Waste in Mexico, Wat. Sci. Tech. 35(5): 271-278.

Santa Monica Bay Restoration Project. 1993. Santa Monica Bay Characterization Study, 1993.

Tennessee Valley Authority (TVA). 1990. Design and Performance of the Constructed Wetland Wastewater Treatment System at Phillips High School, Bear Creek, Alabama. TVA/WR/WQ--90/5.

Tetra Tech, Inc. 2001. Nutrient and Coliform Modeling for the Malibu Creek Watershed TMDL Studies. Prepared for U.S. Environmental Protection Agency (USEPA) and the Los Angeles Regional Quality Control Board.

United States Environmental Protection Agency (USEPA). 1987. Report on the Use of Wetlands for Municipal Wastewater Treatment and Disposal. EPA 430/09-88-005.

United States Environmental Protection Agency (USEPA). 1993. Subsurface flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment, EPA 832-R-93-008.

United States Department of Agriculture (USDA), and Natural Resources Conservation Service (NRCS). 1995. Malibu Creek Water Resources Plan.

United States Environmental Protection Agency (USEPA). 1993. Subsurface flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment. EPA 832-R-93-008.

Vymazal, J. 1999. Removal of phosphorus in: Constructed Wetlands with Horizontal Sub-surface Flow in the Czech Republic. In Nutrient Cycling and Retention in Natural and Constructed Wetlands. 73-83

Water Environment Research Foundation (WERF). 2000. Constructed Wetlands Treatment of High Nitrogen Leachate. Project 94-IRM-U.

White, G. C., Smalls, J. C., Bel, P. A. (1994). Carcoar Wetland – A Wetland System for River Nutrient Removal. Wat. Sci. Tech. 29(4): 169 –176.

Williams, J. B., May, E., Ford, M. G., Butler, J. E. 1994. Nitrogen Transformations in Gravel Bed Hydrophonic Beds used as a Tertiary Treatment Stage for Sewage Effluents. Wat. Sci. Tech. 29(4):29-36.

Wetzel, R. G. 1993. Constructed Wetlands: Scientific Foundations are Critical. In Constructed Wetlands for Water Quality Improvement, G. A. Moshiri (ed), Lewis Publishers, Ann Arbor. McGraw-Hill, Inc. New York. 3-7.

CHAPTER 3: PILOT-SCALE WETLAND FOR THE TREATMENT OF LOW-CARBON NITRIFIED TERTIARY EFFLUENT

Abstract

A pilot scale study was conducted at the Las Virgenes Municipal Water District, Calabasas California from spring 1999 to fall of 2000, to investigate the potential of constructed wetlands to remove nitrogen from nitrified tertiary effluent. The goal of this project was to demonstrate the feasibility of converting existing percolation ponds to treatment wetlands. While the majority of earlier work on nitrogen removal in wetlands has addressed ammonia reduction, this study is concerned with the removal of nitrate without the use of supplemental carbon. Two subsurface flow units, each with 3.25m^2 *surface area and bed depth of 1m, were operated at hydraulic residence times (HRT) of 18- and 48-hrs. Both units were planted with locally obtained cattails (Typha sp.).*

This report presents the results obtained during the final 10-months of the study. Nitrate removal of the 48hr HRT bed averaged 50.6% and was consistently better than the 18hr HRT bed (20.8%). The mass of nitrogen removed by both systems was similar (3.0 and 3.2 g/d for the 48 and 18 HRT beds, respectively). There was a modest variation of performance with changes in season, and a weak correlation ($R^2 = 0.4$*) was observed between water temperature and nitrate removal. While the influent nitrate concentration varied considerably (3.3 – 21.3 mg/l) the removal rate remained relatively constant. At the end of the experiment, carbon (sodium acetate) was added to promote denitrification. The 2-week lag time observed in the response of the system to the acetate suggests that a denitrifying community of organisms was either absent, or not very large. This implies that plant uptake could be the predominant nitrate removal mechanism. Phosphate removal was also monitored and was observed to be 4% (18 hr HRT) and 14.3% (48 hr HRT). With a nitrate removal rate of 1 g/m*^{2}*d, the available land area could remove up to 2.6kg/d of nitrogen without supplemental carbon.*

3.1 Introduction:

The detrimental effects of excess nitrogen and phosphorus compounds entering surface waters have become more evident with the increase in eutrophication and its associated problems. Wastewater treatment plant operators are seeking cost effective means of nitrogen removal. Since new policies and discharge regulations are limiting nutrient loading to receiving water bodies. Constructed wetlands are currently viewed as an effective and low-cost option due to their simplicity of design construction and operation. Until recently, an important objective of many constructed wetlands for nutrient removal in the US has been ammonia and total nitrogen reduction (Wood, 1995; USEPA, 1993). However, an increasing number of studies are focusing on nitrate reduction as their goal (Gersberg et al., 1984a; Barchand, 1996; WERF, 2000).

The purpose of this study was to determine the extent of nitrate reduction possible in a subsurface flow wetland treating nitrified tertiary effluent-in the absence of supplemental organic carbon. Performance at two hydraulic retention times was compared and we investigated possible mechanisms nitrate reduction. Changes in phosphorus concentrations were also monitored.

3.2 Background

Wetlands may remove ammonia or nitrate from treated wastewaters by denitrification, plant uptake, dissimilatory nitrate reduction to ammonia (DNRA), and ammonia volatilization. DNRA occurs under conditions of high carbon loading and low nitrate concentration, which is rarely observed in treated wastewater. Volatilization of ammonia is generally insignificant at the neutral pH of most municipal wastewaters. Table 3-1 shows the results of previous studies conducted to ascertain the contribution of each removal mechanism in constructed wetlands. There is a general consensus among these results and other literature (Ingersoll and Baker, 1998; Hammer and Knight 1994; Brix, 1993) that denitrification is the dominant mechanism.

Denitrification

Denitrification is the process of nitrate reduction to nitrogen gas in the absence of oxygen; nitrate is used as the terminal electron acceptor in respiration. Organic carbon compounds serve as hydrogen donors as well as an energy source for growth of the denitrifying bacteria (Nichols, 1983). The organic carbon requirement for denitrification varies between 2.5-10 g BOD per gram of nitrate depending on the carbon source (Reed et. al., 1995; Aesoy et al., 1998).

The nature of the carbon determines efficiency of the reduction. Compounds that are easily biodegraded such as simple sugars, alcohols, or organic acids are oxidized quickly and promote high denitrification rates (Mitsch and Gosselink, 1993; Alexander, 1961). Acetate and methanol are effective artificial carbon sources in wastewater treatment when applied at weight ratios of 1.46 (Constantin and Fick, 1997) and 1.08 (Metcalf and Eddy, 1991), respectively. Carbon contained in raw sewage has a moderate reaction rate while reaction with carbons in treated wastewater and plant matter (Mitsch and Gosselink, 1993) is slowest.

Table 1: Contribution of different mechanisms to nitrogen removal in constructed wetlands.

Author	Method	**DN**	PU	**DNRA**	Comments
Gersberg et al, (1983)	Estimation from pilot study results	84 - 88%	12 - 16%		SSF
Van Ostrom & Russel (1994)	^{15}N-labelling	95%	Inhibited by darkness	<5%	FWS
Cooke, (1994)	Isotope (^{15}N) Dilution	60 - 70%	5-10%	25 - 35%	FWS
Reed et al, (1995)	Harvesting trials	> 90%	<10%		FWS
Raman et al, (1999)*	Estimations from literature values	44%	49%		FWS 8% volatilization

DN = denitrification, DNRA = dissimilatory nitrate reduction to ammonia. PU = plant uptake
Wetland types: FWS = Free water surface; SSF = Subsurface flow
* Influent nitrogen in the form of ammonia, all others were as nitrate

Dissolved oxygen interferes with denitrification by suppressing the enzyme system necessary for the process to occur (Metcalf and Eddy, 1991). Diffusion of atmospheric oxygen is limited in a SSF wetland; therefore, the roots and rhizomes of the wetland plants are the main source of oxygen (Reed et al, 1995). The rate of oxygen production of these plants is estimated at 4.6 – 7.2 g/m^2/d (Amstrong et al., 1990; Brix and Schierup, 1990; Gersberg et al., 1986), most of which is consumed by the respiratory demands of the plants (Brix and Schierup, 1990). Even with oxygen present in the bulk water, the biofilms on the rhizomes and media are able to maintain anoxic internal regions where denitrification takes place (Williams et al, 1994).

The temperature for denitrification ranges from 25 to 65°C (Alexander, 1961; Hammer and Knight, 1994) with rates dropping off above and below this range. Within this range, the optimal temperature varies among wetlands (Wood et al., 1999; Ingersoll and Baker, 1998). Pfenning and McMahon (1996) observed that potential denitrification rates at 4°C were about 77% lower than at 22 °C in riverbed sediments. Stengel and Schultz-Hock (1989), however, demonstrated that the impact of low temperatures on denitrification rates in a wetland could be mitigated by the addition of supplemental carbon.

Plant Uptake

Nitrate assimilation by plants can be another significant removal mechanism in constructed wetlands. This process varies seasonally with high uptake rates in the spring and early summer. Senescence in the fall and winter months returns the nutrients to the wetland (Gersberg et al., 1983; Howard-Williams and Downes, 1993; Kadlec, 1995). Nitrogen uptake rates reported in the literature range from 0.16 to 0.71 g/m^2-d (van Oostrom and Russell, 1994; Wood, 1995; and Raman et al, 1997).

Experimental Studies

Table 3-2 is a compilation of studies of nitrate removal in subsurface flow constructed wetlands and contains thirteen references to previous bench- to full-scale research. The feed waters for these studies were mostly treated municipal effluent, but also included urban runoff (Reuters et al; 1989) and treated landfill leachate (WERF, 2000). Nitrate reductions of 12 to 94% were achieved for influent nitrate concentrations of 114μg/l to 105 mg/l. In some cases the authors reported total organic carbon concentration (TOC), biochemical oxygen demand (BOD) and chemical oxygen demand (COD). To create a basis for comparison, TOC was estimated by multiplying the BOD by 2.5. The TOC results will be discussed later.

Green and Verhoeven (1999) studied the effect of different hydraulic retention times (6, 12, 24, 48, and 120-hrs) on nitrate removal. He showed improvement with increasing residence time to 48 hrs, after which it was no longer beneficial. Gersberg (1983, 1984) reported on the ability of different supplemental carbon sources to improve nitrate removal rates in SSF wetlands treating nitrified secondary effluent. Additions of methanol, plant mulch and primary effluent improved treatment performance to varying degrees. This work was concerned with determining the extent of nitrate removal possible in a sub-surface flow wetland with nitrified tertiary effluent as influent, and the governing mechanism of nitrate reduction under these conditions.

Table 3-2: Studies of Denitrification in Subsurface flow Wetlands

Reference/ Location	Dimensions (Area x Depth)	Media/ Plants	Type of Waste	Retention Time (Days)	Influent Nitrogen (mg/l)	Carbon Source (mg/l)	Removal Efficiency	Comments
Pilot Scale								
Gersberg 1983 Santee, CA	$65 m^2$ x 0.76m	Scirpus and Typha sp.	nitrified 2° effluent	0.36	TN =18 Nitrate = 17.3 NH3 = 0.1	BOD: 2.0 Plant mulch 170gC/m3	TN = 27% TN = 87% TIN = 91%	
Vymazal, 1990 Prague, Czech.	Pilot $0.5 m^2$ x 1m	0.5-8mm gravel 0.5-3mm limestone Phragmites sp	diluted chicken waste	13	TON: 24.4-25.7 NH_3N: 155-270	BOD: 2340-3220	TON: >83% NH_3N:>91%	
Zhu and Sikora, (1995) Alabama	$0.14 m^2$x0.5m	5-10mm/ varied	synthetic solutions	~5 ~2.1	TON: 48	organic C: 12-30 acetate: 118mgC/l	100% 100%	short-term batch experiments 9-days
Laber et al., 1997 Austria	Small units for 2 farmhouses (VF) $5 m^2$ x 0.6m	sand + gravel 1 – 4mm/ Phragmites sp.	effluent from nitrifying wetland	n.a	Nitrate: 36 36 45	methanol	Nitrate: 89% 58% 72%	intermittent feed continuous single batch
Green & Verhoeven (1999) New Zealand	$2.53 m^2$ x 0.6m	washed gravel/ reeds	2° sewage effluent	0.25 - 5	TON: 12-18.5 NH_3N: 5.1-8.7	BOD: 11-19	TON: 29-64% NH_3N: 13.5-96%	very short term study
WERF, 2000 New Hanover, NC	$67.5 m^2$ x 0.6m	river rock/ varied	treated landfill leachate	14	TN: 125 TON: 106 NH_3N: 12	BOD: 24±15	TN: 12-19% TON: 13-24% NH_3N: n.a	actual HRT reduced by short circuiting
Full Scale								
Gersberg et. al (1984a) Santee, CA	$824 m^2$ x 0.76m & $742 m^2$ x 0.6m	Scirpus and Typha sp.	1: 2 mix of 1° and 2° effluent	0.33	TN =21.3 TIN = 20 NH3 = 7 Nitrate = 12.3	BOD: 51.5	TN = 80% TIN = 80%	

Table 3-2 cont'd: Studies of Denitrification in Subsurface flow Wetlands-Cont'd

Reference/ Location	Dimensions Area x Depth	Media/ Plants	Type of Waste	Retention Time (Days)	Influent Nitrogen	BOD/Carbon (mg/l)	Removal Efficiency	Comments
Gersberg et. al (1984b) Santee, CA	824 m^2 x 0.76m & 742 m^2 x 0.6m	Scirpus and Typha sp.	nitrified 2° effluent	0.3 – 0.7	TN =19 TON = 18.3 NH3 = 0.1`	BOD: 3.3	TN: 9-19%	A series of experiments to determine the effect of different carbon sources on nitrate removal.
				0.24 – 0.3		Methanol MeOH:NO3 = 4.5 C:NO3 = 1.7	TN 94% TIN 97%	
				0.5 – 0.7		Plant mulch Plant = 23g/m3	TN 89% ± 7 TIN 95% ± 6	
				0.3 – 0.4		Plant = 13g/m3	TN 70% ±21% TIN 65%± 18	
Reuter et al; 1989 Lake Tahoe, CA	660 m^2	19 mm gravel/ cattails	urban run-off	4	TON: 0.114	n.a	TON:85-87%	
TVA, 1990 Bear Creek, AL	2033 m^2 x 0.3m	12.7 mm river gravel/ Typha sp Carex sp.	2° sewage effluent		TON: 26 Org-N: 12 NH_3N: 10.7	BOD: 13	TON: 75% Org-N: 95% NH_3N: 83%	
Williams et al; 1995 United Kingdom	140 m^2 x 0.2m	flint gravel 10-20 mm/ Phragmites sp.	nitrified 2° sewage effluent	~0.25	TON: 16.7 NH_3N: 4.1	BOD:21.5	TON: 53% NH_3N: 93%	Intermittent flow 12hrs/day
Coombes and Collett, 1995 Devon, UK	500 m^2 x 0.4-0.5m	4 – 15 mm basalt & limestone aggregate	2° sewage effluent	1.2 - 3.6 *	TON: 14.7-19.6 NH_3N: 0.6-4.3	BOD: 9.6-13.9	TON: 22% NH_3N: 15%	
Green & Verhoeven (1999) New Zealand	825m^2	n.a./ Phragmites australis	2° sewage effluent	1	TON: 18.8 TKN: 9.8	BOD: 24.6	TON: 76% TKN: 90%	based on a 5-day and 3-day survey.

*Calculated from available data; VF = vertical flow; n.a = not available; TON = Total Oxidized Nitrogen (NO3 + NO2); TN = Total Nitrogen; TKN = Total Kjeldahl Nitrogen
1° = primary, 2° = secondary

3.3 Materials, Methods & Analysis

Materials

Two wetland cells (Figure 3-1) were constructed from plywood with dimensions of 3m (10ft) L x 1.1m (3.6ft) W x 1.2m (4ft) H each. Flow was controlled with 2.5-cm ball valves; a standpipe was used to control liquid level. Influent and effluent were distributed with perforated piping wrapped in nylon mesh that traversed the entire bed width. Sampling ports were located at depths of 0 cm, 20 cm (8"), 41cm (16"), 61cm (24"), and 81cm (32") from top of the gravel bed. The interior of the boxes was lined with plastic, and Pea-sized gravel (13 mm) was laid to a height of 1 m. The gravel was inoculated with organisms from the return sludge of a nitrifying activated sludge treatment plant. Porosity of the gravel was 40%. Dormant Cattails (*Typha sp.*) were transplanted from a nearby pond at a density of 30 clumps per cell. The cells were located at the Tapia Wastewater Reclamation Facility (TWRF) in Calabasas, California.

Source of Influent

Nitrified tertiary effluent obtained from the plant fed the pilot cells. The Tapia facility is a fully nitrifying 60,567-m3/sec (16 mgd) activated sludge plant. In addition to the nitrogen load in the influent wastewater, centrate from de-watering of the primary- and some part of the secondary- sludge is returned to the head-works for treatment. This centrate, which has ammonia concentrations from 500-800mg/l and is pumped at the rate of 75,000 gallons per day, is a significant source of nitrogen.

Figure1a: Pilot cells constructed with plywood and lined with plastic, showing ten depth-profile sampling ports. Inlet and outlet portions are shown below

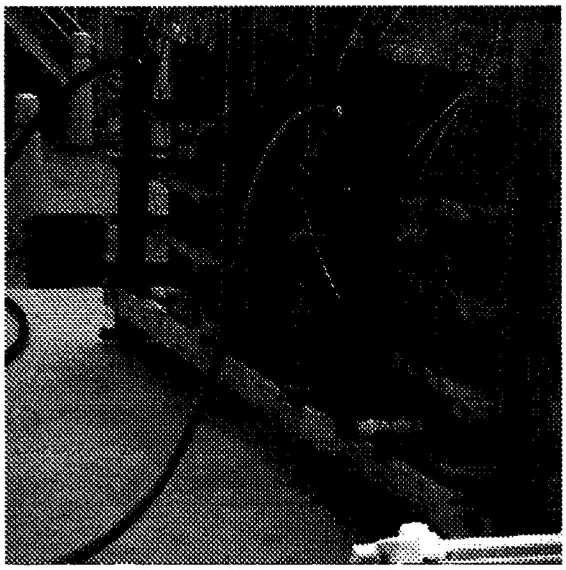


Figure 1b: Standpipe at effluent end of pilot cells to keep water level constant.

Figure 1c:Influent line for pilot cells with inductor for chemical addition.

Treatment at the Tapia Facility includes a single-stage nitrification process. In the summer months there is greater operational flexibility and the process is modified to achieve partial denitrification. These modifications cannot be sustained during the wetter months when higher flows are received and the plant operates at full capacity. The continuous process modifications created variations in feed nitrate concentrations to the cells (Figure 3-2a). Table 3-3 shows the composition of the plant's tertiary effluent for the period of study.

Sampling

The wetland cells began operation in January 1999, once all the rhizomes were planted. Flow meters were initially used to control flow rate but failed due to fouling and scale accumulation. A graduated bucket and stopwatch were later used to control flow. Standpipes at the effluent ports maintained water depth within the beds.

The two cells were operated at different hydraulic retention times (HRT) 18- and 48-hours, by adjusting the feed rate for each cell. This resulted in different nutrient loading rates, as the nitrate concentrations for both influents were the same. These HRTs were selected based on results obtained from preliminary operation of the cells during March to September 1999 (see Appendix A). Beginning in October 1999, weekly composite samples of the influent and effluent were taken to determine nitrate removal. The sampling frequency increased to four 24-hr composite samples per week in January 2000. Phosphate and total organic carbon (TOC) analyses of influent and effluent began in

February with the same frequency as the nitrates. Samples for BOD analyses were collected a few times during the study

In order to determine concentration profiles of nitrate, ammonia, and TOC within the bed, samples were collected at all ten sampling ports on a bi-weekly basis, while phosphate samples were collected once per month. In addition, twice-weekly measurements of grab samples for pH, temperature, dissolved oxygen and conductivity were taken. Daily maximum and minimum air temperatures were obtained from plant records, and from January 2000, Optic Stowaway temperature loggers (Onset Computer Corp., Procasset MA) were placed within the gravel beds of the cells to continuously record temperature. Data from these loggers were downloaded using BoxCar Pro software (Onset Computer Corp., Pocasset, MA).

Acetate addition

Towards the end of the study period supplemental carbon was added to the pilot cells to determine its effect on performance and to help shed light on the mechanisms involved in the nitrate removal. Sodium acetate trihydrate (NaAc) crystals dissolved in de-ionized water were used. The amount of acetate added for denitrification was 7.1g NaAc/gN, and was based on stoichiometry (Constantin and Fick, 1997). With an average influent of 8mg/l NO_3 and 4.5mg/l DO, 124.4g NaAc/d and 46.7g NaAc/day were estimated for the cells with HRTs of 18- and 48hrs respectively.

Concentrated NaAc solution was placed in plastic containers and applied to each bed at the influent (Figure 1). The feed rate was 5ml/min continuously for three weeks. During this period profile samples were collected for nitrate analysis while dissolved oxygen measurements were taken daily during the last two weeks.

Water Samples Analysis

Samples were analyzed according to Standard Methods (SM 1999) or EPA approved methodology. Nitrate- and Nitrate-N were determined by the automated cadmium reduction method (SM 4500NO3 F) using a Technicon Auto-analyzer II system. Ammonia-N was analyzed using the automated phenate method (SM 4500NH3 H) with the same Technicon system. Total phosphorus was determined using the colorimetric, ascorbic acid, two-reagent method (EPA 365.3). Samples for BOD analysis were filtered and then analyzed using a 5-day incubation at 20°C (SM 5210 B). The combustion - infrared method (SM 5310 B) using a total organic carbon analyzer was used to determine TOC concentrations. The TWRF laboratory performed all analyses. Dissolved oxygen and temperature were measured using a YSI model 57 Oxygen Meter.

Plant tissue analysis

Samples of the dormant cattails were collected for analysis during transplanting in January 1999. At the end of the study, in October 2000, more plant samples were collected from both pilot cells. Tissue analysis was also performed on cattails obtained from a natural site upstream of the treatment plant. Wallace Laboratories in El Segundo

California analyzed all samples. About 10% of the total tissue, representing the entire plant sample, was cut into about 2 to 3 centimeter segments. The cut tissue was dried at 70°C to a constant weight and ground with a stainless steel Waring blender then reground with a Wiley mill. The tissue was digested using the EPA 3051 procedure. Elemental and heavy metals analysis was conducted on the digested tissue with a Thermo Jarrell Ash IRIS ICP (EPA 6010). Carbon and nitrogen were measured with a Carlo Erba 2500 NA analyzer.

3.4 Results

Nitrate Removal: Figure 3-2a shows influent and effluent nitrate concentrations over the period of the study. Influent nitrate varied considerably according to season (3.3 – 21.3 mg/l) and averaged 10 mg/l. Concentrations were highest in the winter through early spring, but dropped in late spring, and remained low till through the fall season. There were corresponding variations in the effluent nitrate concentrations (1.6 – 18.9mg/l for $Cell_{18}$ and 0.4 – 15.7mg/l for $Cell_{48}$). In a few instances during rainstorms, the effluent nitrate concentrations matched or surpassed that of the influent. Reduction in nitrate concentration was lowest in the winter, highest in September, and mostly constant throughout spring and most of summer. A regression of effluent concentration as a function of influent concentration for both cells (not shown) yielded the following relationships:

$$\text{Effluent}_{(18hr)} = 0.98\,(\text{Influent}) - 1.53 \quad (R^2 = 0.94,\ n = 127)$$

$$\text{Effluent}_{(48hr)} = 0.89\,(\text{Influent}) - 3.34 \quad (R^2 = 0.87,\ n = 127)$$

Nitrate reduction, as a percentage, was used to compare performance between the two cells. Figure 3-3a shows the monthly performance of both units over time. The variations are a result of the changing influent concentrations; low performance values coincided with high influent nitrate and vice-versa. $Cell_{48}$ (50.2 ± 15%) consistently out-performed $Cell_{18}$ (20.8 ± 10%).

The beds were loaded at different rates: 18 g/d average and 7g/d for $Cell_{18}$ and $Cell_{48}$, respectively. However, removal rates were similar for both units - 3.1 grams/day for $Cell_{18}$ and 3.0 grams/day for $Cell_{48}$ (Figure 3-3b). The difference between the removal rates was not statistically significant (95% CL). Rates were lowest in January, increased through May, and remained relatively constant throughout the rest of the study.

Nitrate profiles within the bed show similar concentrations throughout in $Cell_{18}$ (Figure 3-4a). However, in $Cell_{48}$, nitrate levels were lowest at the top of the bed and increased gradually towards the bottom. Levels of ammonia in all samples from both cells always remained below detection limit (0.2 mg/l)

Carbon: The total organic carbon in the influent ranged from 6.2 to 13.7 mg/l, with an average of 9.0 mg/l. Effluent concentrations showed similar variation - 6.4 to 12.6mg/l ($8.2_{ave.}$) for $Cell_{18}$ and 6.4 to 12.6mg/l ($8.4_{ave.}$) for $Cell_{48}$ (Figure 3-2b). While influent values were mostly greater than effluent values for the first part of the study, from mid-

June, the effluent TOC concentrations of both cells exceeded that of the influent several times.

The average overall change in TOC concentration throughout the study was +0.8mg/l for $Cell_{18}$ and +0.6mg/l for $Cell_{48}$. T-tests showed no significant difference between the TOC levels in the effluents of the two cells (95% CL). Analysis of the profiles within the bed showed TOC up to 2mg/l greater than influent values at the top of the bed (Figure 3-4b). These levels were reduced by 3-4mg/l at the other depths. BOD values were generally equal to or less than 3mg/l for both influent and effluent.

Temperature: Temperatures within the bed were always 3 to 4°C degrees cooler than the influent (Figure 3-4c). Figure 3-5 compares air and water temperatures and shows air temperatures to have higher maximum and lower minimum values than water temperatures. Also, temperature fluctuations were more erratic in air than in the water. Throughout the period of study, $Cell_{48}$ showed lower maximum water temperatures than $Cell_{18}$. However its minimum temperatures were higher for the colder months (January to mid June) and lower during the warmer months.Figure 3-6 shows plots of average water temperature versus reductions in nitrate concentration for $Cell_{18}$ and $Cell_{48}$. Weak correlations of $R^2 = 0.40$ were determined for both cells.

***Phosphate*:** Figure 3-2c is a plot of influent and effluent total phosphate concentrations for both cells. Influent concentrations of phosphate ranged from 1.9 to 3.7mg/l, with an average value of 2.7mg/l. Phosphate removal was low in both cells. $Cell_{18}$ produced effluents with concentrations ranging from 2.3 to3.3 mg/l (2.6mg/l average), while $Cell_{48}$ had phosphate levels ranging from 1.8 to 2.9mg/l (2.3mg/l average). Overall, removal was consistently better in $Cell_{48}$ (95% CL). Phosphate profiles showed high concentrations at the top of the bed that declined with depth (Figure 3-4d).

Dissolved Oxygen – Figure 3-4e shows the relationship between dissolved oxygen (DO) in the influent and within the cells. The drop in dissolved oxygen ranged from 3 to 4mg/l (60 – 80%) in $Cell_{48}$ and 2 to 3mg/l (40 – 60%) in $Cell_{18}$. These values did not drop below 0.5mg/l at any point, therefore the water column was never anoxic. However anoxic zones do occur in wetlands such as these within microbial films. In both cells, there is a reduction towards the middle of the bed followed by an increase towards the bottom.

pH- Figure 3-4f shows the changes in pH with depth over time, and the changes in relation to the influent value (pH/pHo) for both cells. There was a 0.2 to 0.4 drop in pH in $Cell_{48}$. The difference in $Cell_{18}$ was less, ranging from 0.1 to 0.2 pH units.

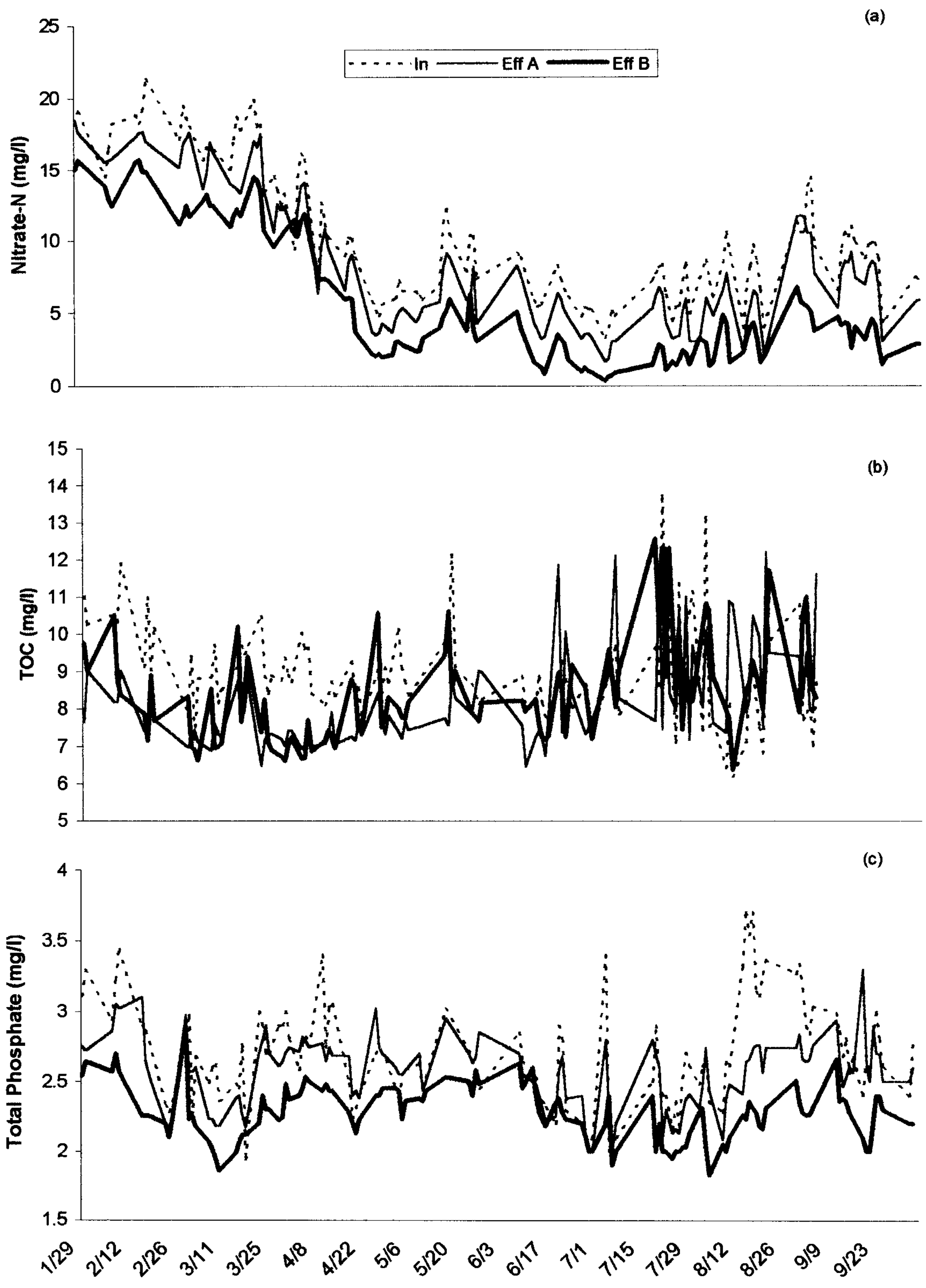
Sodium Acetate (NaAc) addition

During the period of supplemental carbon addition, influent nitrate concentrations averaged 8.7mg/l, and average effluent concentrations of $Cell_{18}$ and $Cell_{48}$ were 5mg/l and 1.6 mg/l, respectively. There was a delay in reaction time once NaAc addition began. Initially removal rates were 1.2 mg/l for $Cell_{18}$ and 5.9mg/l for $Cell_{48}$. This increased to

5.2 mg/l and 8mg/l, respectively, by the third week. Complete denitrification was not achieved since a sudden change in plant operations resulted in higher nitrate levels than had been estimated for carbon addition. In addition to this, it is likely that some of the carbon was used for processes other than denitrification, e.g. plant growth.

Mass removal rates of 6.9g/d and 4.8g/d were observed for $Cell_{18}$ and $Cell_{48}$, respectively. Figures 3-7a-b profile the effect of NaAc addition on nitrate removal in the pilot cells. The increase in nitrate removal indicates the systems' response to the availability of supplemental carbon. Changes in dissolved oxygen levels were also profiled. Significant reductions were observed in $Cell_{18}$ while reductions in $Cell_{48}$ were more modest (Figure 3-7c-d).

Figure 3-2: Influent and Effluent Concentrations over Time (a) Nitrate, (b) TOC, (c) Total Phosphate.



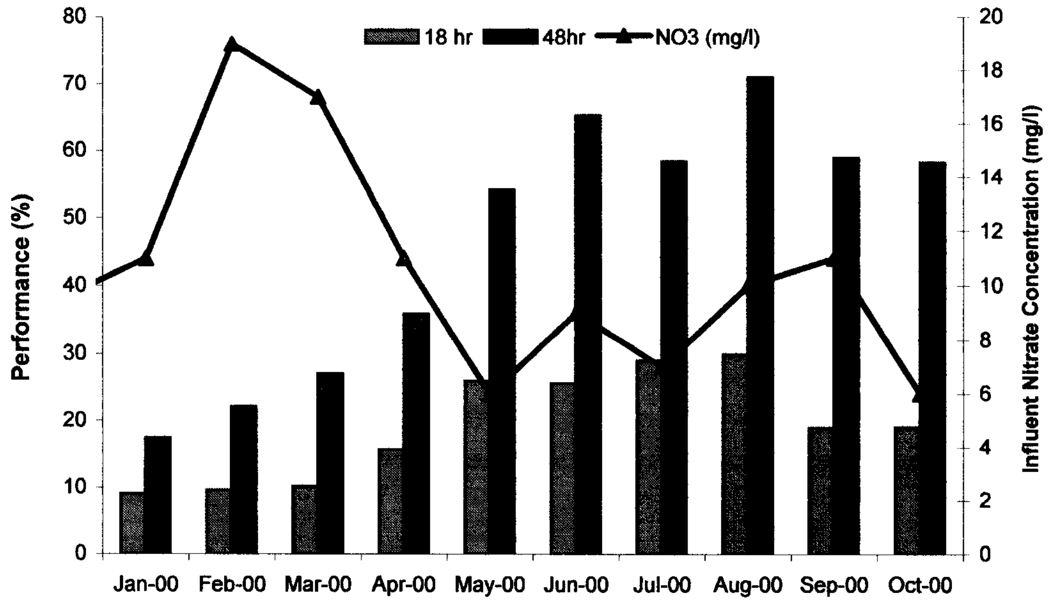


Figure 3-3a: Nitrate removal performance of wetland cells at 18- and 48-hr HRTs

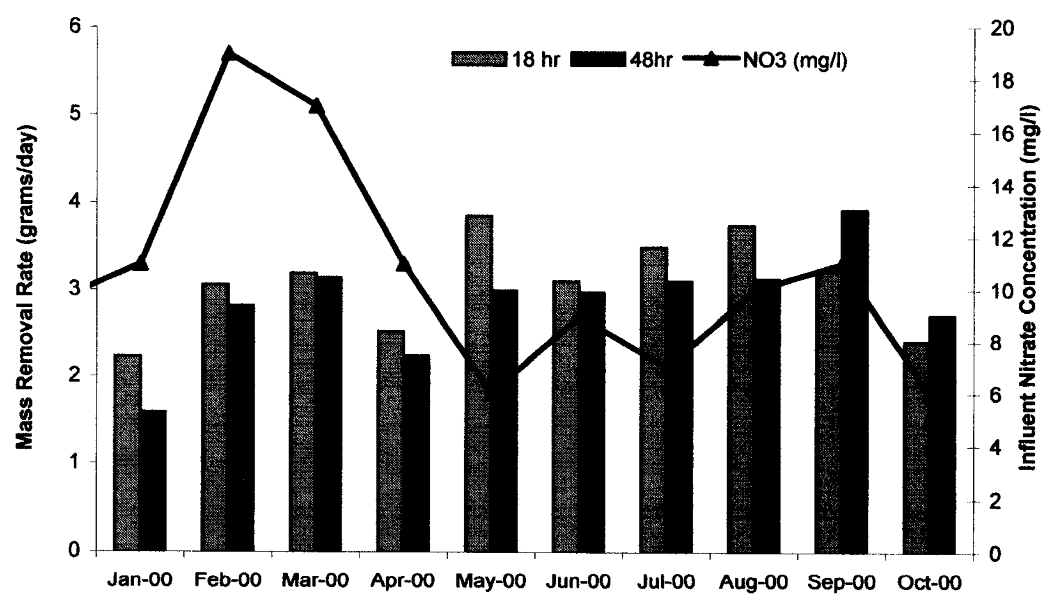


Figure 3-3b: Nitrate mass removal rates for wetland cells at 18- and 48-hr HRTs

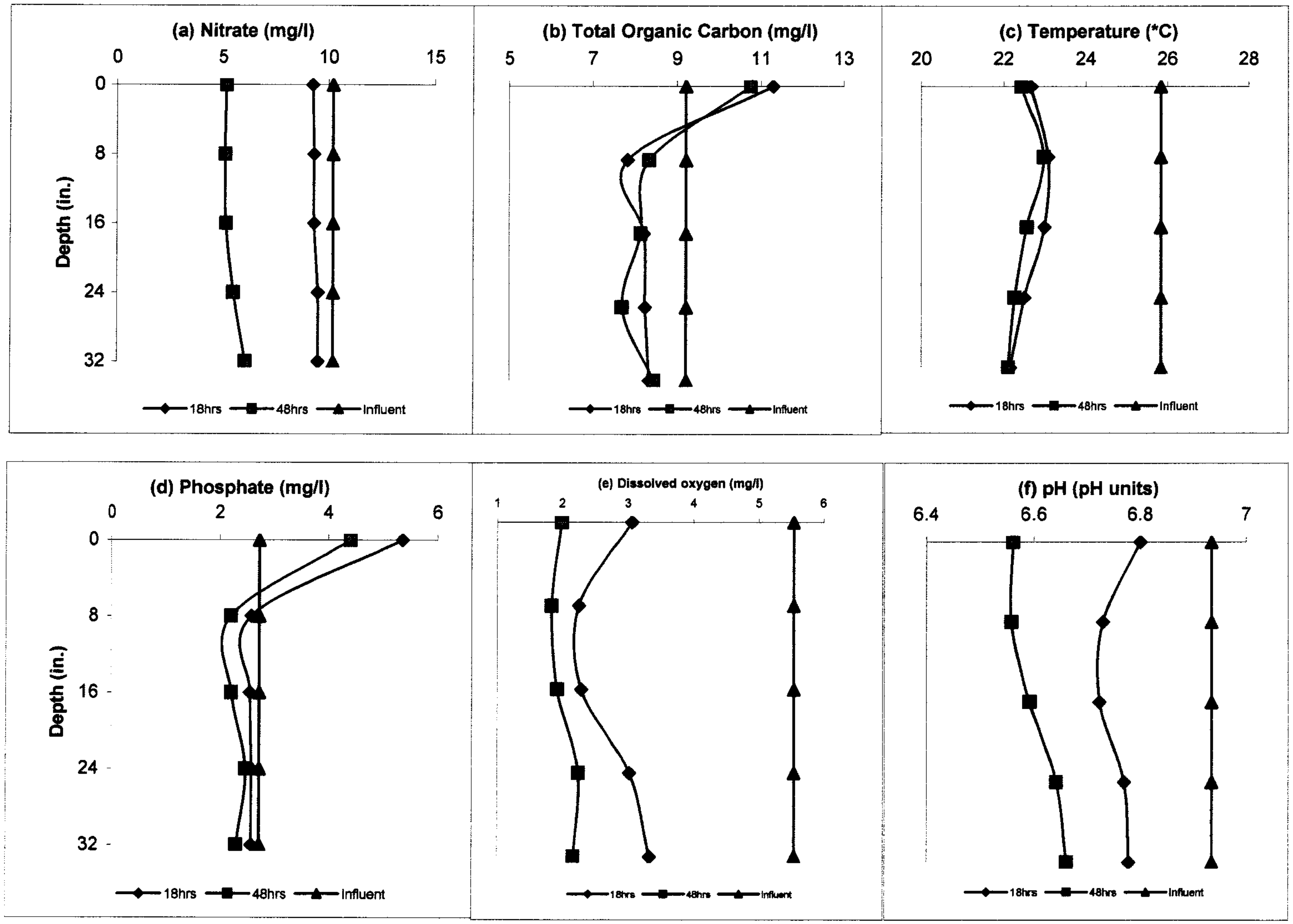


Figure 3-4: Depth profiles of (a) Nitrate, (b) TOC, (c) Temperature, (d) Phosphate, (e) Dissolved Oxygen, and (f) pH. Derived from average values of bi-weekly samples taken from February to September 2000.

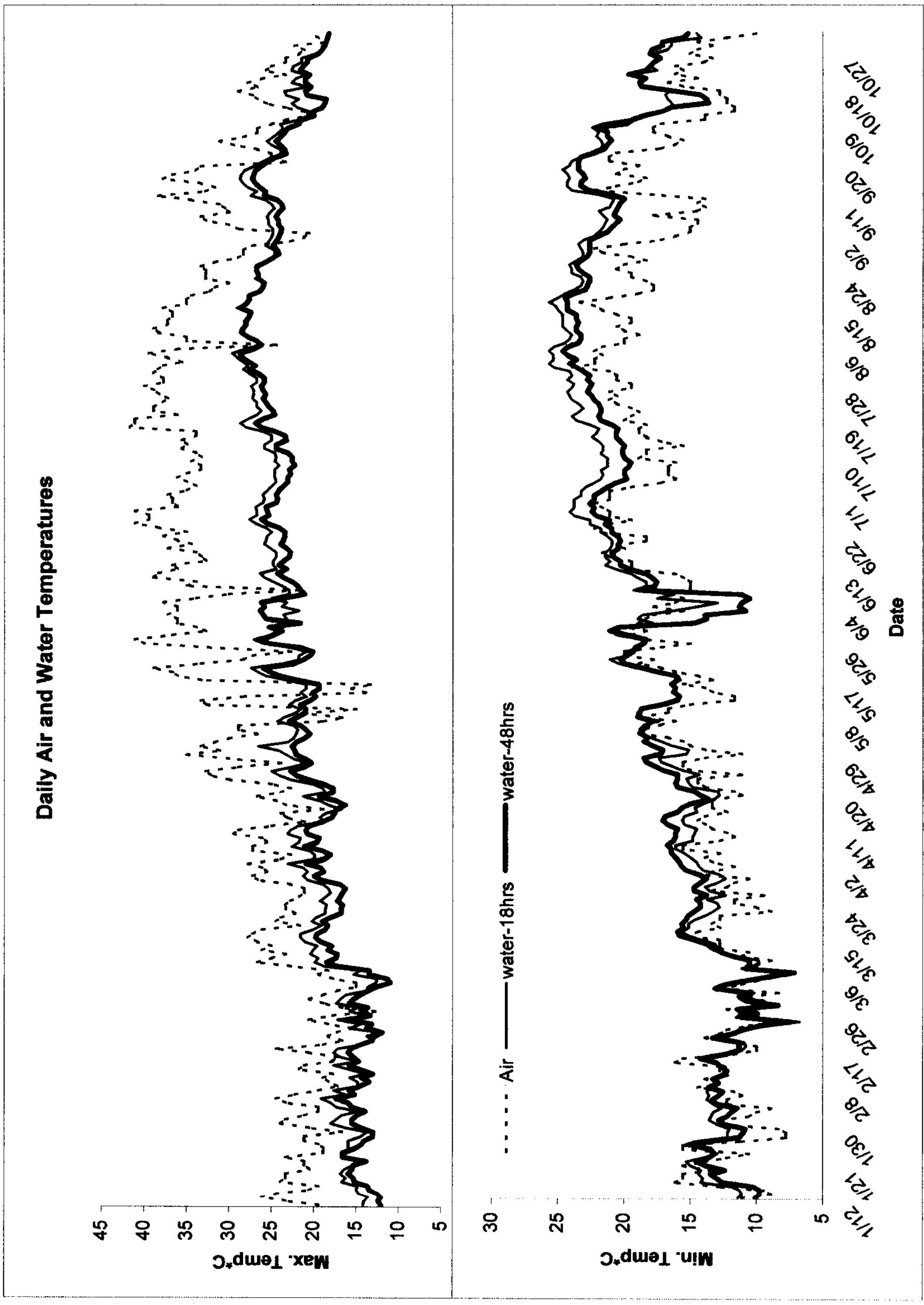
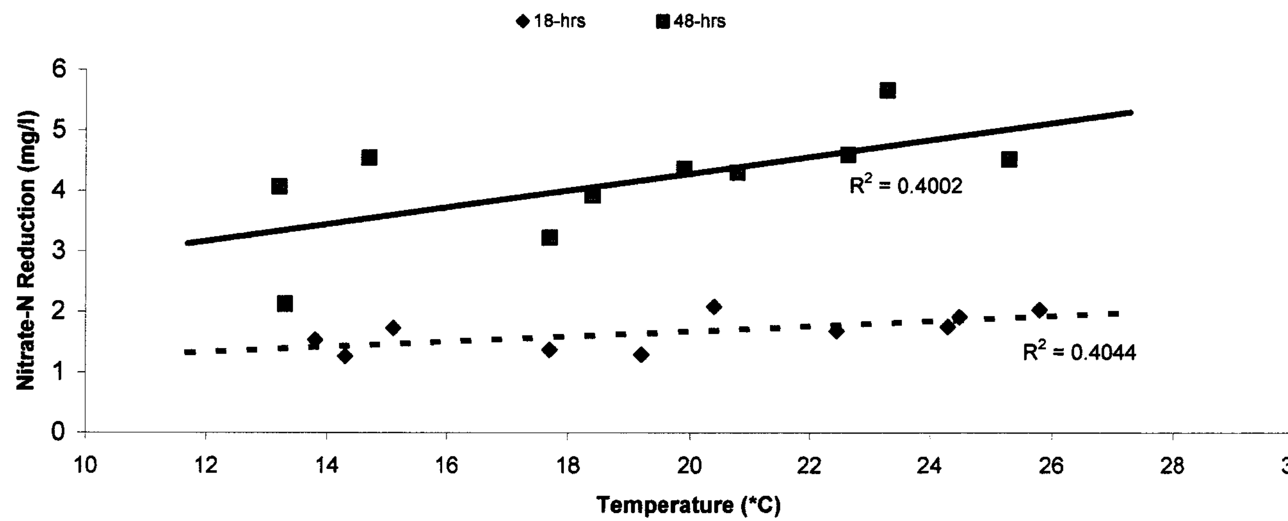


Figure 3-5: (a) Daily maximum and (b) minimum air and water temperatures over study period

Figure 3-6: Correlation between average monthly water temperature and nitrate reuction in wetland cells.



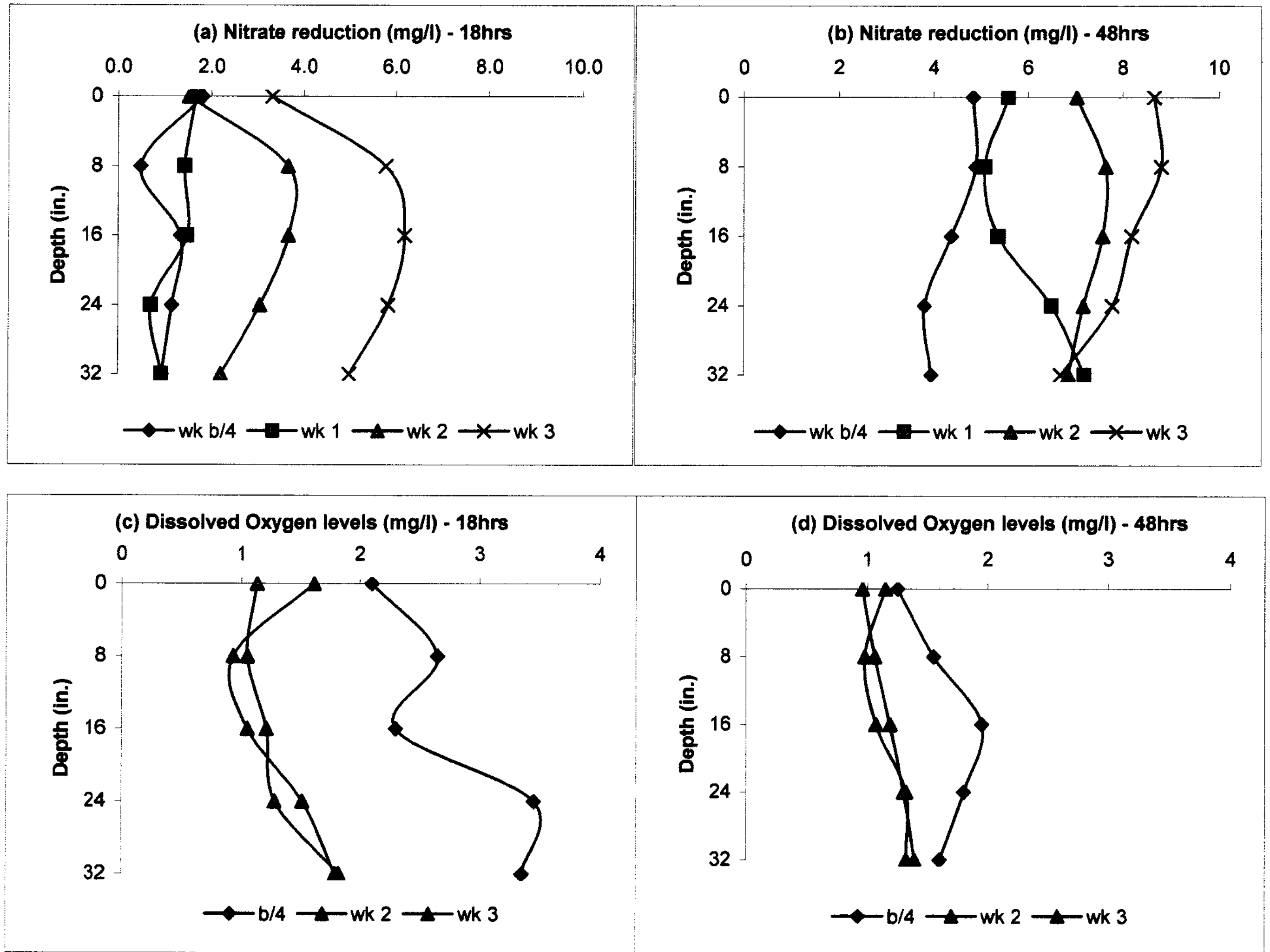


Figure 3-7: Effect of Sodium Acetate addition on (a),(b) nitrate reduction, and (b),(c) DO depletion at different HRTs
b/4 indicates the average results obtained prior to the addition of sodium acetate.

Plant analysis

Table 3-4 presents the results of the plant tissue analysis of the cattails in the pilot units along with matching parameters of the cells' influent. Higher concentrations of nitrogen, phosphorus and potassium were found in plants from the cell with the 18-hr HRT when compared to plants from the 48-hr HRT cell. This is probably a result of the higher mass loading rate of $Cell_{18}$. All non-detectable metals in the plant tissues were non-detectable in the influent to the cells. However cadmium, chromium VI, copper, and nickel, while below detection limit in the influent, were detectable in the plant tissues. Concentrations of zinc were noticeably high in plant tissue from $Cell_{18}$. All the elements in the pilot cell plant samples occurred in concentrations comparable with or lower than the amounts detected in naturally occurring samples obtained from the creek adjacent to the treatment plant.

Literature Data

Figures 3-8a and b show plots of HRT and Carbon-Nitrogen (C:N) ratios versus performance, using data from Table 3-2. Carbon values were derived from BOD concentrations provided by the literature. The analysis included data with influent nitrate concentrations of 12.6 to 26mg/l and HRTs of 6 to 120hrs. Figure 3-8a shows a modest correlation of $R^2 = 0.74$ suggesting that the C:N ratio affects performance to some degree. Hydraulic residence time showed no correlation ($R^2 = 0.06$) with performance when comparing different systems. Inclusion of the results of this study in the analysis showed that our systems' performance coincided with those studies with lower C:N ratios. The data were insufficient to perform multiple regression analysis.

Table 3-3: Composition of influent to the pilot wetland cells.

Parameter	Concentration	Unit
pH	6.9	pH units
Temperature	21.4	deg C
Dissolved Oxygen	7.5	mg/l
Biological Oxygen Demand	2.4	mg/l
Total Organic Carbon	9.0	mg/l
Total Suspended Solids	1.3	mg/l
Total Dissolved Solids	742	mg/l
Nitrate-N	10.5	mg/l
Nitrite-N	<0.01	mg/l
Ammonia-N	<0.2	mg/l
Organic Nitrogen	0.6	mg/l
Phosphate	2.6	mg/l
Total Coliform	<1.1	MPN/100ml

Table 3-4: Results of plant tissue analysis of cattails in pilot cells. N=5 for each cell.

Elements	$Cell_{18}$ Plant Tissue Conc. (mg/kg) d.w.*	$Cell_{48}$ Plant Tissue Conc. (mg/kg) d.w.	Influent Conc. (mg/l)
Essential Nutrients			
Nitrogen	10,092 ± 2,168	5640 ± 3769	10.0
Phosphorus	2,232 ± 691	1,604 ± 565	2.7
Potassium	36,341 ± 9682	18,181 ± 5,943	12.5
Priority Pollutants			
Arsenic	<0.26	<0.26	<0.010
Barium	7.6 ± 3.2	13.7 ± 7.6	0.015
Cadmium	0.30 ± 0.07	0.18 ± 0.07	<0.010
Chromium VI	13.7 ± 12.2	17.7 ± 20.3	<0.010
Cobalt	0.73 ± 0.70	0.98 ± 0.95	n.d
Copper	7.4 ± 2	5.4 ± 2	<0.008
Lead	< 0.62	<0.62	<0.010
Mercury	< 0.6	< 0.6	<0.010
Nickel	5.6 ± 4.5	6.6 ± 6.9	<0.010
Selenium	<0.98	<0.98	<0.010
Silver	<0.03	<0.03	<0.010
Zinc	238 ± 35	99 ±46	0.058

d.w.: dry weight; n.a: not available; n.d: non-detect

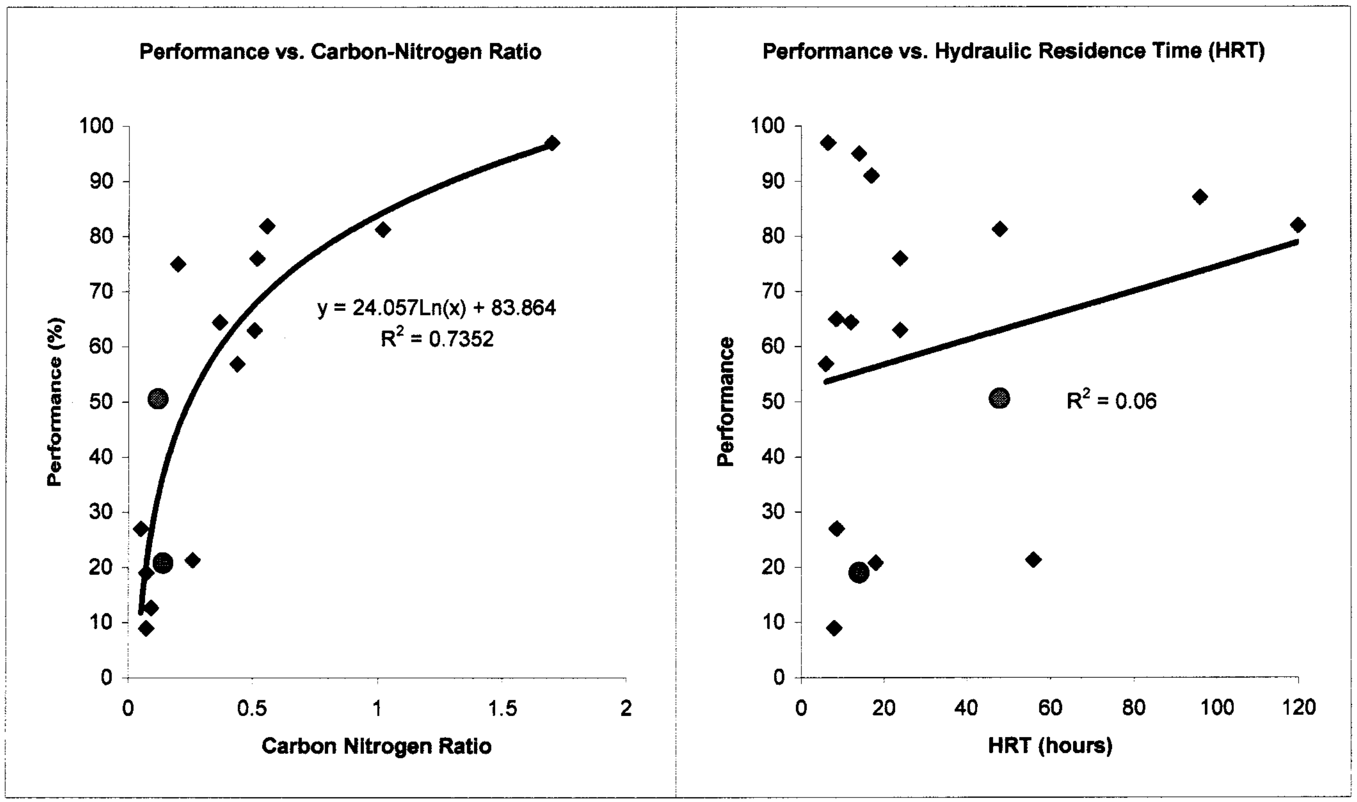


Figure 3-8:(a) Impact of C:N ratio on performance, (b) Impact of temperature on performance. Data obtained from studies reported in Table 2. The two larger data points represent results from this study.

3.5 Discussion

Nitrate Removal: Nitrate reduction is possible to some extent throughout the year in these wetland cells. Since the nitrate is predominantly in the form of nitrate, denitrification and/or plant uptake should be the major removal mechanisms involved. However, to achieve the average removals observed in the cells by denitrification, 9mg/l and 22mg/l BOD would be required in $Cell_{18}$ and $Cell_{48}$, respectively. Influent analysis showed BOD levels ranging between <2 to 3mg/l, which is insufficient to support this removal mechanism.

In addition, reductions of this magnitude were not observed upon analysis of effluent TOC. Samples collected over time revealed little change between influent and effluent concentrations for both 18- and 48-hour detention times. The type of carbon present in the highly polished influent entering the bed is less prone to degradation and hence is not able to facilitate denitrification. There was also a possibility that organic carbon is being produced within the bed from rhizome secretions and decomposition of plant material. However, continuous analysis of samples obtained from within the cells did not show significant increases in TOC levels.

With the addition of NaAc, it took over 7 days for significant changes in nitrate removal to occur. This suggests that there was little denitrification activity, and that a denitrifying population needed time to develop in the presence of increased carbon availability.

In the absence of a useable carbon source for denitrification, uptake by plants is the more likely removal mechanism. Average annual rates of nitrogen uptake have been reported between the range of 0.16 – 0.71g/m^2/d (Howard-Williams and Downes, 1993; van Oostrom and Russell, 1994; Wood,1995; DeBusk and Reddy, 1987;and Raman et al, 1997). Removal rates in this study averaged 1g/m^2/d, with lower values in the winter and the highest removal rates in late spring and throughout summer. This coincides with the literature stating that the highest uptake rates occur in spring and early summer (Kadlec, 1995; Howard-Williams and Downes, 1993). However there was no sign of the release, which is believed to occur during fall and winter. This could be explained by the continuous production of new shoots within the cells during the fall and, at a slower rate, during the winter.

In a series of small-scale greenhouse experiments using labeled ^{15}N, Zhu and Sikora (1995) observed that plant uptake was the predominant nitrate removal process (70 – 85%) for cells planted with bulrush (*Scirpus sp.*) and cattails (*Typha sp.*) in the absence of carbon. With supplemental carbon, denitrification and immobilization became the major mechanisms (55-70%). Gersberg et al. 1983 observed that nitrate removal efficiencies were greatest in their vegetated beds during the spring and summer period when plant growth and nutrient uptake were greatest. However, there was also a corresponding increase in the un-vegetated cells during the same period which implied that plant uptake was negligible. Algal production of carbon (Wood et al, 1999; Zhu and

Sikora, 1995) and immobilization (Zhu and Sikora, 1995) were suggested as mechanisms of nitrate removal in unplanted cells without supplemental carbon.

Wood et al. 1999 were able to achieve a 90% reduction in synthetic solutions containing 100mg/l nitrate in bench-scale SSF constructed wetlands in the absence of carbon. They attributed this to the availability of sufficient internal carbon (including plant root exudates and decomposing plant material) provided by the plants (*Iris pseudocoras*). Such high performance could be a result of the rapid plant uptake that occurs in new wetland systems as the plants become established. A similar effect was observed during the first year of this study.

Phosphate Removal: The low phosphate removal by the cells is consistent with what has been reported in literature. Removal mechanisms for phosphates in subsurface flow wetlands include sorption onto substrate, plant uptake, and bacterial assimilation. In subsurface flow beds, sorption is limited by the large size of the media that results in reduced available surface area, and saturation is soon reached.

The ability of $Cell_{48}$ to remove more phosphate may have been due to the additional time that would allow for additional plant uptake or adsorption. It is interesting to note that high concentrations of phosphate, at the top of the bed, indicate some form of release from the plant material However, there was no corresponding increase in nitrogen, and therefore, the source of this increase is unclear.

To effectively remove phosphorus, media with smaller grain sizes, such as clay, could be used, but these result in significant hydraulic problems. Addition of lime and alum (Davies and Cottingham 1993), and coating gravel with iron or aluminum oxides (Mann, 1990) have been suggested as other means of improving phosphorus removal performance.

3.6 Conclusions

These results indicate that a constructed wetland fed with low-carbon tertiary effluent can achieve nitrate reduction. In systems such as these with minimal supplemental carbon, plant uptake is likely the major mechanism involved in nitrate reduction. While there is probably some denitrification occurring, the lag time observed in both cells' response to supplemental carbon addition indicates that if any denitrification population was present, it was not very large. With the passage of time and the build up of organic carbon within the beds, denitrification would play a more significant role in nitrate removal. This would occur when the vegetation has come to equilibrium and plant uptake rates match release rates.

With a nitrate removal rate of 1 g/m^2d, the available land area could remove up to 2.6kg/d of nitrogen without supplemental carbon. In the event that greater removal is required, increasing the detention time of the system should result in improved treatment. However, this may be uneconomical in areas where land acquisition costs have to be taken into consideration. No significant reduction in phosphorus is expected for this system.

3.7 References

Æsøy, A., Odegard, H., Bach, K., Pujol, R., Hamon, M. 1998. Denitrification in a Packed Bed Biofilm Reactor (BIOFOR) – Experiments with Different Carbon Sources. Wat. Res. 32(5):1463-1470.

Alexander, M. (1961) Introduction to Soil Microbiology. John Wiley & Sons Inc. New York.

Amstrong, W., Amstrong, J., Beckette, P.M. (1990) Measurement and Modeling of Phragmites Australis. In: Constructed Wetlands for Water Pollution Control. P.F. Cooper and B.C. Findlater ed., Pergamon Press, Oxford, 41-51.

Bachand, P.A.M. (1996) Effects of Managing Vegetative Species, Hydraulic Residence Time, Wetland Age and Water Depth on Removing Nitrate from Nitrified Wastewater in Constructed Wetland Macrocosms in the Prado Basin, Riverside County, California. PhD Dissertation, Submitted to University of California, Berkeley (1996).

Brix, H., Schierup, H. (1990) Soil oxygenation in Constructed Reed Beds: The role of Macrophyte and Soil-Atmosphere Interface Oxygen Transport. In: Constructed Wetlands In Water Pollution Control. Cooper, P.F., Findlater, B.C ed.

Brix, H. (1993) Macrophyte-Mediated Oxygen Transfer in Wetlands: Transport Mechanisms and Rates. In: Constructed Wetlands for Water Quality Improvement. G.A. Moshiri ed.

Constantin, H., Fick, M. (1997) Influence of C-Sources on the Denitrification Rate of a High-Nitrate Concentrated Industrial Wastewater. *Wat. Res.* **31**, No.3. 583-589.

Cooke, J. G., (1994) Nutrient Transformations in a Natural Wetland Receiving Sewage Effluent and the Implications for Waste Treatment. *Wat. Sci. Tech.* **29,** No. 4. 209-217.

Coombes, C., and Collett, P.J. (1995) Use of Constructed Wetland to Protect Bathing Water Quality. *Wat. Sci. Tech.* **32**. No 3. 149 –158

Davies, T.H., Cottingham, P.D. (1993) Phosphorus Removal from Wastewater in a Constructed Wetland. . In Constructed Wetlands for Water Quality Improvement. G.A. Moshiri ed.

DeBusk, W.F., Reddy, K.R. (1987) Removal of Floodwater Nitrogen in a Cypress Swamp Receiving Primary Wastewater Effluent. *Hydrobiologia.* **153**. 79-86.

Gersberg, R.M., Elkins, B.V., Goldman, C.R. (1983) Nitrogen Removal in Artificial Wetlands. *Water Res.* **7**, No.9. 1009-1014.

Gersberg, R.M., Elkins, B.V., Goldman, C.R. (1984a) Use of Artificial Wetlands to Remove Nitrogen from Wastewater. *Journal WPCF* . **56,** No. 2. 152-156.

Gersberg, R.M., Elkins, B.V., Goldman, C.R. (1984b) Wastewater Treatment by Artificial Wetlands. *Wat. Sci, Tech* **17**. 443 –450

Gersberg, R.M., Elkins, B.V., Lyon, S.R., Goldman, C.R. (1986) Role of Aquatic Plants in Wastewater Treatment by Artificial Wetlands. *Water Resources* **20**: 363-368.

Green, B., Verhoeven, C. (1999) Removal of Ammoniacal and Total Oxidized Nitrogen in Subsurface flow Wetlands used for Tertiary Treatment. In: Nutrient Cycling and Retention in Natural and Constructed Wetlands. J. Vymazal ed., Backhuys Publishers, Leiden, 19 – 29.

Hammer, D. A., Knight, R. L. (1994) Designing Constructed Wetlands for Nitrogen Removal. *Wat. Sci. Tech.* **24** No.4. 15-27.

Howard-Williams, C., Downes, M.T. (1993) Nitrogen Cycling in Wetlands. In: Nitrate: Processes, Patterns and Management. T. P. Burt, A. L. Heathwaite, and H. G. Trudgill ed.

Ingersoll, T.L., Baker, L.A. (1998) Nitrate Removal in Wetland Microcosms *Wat. Res.* **32,** No. 3. 677-684

Kadlec, R.H. (1995) Overview: Surface Flow Constructed Wetlands. *Wat. Sci. Tech.* **32,** No.3. 1-12.

Laber, J., Perfler, R., Haberl, R. (1997)Two Strategies for Advanced Nitrogen Elimination in Vertical Flow Constructed Wetlands. *Wat. Sci. Tech.* **35**. No 5. 71-77.

Mann, R.A., "Phosphorus Removal by Constructed Wetlands: Substratum Adsorption". In: Constructed Wetlands In Water Pollution Control. Cooper, P.F., Findlater, B.C ed. 97-105

Metcalf and Eddy Inc. (1991) Wastewater Treatment, Disposal, and Reuse. McGraw Hill, New York.

Mistch , W. J., Gosselink, J. G. (1993) Wetlands. Van Norstrand Reinhold. New York.

Nicols, D. S. (1983) Capacity of Natural Wetlands to Remove Nutrients from Wastewater. *Journal of WPCF*, **55,** No. 5. 495–505.

Pfenning, K. S., McMahon, P.B. (1996) Effect of Nitrate, Organic Carbon, and Temperature on Potential Denitrification Rates in Nitrate-Rich Riverbed Sediments. *Journal of Hydrology* **187**. 283-295.

Raman, D. J., Beal, L. J., Rye, S. W. Stiles, A.C., Yoder, R. E., and Eash, N.S. (1999) Kinetics of COD and N Removal in Constructed Wetlands Treating Dairy Wastewater. Presented at the 1997 *ASAE International Meeting*, Minneapolis, MN. Paper No. 97411.

Reed, S.C., Crites, R.W., Middlebrooks, E.J. (1995) Natural Systems for Waste Management and Treatment. McGraw-Hill, Inc. New York.

Reuter, J.E., Djohan, T., Goldman, C.R. (1989) The use of Wetlands for Nutrient Removal from Urban Runoff in a Cold Climate Region of California – Preliminary Results from a Newly Constructed Artificial Wetland at Lake Tahoe. Division of Environmental Studies, University of California, Davis.CA. Report.

Stengel, E., and Schultz-Hock, R. (1989) Denitrification in Artificial Wetlands" In Constructed Wetlands for Wastewater Treatment Edited by D.A.Hammer.

Tennessee Valley Authority (TVA) 1990. Design and Performance of the Constructed Wetland Wastewater Treatment System at Phillips High School, Bear Creek, Alabama. Report TVA/WR/WQ—90/5.

United States Environmental Protection Agency (USEPA). (1993) Subsurface Flow Constructed Wetlands for Wastewater Treatment: A Technology Assessment. Report EPA 832-R-93-008.

Van Oostrom, A. J., Russell, J.M. (1994) Denitrification in Constructed Wastewater Wetlands Receiving High Concentrations of Nitrate. *Wat. Sci. Tech.* **29,** No. 4. 7-14.

Vymazal, J. (1990) Use of Reed-Bed Systems for the Treatment of Concentrated Wastes from Agriculture. In: Constructed Wetlands In Water Pollution Control. P.F. Cooper and B.C. Findlater Ed. 347 – 358

Water Environment Research Foundation (WERF). (2000) Constructed Wetlands Treatment of High Nitrogen Leachate. Project 94-IRM-U.

Williams, J.B., May, E., Ford, M.G., Butler, J.E. (1994) Nitrogen Transformations in Gravel Bed Hydrophonic Beds used as a Tertiary Treatment Stage for Sewage Effluents. *Wat. Sci. Tech.* **29,** No.4. 29-36.

Williams, J., Baghat. M., May, E., Ford. M., Butler, J. (1995) Mineralization and Pathogen Removal in Gravel Bed Hydrophonic Constructed Wetlands for Wastewater Treatment. *Wat. Sci. Tech.* **32,** No 3. 49-58.

Wood, A. (1995) Constructed Wetlands in Water Pollution Control: Fundamentals to their Understanding. *Wat. Sci. Tech.* **32,** No. 3. 21-29.

Wood, S.L., Wheeler, E.F., Berghage, R.D., Graves, R.E. (1999) Temperature Effects on Wastewater Nitrate Removal in Laboratory-Scale Constructed Wetlands. *Transactions of the ASAE* **42,** No.1 185-190

Zhu, T., Sikora, F.J., (1995) Ammonium and Nitrate Removal in Vegetated and Unvegetated Gravel Bed Microcosm Wetlands. *Wat. Sci. Tech.* **32,**. No 3, 219-228.

CHAPTER 4: CONSTRUCTED WETLANDS TO REDUCE NUTRIENT LOADING IN A WATERSHED: A CASE STUDY OF MAILBU, CALIFORNIA

Abstract

Constructed wetlands are natural treatment systems that are able to serve a dual purpose of removing water borne pollutants and enhancing natural habitat. These attributes make such systems a feasible treatment option for addressing water quality concerns in watersheds such as the Malibu Creek Watershed in Los Angeles California. Nutrient enrichment by nitrogen and phosphorus compounds, and high levels of coliform bacteria in Malibu Creek and Lagoon are two major problems in the watershed and constructed wetlands are capable of reducing these pollutants.

This paper discusses the effective application of wetland technology in the Malibu Watershed based on results of a pilot study. Constraints on the implementation of this technology are discussed along with benefits and concerns. Wetlands also create and enhance natural habitats, which is one of the long-term goals of the watershed. It is expected that long-term operation of a demonstration wetland would provide further insight for the use of this technology as a watershed management tool.

4.1 Introduction

Nutrient enrichment, from point and non-point sources, is one of the contributing factors to the problem of algal growth and blooms in Southern California Mediterranean-type watersheds. Other factors such as temperature and light intensity are less controllable; therefore the main focus of algae abatement efforts are geared towards reducing nitrogen and phosphate levels in surface waters. This is the case in the Malibu Creek Watershed, which is located in northwest Los Angeles County, California. Malibu Creek and Lagoon are listed as impaired for nutrients by the California State Water Resources Control Board (CSWRCB, 1996).

The Malibu Watershed is 79% undeveloped and is made up of federal and state parklands and other open spaces. Non-point sources of pollution include animal waste, septic systems, wildlife contributions, and excess home fertilizer application (CWRCB, 1990). Non-point source control policies have primarily involved the promotion of best management practices (BMPs) such as control of construction site run-off and public education on proper handling of livestock waste and the application of fertilizers. The single point source discharge of nutrients to the watershed is the Tapia Wastewater Reclamation Facility (TWRF), which is owned and operated as a joint venture between the Las Virgenes Municipal Water District (LVMWD) and the Triunfo Sanitation District. To date control of this point source has involved regulatory limits on nutrient levels in effluent discharges, and a 7-month dry-weather discharge prohibition.

Treatment wetlands can reduce both point and non-point sources of nitrogen and phosphorus compounds (Hammer, (1989) and Moshiri, (1993)). They have been used successfully in California and throughout the US to treat municipal wastewater and stormwater containing these compounds; and also provide the added benefit of habitat creation and enhancement. The objective of this paper is to discuss the effective application of wetland technology in the Malibu Watershed based on results of a pilot-study. Constraints on the implementation of this technology and relative benefits and concerns are also presented.

4.2 Background: Application of Wetland Technology in Watershed Management

The concept of applying treatment wetland technology to address water quality problems was introduced to the Malibu Creek Watershed in 1989 when Save Our Coast, an environmental advocacy group, spearheaded an attempt to look at potential wetland use for the elimination of pathogens entering Malibu Creek. This effort resulted in a report that identified potential constructed wetland sites in the watershed (Figure 4-1), with suggested treatment objectives for each (Gearhart and Waller, 1989). Ambrose and Orme (1999) revisited the issue and listed treatment wetland construction as one of the water resource management alternatives for lower Malibu Creek and the Lagoon. The study identified six potential sites in the area, and determined that the limited availability of land for such systems could reduce the feasibility of their future implementation. Most recently a report by the Malibu Coastal Land Conservancy analyzed the feasibility of

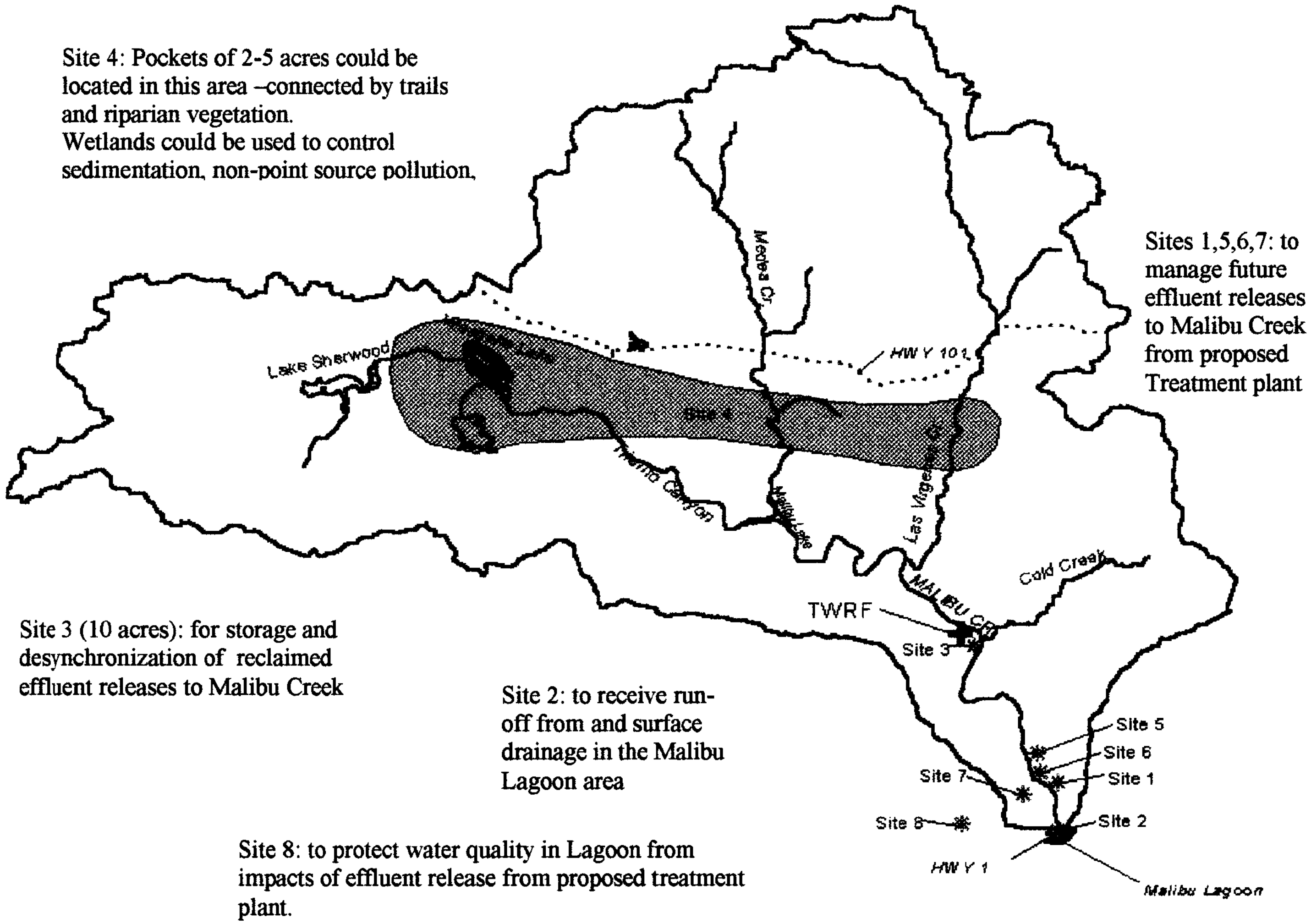


Figure 4-1: Potential Treatment Wetland Sites in the Malibu Creek Watershed
(adapted and modified from Gearhart and Waller, 1989)

restoring a wetland area adjacent to Malibu Lagoon to divert and treat dry-weather creek flows and release treated water to the lagoon. The study concluded that while technically feasible, land acquisition for the project would be the limiting factor (Huffman and Carpenter, 2000).

In 1994, the LVMWD began considering converting a percolation pond to a treatment wetland. The purpose of the wetland was to reduce nutrient levels in their effluent for part of the year, and to divert and treat a portion of Malibu Creek flows for pollutants carried by urban run-off. The ponds are located on the 10-acre site adjacent to Malibu Creek (Site 3 in Figure 4-1), earlier identified by Gearheart and Waller (1989) as having the potential for storage and de-synchronization of reclaimed effluent. The ponds themselves occupy less than 2-acres and consist of nineteen 8ft wide x 5ft deep trenches dug in parallel formation, with lengths varying from 80 - 350 ft.

The California State Department of Parks and Recreation (DPR) maintains ownership of the land on which these ponds are sited and allows LVMWD to operate them under an informal agreement. DPR was willing to consider the treatment wetland concept as long as (i) the potential for water quality improvement could be demonstrated, (ii) any proposed modifications would be limited to the existing pond area, and (iii) the finished project would leave the site close to natural conditions.

Given the limited area available, a sub-surface flow (SSF) system was deemed more practical than a free water surface (FWS) wetland, which would require more area per to treat a given volume (Wood, 1995). SFF systems provide greater contact opportunities between water and microorganisms, and therefore require shorter detention times, which translate to reduced land requirements. Another factor considered was TWRF's need to use the percolation ponds as a disposal site for excess reclaimed effluent. The ponds originally had a disposal capacity up to one million gallons per day. However, a brief study conducted in the spring of 1999 estimated the capacity to be 700,000 to 800,000 gallons per day (see Appendix B).

4.3 Nutrient-Reduction Performance of Pilot Wetlands

Since the performance of wetlands in nutrient removal varies depending on wetland type, nature of the influent and environmental conditions, a pilot study was conducted to obtain site-specific operational information - prior to actual construction. Two wetland cells, each with a surface area of $35ft^2$, were operated at different hydraulic retention times (HRTs) - 18 and 48 hours. Tertiary treated effluent served as the influent to these wetlands. TWRF effluent has high nitrate (10 mg/l average) and lower phosphate concentrations (3 mg/l). The performance of the pilot units with regard to nitrate and phosphate reduction is reported in Tables 4-1a-c. Nitrate removal was higher in the summer and fall than in spring and winter. The longer HRT consistently produced a lower effluent nitrate concentration (Table 4-1a); however a removal rate of about $1 g/m^2/d$ was observed in both systems for the duration of the study. Nitrogen uptake by wetland plants seemed to be the mechanism of removal in this instance. The application of supplemental organic carbon in the form of sodium acetate (NaAc) resulted in a gradual improvement in nitrate reduction for both units (Table 4-1b). As is common in most sub-surface flow wetlands, phosphate removal was poor for both units - though the longer HRT produced effluents of lower concentration throughout the study (Table 4-1c).

Table 4-1a: Results of pilot studies for nitrate reduction

	Summer 99		Fall 99		Winter 00		Spring 00		Summer 00	
Parameter	18hr	48hr	18hr	48hr	18hr	48hr	18hr	48hr	18hr	48hr
Influent	18.6	10.25	9.5	8.29	16.5	16.3	8.4	8.4	7.9	7.9
Effluent	13.5	0.7	7.3	4.16	14.9	12.3	6.7	4.5	6.0	2.9
mg/l reduction	**5.1**	**9.6**	**2.2**	**4.13**	**1.6**	**4.0**	**1.7**	**3.9**	1.9	4.9
Performance (%)	**27.4**	**93.2**	**23.2**	**49.8**	**9.7**	**24.5**	**20.2**	**46.4**	26.9	63.7

Table 4-1b: Effect of supplemental carbon addition on nitrate removal in pilot cells.

	Week 1		Week 2		Week 3	
Parameter	18hr	48hr	18hr	48hr	18hr	48hr
Influent	8.1	8.8	7.9	7.3	10.2	10.2
Effluent	5.2	2.7	5.0	1.4	4.6	0.5
mg/l reduction	2.9	6.2	2.9	5.9	5.6	9.0
Performance (%)	34.7	70.2	35.7	81.7	54.9	93.9

Table 4-1c: Phosphorus removal in pilot cells

	Winter 00		Spring 00		Summer 00	
Parameter	18hr	48hr	18hr	48hr	18hr	48hr
Influent	2.73	2.73	2.57	2.57	2.77	2.77
Effluent	2.57	2.25	2.56	2.36	2.54	2.20
mg/l reduction	0.16	0.48	0.01	0.21	0.22	0.57
Performance (%)	5.9	17.6	0.3	8.0	8.0	20.6

Projected performance of Proposed Wetland

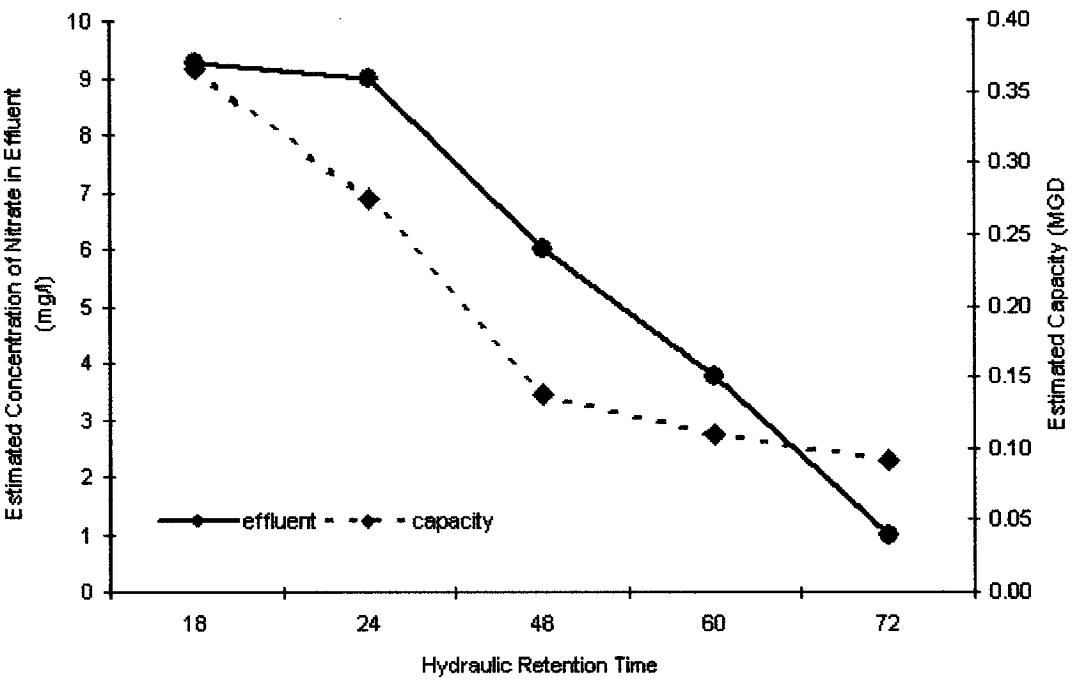
Nitrogen Removal: Results from the pilot-study indicate that the available land area is capable of removing $1 g/m^2/d$ (2.6 kg/d) of Nitrate-Nitrogen. Figure 4-3 translates this into the volume of tertiary effluent that can be treated per day (wetland capacity), and estimates the quality of the discharged effluent based on an influent nitrate concentration of 10 mg/l. The impact of residence time is evident. When operating at an 18hr HRT the capacity of the wetland would be as high as 360,000 gallons per day (gpd) however it would drop to less than 100,000 gpd at an HRT of 72 hours. In contrast, the quality of effluent would improve significantly from 9 mg/l at an 18-hr HRT to 1 mg/l at 72-hr HRT. In the instance where plant uptake is the major mechanism of nutrient reduction, and available wetland area is limited, the residence time of the influent in the wetland will determine the extent of treatment. Since the mass removal rate is constant determining the optimal retention time would depend on whether a lower effluent concentration or a higher disposal capacity is desired.

These performance estimates are made with the assumption that nitrate removal rates will remain dependent on plant uptake. This is unlikely since SSF wetlands are able to build up sufficient organic carbon stores over time to make denitrification a significant removal pathway. Once this occurs nitrogen removal would be more a function of optimal denitrification conditions within the wetland; and the treatment capacity of the wetland is likely to increase. Results from the study also indicate that the addition of supplemental carbon could increase the nutrient reduction capacity of the current site to as much as 5.7

kg/d. This increase in efficiency could either reduce the land requirements for a given volume of water, or significantly decrease the nutrient levels in the wetland effluent. However, potential drawbacks to this option include (i) higher operation and maintenance cost, (ii) increased organic loading to the receiving water if excess amounts are applied, and (iii) potential resistance by stakeholders and environmental agencies to the application of chemicals within the wetland. If the natural organic matter of the wetland develops over time as expected, the need for carbon addition may only be temporary.

Phosphate removal: The mass removal rate of phosphorus within the wetland was 0.7 - 0.09 $g/m^2/d$. Removal of this constituent is by sorption to the gravel media and/or plant uptake. SSF wetlands usually have limited sorption capacities, and this removal rate may not be sustained over time as sorption sites get used up. Also phosphorus release from decaying plant litter may match uptake rates as the system matures, leading to the elimination of this mechanism of removal. Table 4-2 provides estimates of the potential reduction in nitrogen and phosphorus loading in LVMWDs effluent using the demonstration wetland and with possible expansion to the entire 10 acres. These reductions are just a fraction of the current loading however; they are really an indication of the degree of treatment possible with a more widespread application of the technology.

Figure 4-2: Estimated Changes in Wetland Capacity and Performance with varying Hydraulic Retention Times



Capacity based on available wetland area for treatment and 40% substrate porosity. Treatment efficiency based on an assumed influent concentration of 10mg/l NO3-N. Results based on plant uptake as sole removal mechanism.

Figure 4-2: Estimates of potential nutrient load reductions at wetland site

Nutrient	Nutrient Reduction (lbs/year)			
	Current Project (< 2 ac.)		Expanded Project (10 acres)	
		with carbon supplement		with carbon supplement
Nitrate	431	916	2873	6101
Phosphate	38	38	252	252

Potential use of wetland technology elsewhere in the watershed

Incorporating wetlands into septic system technology: Septic systems contribute 9.5% of the nitrate and 5.3% of the phosphorus load to the Malibu Creek Watershed (Tetra Tech, 2001). Septic systems are usually made up of a septic tank for anaerobic digestion of wastes, and a leach-field for adsorption, filtering and subsequent disposal. In MCW there are estimated 2400 residential septic systems and 20 commercial ones and it is estimated that 20-30% of the septic systems are failing due to high water tables and unsuitable soils. This results in septic system discharges to the creek and lagoon via groundwater. Table 4-3a shows estimates of septic tank discharge concentrations (LARWCB, 2000) and the amount estimated to be reaching the groundwater from normal, failing, and short-circuited systems. Nitrogen in these systems is predominantly in the form of ammonia.

Individual subsurface flow wetlands have been used to replace the leach fields of septic systems with promising results. Wolverton (1989) reported on the performance of one such system that had been in operation for several years. High removal rates for ammonia (75%) and fecal coliform (97%) were achieved with this system. A more recent study by Steer et al; 2002, reported ammonia reduction of 56 % phosphorus 80 %, and fecal coliform 88% in single-family constructed wetland systems in Ohio - prior to percolation of the wetland effluent. Table 4-3b shows the potential reductions in septic system discharge in the Malibu Creek Watershed prior to percolation if used in conjunction with SSF wetlands. These modified systems can be used to gradually replace conventional septics as they fail or malfunction or could be used in new construction projects. This technology may also assist commercial systems in meeting new limits being set for the nutrient and coliform TMDLs.

Figure 4-3a: Estimates of pollutant concentrations reaching waterbodies in the Malibu Creek Watershed

Parameter	Septic Tank Effluent Concentrations*	Condition of System		
		Normal	Failing	Short Circuiting
Nitrogen	59.2 mg/l	51.0 mg/l	51.0 mg/l	51.0 mg/l
Phosphorous	9.9 mg/l	-	6.93	8.6
Fecal Coliform	1.0×10^7	-	4×10^6	1.0×10^7

*Source: LARWQCB (2000)

Figure 4-3b: Estimates of potential nitrogen load reductions based on performance reported in literature

Type of System	Total flow (gpd)	Potential Reductions in Nitrogen Load (lbs/yr)	
		Wolverton (1989) –75%	Steer et al (2002) –56%
Residential	657,600	15,791	11,778
Commercial	75, 000	2,090	1,345

gpd = gallons per day

4.4 Discussion

Land Availability: One issue common to all constructed wetland projects is their extensive land requirement. The size of a wetland usually depends on the amount of water to be treated, type of wetland, desired effluent quality, and other proposed uses of the wetland. Land acquisition is not usually an issue in some rural areas where land is abundant and cheap. This is not the case in the Malibu Creek Watershed. The potential sites identified by Ambrose and Orme (1999), and Gearhart and Wallace (1989) exist on state-owned lands and privately owned areas of rapid development.

The limited availability of land highlights the potential role that the California Department of Parks and Recreation (CDPR) could play in the watershed approach to improving water quality. CDPR is a major landowner in the Malibu Creek Watershed and has jurisdiction over most of the prime watershed land adjacent to the creek. This agency is constantly involved in the further acquisition of land for preservation therefore acquisition of land from them via purchase is a very unlikely option. CDPR conceded to the proposed construction of the wetland on the project site in part because it was already "disturbed" by the existing percolation ponds that were constructed prior to their gaining ownership of the land. Creation of wetlands at this site was considered an improvement of its current condition.

CDPR could consider using more of its lands for water quality improvement in the watershed. This may be done in a similar manner to what transpired in the case of this project. The State would maintain ownership and jurisdiction over the land and grant authority to stakeholders and interested agencies to construct, operate and maintain wetlands for the purpose of habitat creation and enhancement, and water quality improvements. Since CDPR has the responsibility of preserving these lands in their natural condition for public benefit, it needs to be demonstrated that the creation of such systems and the subsequent improvement of water quality within the watershed are public benefits well worth considering. The demonstration-scale wetlands project is an opportunity to make such a showing and if successful could be expanded in scale at the site, or elsewhere in the watershed.

Permitting Issues: A number of permits were required for this project to go to completion:

Use-Permit from the CDPR: This permit grants the use of the proposed site under CDPRs jurisdiction for the wetland project. The conditions of use were as described earlier. To date a CEQA document with a finding of no significant impact has been prepared by the agency but final approval is still pending.

Waste Discharge Requirement: The Regional Water Quality Control Board issued this permit. It was necessary since discharge of effluent to a wetland is considered a form of land disposal; a WDR was required prior to construction of the project, to setup conditions of use and monitoring protocol. Some controversy arose since release of effluent to the wetland would occur at some point during an existing discharge prohibition to the creek. Environmental groups expressed concern that a discharge of effluent to the wetland during this period constituted an indirect discharge to Malibu Creek and should be disallowed. These concerns were addressed by the provision of data to show that impacts to lagoon water levels were unlikely, and the WDR was granted.

Streambank Alteration Permit: This permit will be obtained from the California Department of Fish and Game for the diversion of Malibu Creek flows to the wetland. It has to be demonstrated that there will be no significant harm to wildlife and aquatic organisms caused by the diversion.

Coastal Development Permit: Since the site is located within the coastal zone, this permit will be obtained from the California Coastal Commission.

Benefits and Limits of technology

Pollutant Reduction in the Watershed: This project has the potential to reduce nutrient loading to the creek (Table 4-2 and 4-3b). It also has the capability to reduce high levels of coliform bacteria in portions of Malibu Creek flows. Removal efficiencies of 99.9% have consistently been documented in literature. Since creek flows are lowest in the summer, treatment of the greatest proportion of flow would coincide with the highest recreational use period.

Creation and Enhancement of habitat: These wetlands could provide additional habitat that will attract wildlife such as insects and other invertebrates, birds (ducks, geese) amphibians (frogs, toads) and small mammals such as marsh rats, rabbits and beavers). Filling and planting the existing ponds will bring the site closer to natural conditions than what currently exists. Also given the aerial extent of native plantings required, wetlands may assist in stalling the spread of invasive plant species that is becoming a concern in the watershed.

Recreation and Education Opportunities: As a disposal site the area had limited public access since there were safety concerns associated with the percolation ponds. This project will allow this segment of parklands to be enjoyed by the public. The creation of new habitat might foster bird watching and nature-walking opportunities at the site. Also, since the surrounding area is used for other outdoor activities such as hiking, horseback riding, biking, fishing, and camping, having wetlands in their vicinity should enhance enjoyment of these activities. The project will also provide an educational opportunity highlighting the use of natural systems in environmental remediation. It can serve as a model for future wetland projects in the watershed

Resurfacing during discharge prohibition: There has been concern that water applied to the wetland will eventually end up in the creek resulting in elevated water levels in the lagoon, which will lead to the failure of septic systems in that area. However, available data suggests that this is not the case, as lagoon water levels monitored during periods of discharge to the percolation pond showed no increase from periods of no discharge (LVMWD, 2001).

Metal toxicity: There is the concern that plants receiving continuous effluent could build up high concentrations of pollutants that may be detrimental to wildlife foraging among them. Plant tissue analysis was conducted on some plants from the pilot wetlands. These plants had been receiving tertiary effluent continuously for 20 months at the time of sampling. Metal concentrations were compared to those of plants collected from Malibu Creek (upstream of TWRF) that were not subject to effluent discharges. Table 4-4 presents the results of the comparison and shows that aside from zinc, all the pollutants in the pilot wetland samples occurred in concentrations comparable or lower than the amounts detected in the samples collected from the creek. These results imply that there may be no net increase in exposure to toxic metals from the wetlands.

Table 4-4: Results of plant tissue analysis of samples from Malibu creek (upstream of TWRF), and the two pilot cells. All values are in mg/kg.

Elements	Malibu Creek	Pilot 18hrs	Pilot 48hrs
Priority Pollutants			
Arsenic	< 0.26	<0.26	<0.26
Barium	12.7	7.6 ± 3.2	13.7 ± 7.6
Cadmium	0.48	0.30 ± 0.07	0.18 ± 0.07
Chromium VI	28.66	13.7 ± 12.2	17.7 ± 20.3
Cobalt	3.63	0.73 ± 0.70	0.98 ± 0.95
Copper	6.94	7.4 ± 2	5.4 ± 2
Lead	< 0.62	< 0.62	<0.62
Mercury	< 0.06	< 0.06	< 0.06
Nickel	16.72	5.6 ± 4.5	6.6 ± 6.9
Selenium	< 0.98	<0.98	<0.98
Silver	< 0.03	<0.03	<0.03
Zinc	35	238 ± 35	99 ±46

4.5 Conclusion

The final design of the proposed Malibu site serves as a constructed wetland and land disposal system. Pollutant reduction is expected to occur in two phases. Prior to percolation, the incoming flow will be treated within the gravel bed, and further treatment will occur during the downward passage of the water through the sediment. Traditionally, water within SSF wetlands is prevented from percolating into the underlying soil by the use of compacted soil, synthetic materials, or under-drain systems. However since effluent discharge is prohibited from this site for seven months of the year, percolation will serve as a means of disposal – with the added benefit of natural treatment. The limited removal of phosphate by the wetland should be boosted by subsequent soil infiltration. The detention time and hence the treatment capacity of this wetland will depend primarily on the rate of infiltration through the underlying soil.

While modest in size and treatment objectives, implementation and successful operation of the project will encourage similar ones within the watershed in the future. Also demonstration of the immense public benefit to be gained by the use of such technology may orchestrate some provision of more public lands for the improvement of water quality in the watershed. Treatment wetlands could be even more effective when used in conjunction with other nutrient-reduction measures throughout the watershed.

Future studies

The pilot-scale study was conducted for a period of 20-months on LVMWD effluent alone. It is expected that the demonstration project will be able to give broader insight into the capabilities of this technology. Some issues that may be clarified include (i) the optimal conditions for phosphate reduction, (ii) the extent of metals uptake, (iii) the impact of maturation on performance, and (iv) the ability of wetland plants to ward off invasive plant species. Such information will be useful for future operations and when applying this technology elsewhere in the watershed.

4.6 References

Ambrose, R., Orme, A. 2000. Lower Malibu Creek and Barrier Lagoon System Resource Enhancement and Management. University of California, Los Angeles/

California Regional Water Quality Control Board Los Angeles Region (CRWQCB-LA). 2000. Preliminary Results of the Malibu Technical Investigation.

California Sate Water Resources Control Board (CSWRCB). 1996. Water Quality Assessment Report.

Gearhart, R.A., Waller, G. 1989. Overview of Wetland Opportunities in the Malibu Creek Watershed. Environmental Resources Engineering Department, Humboldt State University Arcata, California.

Hammer, D.A. (Ed). 1989. Constructed Wetlands for Wastewater Treatment; Municipal, Industrial and Agricultural. Lewis Publishers, Chelsea, Michigan.

Hoffman and Carpenter Inc. August 2000. Malibu Wetland Feasibility Study.

Las Virgenes Municipal Water District (LVMWD). 2000. Unpublished Data.

Moshiri, G. A. (Ed). 1993. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor. McGraw-Hill, Inc. New York.

Steer, D., Fraser, L., Boddy, J., Seibert, B. (2002). Efficiency of Small Constructed Wetland for Subsurface Treatment of Single-Family Domestic Effluent. *Ecol. Engr*. 18. 429-440

Wolverton, B. C. 1989. Aquatic Plant Microbial Filters for Treating Septic Tank Effluent. In Constructed Wetlands for Wastewater Treatment; Municipal, Industrial and Agricultural. D.A. Hammer (Ed). Lewis Publishers, Chelsea, Michigan. 173-178.

Wood, A. 1995. Constructed Wetlands in Water Pollution Control: Fundamentals to their Understanding. Wat. Sci. Tech. 32(3): 21-29.

CHAPTER 5: CONCLUSIONS

5.1 Synopsis

The algae blooms that plague Malibu Creek and Lagoon during late spring and summer, are a major water quality concern in the Malibu Creek Watershed. Nutrient enrichment of these water bodies is a significant factor that contributes to this phenomenon. Sources of nitrogen and phosphorus in the watershed include the tertiary effluent of the Tapia Wastewater Treatment Facility (TWTF), and several non-point sources such as residential and commercial septic systems, golf courses, and animal waste. Past efforts to curtail nutrient inputs have focused on restricting TWTF's discharge of effluent to Malibu Creek, via increasingly stringent nutrient limits. These efforts have been ineffective in controlling the proliferation of algae.

More recently the focus has broadened to encompass non-point sources like commercial and multi-family septic systems, which are now regulated. In addition best management practices for the handling of animal waste, and excess fertilizer applications, are being encouraged. This more inclusive approach to addressing nutrient enrichment can be further enhanced by applying innovative technologies to reduce nitrogen and phosphorus on a watershed scale.

Constructed wetlands are effective in removing pollutants such as nitrogen and phosphorus from their influent. These systems rely on natural processes such as denitrification, plant uptake, sorption and precipitation for pollutant reduction. This results in low operational and maintenance costs. The additional benefits of habitat enhancement and creation have generated increasing interest in this technology.

In a pilot plant project, subsurface wetland technology was shown to be effective in reducing nitrate in the effluent of a wastewater treatment facility (TWTF). The mass removal rate of $1 g/m^2/d$ was attributed to plant uptake, as there was insufficient carbon within the wetland to support denitrification. Longer detention times within the wetland resulted in lower effluent nitrate concentrations. With the addition of a supplemental carbon source, towards the end of the study period, nitrate removal rates showed a steady increase, suggesting that denitrification was taking on a more significant role. Phosphate removal in these wetlands was not significant. This is likely a result of poor sorption by the gravel media. Media with smaller grain sizes, such as clay usually have much higher sorption capacities.

The Las Virgenes Municipal Water District is conducting the first attempt at wetland creation in the Malibu Creek Watershed. This involves the conversion of a 2-acre percolation pond site to a series of subsurface wetland cells to remove nitrate from their TWTF effluent, and coliform bacteria from Malibu creek flows. Results from the pilot study indicate that the site could achieve a 90% reduction in nitrate concentration of up to 100,000 gallons/day of effluent prior to discharge to Malibu Creek.

Expansion of the constructed wetland technology throughout the watershed is limited by land availability. A significant portion of the land in the watershed is publicly owned and is operated by agencies committed to their preservation. In order to site wetlands in such areas, it may be necessary to demonstrate that the resulting improvement in water quality and creation of habitat are benefits that will enhance the use of these public lands. Therefore any proposals for wetland creation should involve these agencies in the early stages. Other prime wetland sites occur on privately owned areas of rapid development. Acquisition of these privately owned parcels for wetland creation will be difficult and cost-prohibitive, and is currently not a feasible option.

It is important to acknowledge the limitations of these wetland systems. Phosphorus removal is limited in SSF wetlands; and FWS wetlands, once saturated, can act as a source of phosphorus in their effluent. Consideration of the other benefits of these systems in the creation of natural habitats that enhance the surrounding areas, and the low cost and maintenance requirements, make wetlands a practical alternative to achieve measurable improvements in water quality within the watershed. However, not all sources of nutrient in the watershed can be mitigated with the application of wetland systems, therefore it should be used in conjunction with other technology and/or BMPs.

5.2 Future Work

While this dissertation has focused on sub-surface flow wetland technology, it is important to note that free water surface (FWS) wetlands would be as beneficial to the watershed from a standpoint of treatment capability. FWS wetlands also offer a wider array of habitats and hence more ecological benefits. However as mentioned previously land requirements for these wetlands will serve as a limiting factor in this watershed.

One adaptation of the wetland system could be to create more favorable conditions within the creek itself for nutrient reduction to occur. Providing sufficient available carbon and increasing detention time could be achieved with minimal modification to the creek bed. Planting of cattails and bulrushes over entire segments of the creek bed will serve to slow flows while providing a constant source of organic carbon for processes such as denitrification. Plants would also provide direct uptake of nutrients and pollutants –albeit a small fraction, and provide a substrate for microbial growth. Slowing creek flows could also result in greater opportunities for phosphate sorption to creek sediment. The increase in detention time will also create opportunities for the degradation of pathogenic bacteria. With this approach significant reductions in pollutants may be achieved particularly in the drier months.

Wetlands for stormwater treatment have the potential to alleviate pollution loading to the creek via run-off. These wetlands could also reduce sediment deposition downstream by reducing the velocity of run-off flow through the creek in pre-treatment detention basins.

The down side to this would be the loss of scouring velocity of the storm flows that help rid the creek of its algae every winter. Storm water run-off could be better addressed through low-flow stormwater diversions to treatment plants, particularly since the "first flush" of rain normally carries the majority of the pollutants.

The majority of the efforts to control algae in the watershed have focused on the impact of nutrients loading despite observations that without anthropogenic inputs, background levels are capable of supporting excessive growths. While this should not prevent further nutrient reduction efforts, future work should be geared towards gaining a better understanding of the contribution of environmental factors to the proliferation of algae in the watershed. Chapman (1980) reported on the conditions of flow velocity, light intensity and temperatures at which different algae seemed to flourish. Studies on the impact of modifying some of these conditions on the incidence of algae may provide greater insight to the problem. A wealth of background data on algae and nutrient levels along the creek exist to provide the necessary baseline for comparison. For example the change in algae cover with the increase of shading –provided by trees and other vegetation – at a specific site could be monitored. Shading would have the combined effect of lowering temperatures and lowering light intensity. It is important that future studies be specific to the watershed to remove the uncertainties introduced by extrapolating results from other areas.

Finally, it may very well be that Malibu Creek and Lagoon are naturally prone to eutrophication; and if so it may be prudent to examine continuous physical removal of algae from the creek rather than trying to get nutrient concentrations down to unrealistic levels - at prohibitive costs.

APPENDIX A: PRELIMINARY DATA FROM PILOT-SCALE WETLAND STUDY

This appendix contains preliminary data collected form two-pilot scale wetland cells from March to November 1999. The wetland cells were constructed to determine the possible extent of nitrate removal from a tertiary-treated effluent in subsurface flow wetlands – with limited organic carbon present in the effluent. Reliance was to be solely on the organic carbon generated within the cell from plant detritus and excretions.

The details of the set up are provided in Chapter 2. The units were filled with gravel and planted with rhizomes in January 1999. Following this, the beds were inoculated with activated sludge from the treatment plant. Collection of samples began in March of 1999 when new growth had begun in the units.

Both pilot cells were operated at hydraulic detection times (HRTs) of 6-, 12-, 18-, 24-, and 36-hours from March to June of 1999. During this period grab samples were collected from both cells and analyzed for nitrate, nitrite, ammonia, and phosphate. The resulting data is contained in Tables A-1a and A-1b.

From July to September 1999 the cells were operated at. 18-, 24-, 36-, and 48-hours. Composite samples were taken this time (see Table A-2). In addition, samples were collected at all ten sampling ports and analyzed for nitrates, total organic carbon. Field measurements of pH, dissolved oxygen, and temperature were also taken at these ports. The data from the sampling ports are presented in Tables A-3 through A-7. There was no sampling in August and the early part of September as the plant effluent which served as the influent to the cells contained residual chlorine. Flow to the beds was halted for this period.

In October to November 1999, the pilot cells were operated at different detention times – 18-, and 48hrs and weekly composite sampling for nitrates was conducted. Data obtained from these analyses are presented in Table A-8. The pilot cells were not sampled in December 1999.

During this period of initial data collection, there were earlier problems of incidental flow interruption to the beds and a continuous problem with water level changes within the pilot cells. The flow to the cells had to be adjusted constantly and the water level in the cells frequently needed correction. In January 2000, standpipes were added to each cell, which stabilized water levels in both beds.

The data presented in this appendix was not collected under the same stable operating conditions, as those after January 2000. Therefore they were not included in the report presented in Chapter 3 of this dissertation.

Table A-1a: Results from grab samples of pilot cells' influent and effluent from March to June 1999 - Nitrate and Nitrite Concentrations

HRT	Date	Nitrate (mg/l)			Nitrite (mg/l)		
		Influent	Cell1	Cell2	Influent	Cell1	Cell2
6hrs	3/10/99	17.5	17.2	17.1	<0.5	<0.5	<0.5
	3/24/99	13.4	9.7	12.3			
	3/26/99	15.4	13.4	14.7			
	4/7/99	11.2	12.3	12.15	<0.01	0.01	0.02
	5/5/99	20.2	19.4		<0.8	<0.8	
12 hrs	5/19/99	20.15	19.4				
	6/1/99	10.5	7.95	9.25	<0.2	<0.2	<0.2
18hrs	6/22/99	10.05	6.1	5	nd	nd	nd
	6/24/99	12.35	7.15	4.15	nd	nd	nd
24hrs	6/3/99	16.8	8.2	9.5	<0.2	<0.2	<0.2
	6/8/99	13.5	9.2	10.15	<0.2	<0.2	<0.2
	6/10/99	18.25	13.8	12	nd	nd	nd
36hrs	6/15/99	15.4	8.6	6.75	0.1	0.2	0.3
	6/17/99	13.25	6.4	1.45	nd	nd	nd

Table A-1b: Results from grab samples of pilot cells' influent and effluent from March to June 1999 - Ammonia and Phosphate Concentrations

HRT	Date	Ammonia (mg/l)			Phosphate (mg/l)		
		Influent	Cell1	Cell2	Influent	Cell1	Cell2
6hrs	3/10/99	<1	<1	<1	2.9	2.9	2.8
	3/24/99	nd	nd	nd			
	3/26/99	<1	<1	<1	3.0	2.9	2.9
	4/7/99	<1	<1	<1	2.4	2.8	2.5
	5/5/99	nd	nd	<1	3.0	2.9	2.9
12 hrs	5/19/99	<2	<2	<2	2.9	2.9	2.9
	6/1/99	<1	<1	<1	2.5	2.0	2.1
18hrs	6/22/99	0.3	0.4	0.3			
	6/24/99	0.3	0.3	0.1			
24hrs	6/3/99	<1	<1	<1	2.0	2.0	2.1
	6/8/99	<1	<1	<1	2.1	2.0	2.2
	6/10/99	<1	<1	<1	3.1	2.0	2.2
36hrs	6/15/99	<1	<1	<1	2.7	2.1	2.0
	6/17/99	0.4	0.6	0.4			

Table A-2: Results from composite samples of pilot cells' influent and effluent in July and September 1999 - Nitrate and Ammonia Concentrations

HRT	Date	Nitrate (mg/l)			Ammonia (mg/l)		
		Influent	Cell1	Cell2	Influent	Cell1	Cell2
48hrs	7/14/199	8.4	0.7	0.1	<1	<1	<1
	7/16/99	11.9	1.1	1.2	<1	<1	<1
36hrs	7/22/99	13		7.5	<1	<1	<1
	7/23/99	13.4	9.7	9.5	<1	<1	<1
24hrs	7/27/99	11.5	7.2	6	<1	<1	<1
	7/28/99	15.8	11.8	11.1	<1	<1	<1
18hrs	9/22/99	20.2	13.6	11.2	<1	<1	<1
	9/23/99	17	14.7	14.5	<1	<1	<1

Table A-3 Profiles of Nitrate concentration within the pilot cells at varying hydraulic retention times.

HRT	Depth (in)	Nitrate (mg/l) - 9/22/99 Cell1	Cell2	Nitrate (mg/l) - 9/23/99 Cell1	Cell2
18hrs	0	16.8	11.4	18.2	
	8	13.6	12.7	16.1	16.7
	16	14.2	13.9	15.7	17.2
	24	14.2	14.5	16.4	18.4
	32	14.0	13.7	16.1	18.3
		7/27/99		7/28/99	
24hrs	0	11.0	11.1	15.5	15.7
	8	9.6	10.2	15.3	14.4
	16	8.7	8.7	15.9	14.5
	24	8.7	10.4	16.1	16.0
	32	10.5	10.5	16.3	16.1
		7/22/99		7/23/99	
36hrs	0	11.2		14.2	7.7
	8	13.2	11.6	9.1	8.2
	16	12.5	12.2	8.6	8.2
	24	12.9	14.0	8.6	8.2
	32	13.5	12.6	9.4	8.8
		7/14/99		7/16/99	
48hrs	0	3.6	2.2	10.0	3.0
	8	1.2	1.5	4.9	1.2
	16	1.9	1.2	5.4	1.9
	24	0.8	0.7	5.6	2.5
	32	0.7	0.8	5.9	2.1

Table A-4 Profiles of Dissolved Oxygen concentration within the pilot cells at varying hydraulic retention times.

HRT	Depth (in)	DO (mg/l) - 9/22/99 Cell1	Cell2	DO (mg/l) - 9/23/99 Cell1	Cell2
18hrs	0	4.6	2.1	3.6	
	8	2.5	2.8	2.3	2.2
	16	2.2	2.7	2.4	3.0
	24	3.0	2.6	3.0	4.3
	32	3.4	3.1	3.3	4.6
		7/27/99		7/28/99	
24hrs	0	4.3	2.8	4.0	2.7
	8	2.0	2.2	2.6	3.0
	16	1.7	1.5	2.3	2.5
	24	1.8	2.6	2.7	2.6
	32	2.2	2.4	3.0	2.9
		7/22/99		7/23/99	
36hrs	0	4.4	1.6	4.3	1.3
	8	1.5	2.0	1.1	1.6
	16	1.8	2.6	1.3	1.5
	24	2.7	3.3	2.1	2.3
	32	2.9		2.3	2.4
		7/14/99		7/16/99	
48hrs	0	2.3	1.3	3.2	1.1
	8	1.3	1.6	1.0	1.2
	16	1.3	1.8	1.0	1.3
	24	1.8	1.8	1.5	1.9
	32	2.1	1.3	1.9	2.0

Table A-5 Profiles of Temperature concentration within the pilot cells at varying hydraulic retention times.

HRT	Depth (in)	Temp (degC) -9/22/99 Cell1	Cell2	Temp (degC) - 9/23/99 Cell1	Cell2
18hrs	0	22.5	22.8	22.8	
	8	22.3	23.0	22.3	25.8
	16	22.3	23.0	22.3	25.5
	24	22.0	22.8	22.0	25.0
	32	21.5	22.0	22.0	24.5
		7/27/99		7/28/99	
24hrs	0	25.5	24.0	23.0	23.5
	8	24.0	24.3	23.0	23.3
	16	24.3	23.5	22.8	22.5
	24	23.5	22.5	22.3	23.0
	32	23.3	21.5	21.3	23.0
		7/22/99		7/23/99	
36hrs	0	25.0		24.3	24.5
	8	25.3	24.8	23.8	24.5
	16	24.0	25.0	24.3	23.5
	24	23.0	24.5	22.5	23.3
	32	23.3	24.3	21.5	22.8
		7/14/99		7/16/99	
48hrs	0	28.0	27.3	25.5	26.0
	8	29.0	27.3	24.5	25.8
	16	28.0	27.8	24.3	25.5
	24	28.0	27.0	24.0	24.8
	32	28.0	27.3	23.3	25.0

Table A-6 Profiles of pH within the pilot cells at varying hydraulic retention times.

HRT	Depth (in)	pH - 9/22/99 Cell1	Cell2	pH -9/23/99 Cell1	Cell2
18hrs	0	6.8	6.7	6.7	
	8	6.8	6.7	6.7	6.6
	16	6.7	6.8	6.7	6.7
	24	6.8	6.8	6.7	6.7
	32	6.8	6.8	6.7	6.7
		7/27/99		7/28/99	
24hrs	0	6.4	6.5	6.8	6.6
	8	6.5	6.1	6.6	6.5
	16	6.3	6.2	6.6	6.6
	24	6.2	6.2	6.6	6.1
	32	6.1	6.1	6.6	6.6
		7/22/99		7/23/99	
36hrs	0	6.4		5.8	6.3
	8	6.0	6.2	6.2	6.2
	16	6.1	6.4	6.2	6.5
	24	6.1	6.2	5.9	6.5
	32	5.4	6.3	6.4	6.5
		7/14/99		7/16/99	
48hrs	0	6.4	6.4	6.8	6.7
	8	6.7	6.5	6.8	6.8
	16	6.6	6.5	6.8	6.8
	24	6.4	6.3	6.8	6.8
	32	6.5	6.4	6.8	6.8

Table A-7 Profiles of Total Organic Carbon concentration within the pilot cells at varying hydraulic retention times.

HRT	Depth (in)	TOC (mg/l) - 9/22/99 Cell1	Cell2	TOC (mg/l) - 9/23/99 Cell1	Cell2
18hrs	0	12.4	11.5		
	8	9.2	9.7		
	16	9.5	9.4		
	24	10.4	12.1		
	32	10.2	11.8		
		7/27/99		7/28/99	
24hrs	0			10.8	7.7
	8			7.5	8.2
	16			7.5	7.5
	24			7.1	7.1
	32			7.3	7.0
		7/22/99		7/23/99	
36hrs	0	8.7		7.9	6.7
	8	7.0	6.3	6.1	6.3
	16	6.8	6.2	5.9	6.5
	24	6.7	6.8	6.0	5.9
	32	6.2	6.0	6.1	6.0
		7/14/99		7/16/99	
48hrs	0			8.1	7.9
	8			8.4	7.0
	16			6.8	7.3
	24			8.1	9.3
	32			7.2	7.0

Table A-8 Influent and Effluent Nitrate concentrations of the pilot cells operating at different detention times - Oct to Nov 1999.

Date	Cell 1(18hr HRT)		Cell 2 (48hr HRT)	
	In (mg/l)	Eff (mg/l)	In (mg/l)	Eff (mg/l)
17-Oct	13.68	11.56	14.20	11.94
14-Oct	12.29	8.85	8.85	5.50
21-Oct	9.18	7.60	7.80	4.63
28-Oct	6.73	6.31	7.48	2.11
4-Nov	6.83	5.05	6.40	1.73
11-Nov	7.43	4.20	6.20	1.06
18-Nov	10.58	7.19	7.10	2.17

APPENDIX B: PERCOLATION POND CAPACITY STUDY – SPRING, 1999

Summary

This eight-week study was conducted to determine the optimum disposal capacity of the percolation pond site located across the road from Tapia Reclamation Facility - and adjacent to Malibu Creek. Different operating modes were examined and continuous loading between 700,000 – 8000,000 gallons per day was determined to be the best choice

Objectives

The objective of this study was to determine the most effective use of the ponds as an effluent disposal site during the flow prohibition period from May 1st till October 31st. This involved the study of different loading applications detailed in the methodology section.

Site description

The site originally consisted of nineteen ponds that were operated in parallel (Figure B-1). They were later converted to series mode by the installation of overflow pipes. Flood damage resulted in a channel cutting through the berms of the ponds and ending in the thirteenth pond (Figure B-2). The remaining ponds are cut off from flows.

Effluent from Tapia enters the first pond through a 10-inch diameter concrete pipe (Figure B-3). Flow then continues through the channel and terminates in pond 13 where the water level builds up (Figure B-4). A measuring staff was installed in this pond to determine percolation (infiltration) rates. Once this pond fills up, it spills over (Figure B-5) – it is therefore the discharge-limiting portion of the site. The height difference between the overflow pipes and the final water level prevent flow of water into the adjacent ponds.

The rest of the ponds receive varying amounts of water but are not used to their full capacity.

Methodology

The study was conducted in four phases (see Figure B-6).

1. Single batch daily loading: effluent discharge at a high flowrate – once a day.
2. Semi-continuos flow: continuous loading throughout the daytime for 12-hour periods. There were 12-hour periods of no flow between loadings.
3. Continuos flow: effluent discharge at a constant low rate throughout the study period
4. Double batch daily loading: early morning loading with additional discharge later in the day

In all cases, the loading rates were dependent on the water level in the effluent pond. Also, flows were periodically reduced by start up of pumps sending effluent to the reservoir.

Results and Analysis

Phase I: commenced on February 16th 1999 and lasted ten days. Prior to start up, effluent had not been discharged to the ponds for almost four months (116 days) although some degree of leaking did occur. The last rain of 0.59 inches occurred four days before discharge.

On the first day, a total of 417000 gallons of effluent was discharged over 4 hours until pond 13 was filled up. This entire volume was absorbed overnight, leaving the ponds empty by the next morning. After this, an average of 300,000 gallons per day (gpd) was discharged over 2-2.5 hours.

Additional discharges were made to the ponds in the form of a persistent leak, at low flowrates (≤ 0.2 mgd), over several hours. This contributed a substantial amount (7-37%) to the total discharge to the ponds during this period.

The leakage occurred through the sides of the drop-gate controlling discharge to the ponds (Figure B-7). The magnitude was dependent on the water level in the effluent pond at a given time. When the effluent pumps turned on to supply water to the reservoir, the water level dropped and the leakage reduced or completely stopped. The plot of discharge over time (Figure B-8) shows the drop of batch flow to a constant that is maintained throughout the study period. It also shows the extent of leakage that occurred.

A correlation analysis was performed to determine the effect of leakage, and daily on the pond capacity. A negative correlation was shown between leakage and the amount of water discharged in batch flow ($r = -0.87$), indicating that high leakage may result in reduced capacity for absorption of batch discharge (Figures B-9). However there was little correlation between temperatures and capacity ($r = 0.004$ to -0.25)

The infiltration rate was measured as the water level dropped in pond 13. Short term (3-hr) rates started off at 4 inches per hour (in/hr) on the first day and dropped to between 2 and 2.5 in /hr. The overnight infiltration rate was calculated from the water level at the time discharge stopped, till just before the next application. It decreased from 2.15 in/hr on the first day to about 1.6 in/hr where it remained for the last seven days of the study. Residual water level in the ponds began to build up on the third day and fluctuated between 13 – 15 inches for the last five days of the study. A summary of the results from phase I is given in Table B-1.

Phase II: The pond was dried out over the next five days and phase II began on March 3rd and lasted ten days. This was a semi-continuous loading operation with a discharge rate of 1mgd for 12 hours to achieve 500,000 gallons per day. This required operators to come in 3-hrs earlier than normal every day. The discharge volume was selected based on anecdotal estimates of previous performance. The average controlled discharge, over the 12-hr periods, was 477,300 gpd. Effluent leakage constituted an average of 9.5% of the total discharge (Figure 10) during this phase. Residual water began to build up in pond 13 after the first day and had reached 17" by the tenth day.

Phase III: began on March 18th and lasted for 7-days. It involved the continuous application of effluent at a rate of about 0.7mgd. This rate fluctuated between 0.65 and 0.75 mgd and was completely absorbed by the ponds over the entire period. There was no build up of head in pond 13 as had been with the other operating modes. The average volume discharged was 687,000 gallons. The duration of the study was cut short to allow for unrelated work to be done on the site. Later on, another trial (phase IIIa) was made at a discharge rate of 0.8 mgd for a matching time period (April 5th – 11th). In this case, the average discharge amounted to 803,000 gpd. Since water was applied continuously, leakage was not a factor.

Phase IV: This phase commenced on March 30th after a four-day dry-out period. Effluent was discharged in batch mode twice a day. The first application was made early in the morning (~ 7:00 a.m.), and the second later in the afternoon (~ 3:00 p.m.) when the water level in the ponds had dropped off. An average of about 0.5 mgd was achieved for the first two days. However, capacity had dropped to 200,000gallons by the fifth day. In addition, the infiltration rate dropped to half its initial value within this period. Figure B-11 shows the change in infiltration rate with time single and double batch loading. While the single-batch rate reaches and maintains a constant, the double-batch rate keeps dropping. This is likely as a result of the soil being overwhelmed by this loading method.

Residual water level in the pond 13 was 30 inches the day after the first application. By the fifth day, this mode had lost its feasibility and was discontinued. Leakage constituted 4-60% of water discharged.

The capacity values for all phases were calculated using the flow graphs obtained from the SDCADA system.

Discussion

Table B-2 and Figure B-12 give a summary of results from the different phases. It is evident that a low continuous loading rate yields a greater disposal capacity than high batch loading, and would therefore be the operating mode of choice. Also there is more control in the amount of water that could be sent down to the ponds. While leakage increases the overall amount of discharge to the site, during batch loading, it is unreliable and uncontrollable (dependent on the water level and pumping interruptions). It also reduces the amount of water that can be discharged, as a batch, at any given time. The continuous loading method has the added benefit of being the least labor intensive - requiring only an occasional check on the site, and no additional hours for the operating staff.

While there was no build up of water in the ponds during the continuous-flow phase, it is likely that this would have happened eventually. However, the time to this occurrence is expected to be later than it was for the other phases; allowing for longer operating cycles between dry-out periods.

Update:

On April 12th, the end of pond 13 was washed out as a result of a high-volume accidental discharge. It was reconstructed on the following day, and. from April 15th to the 26th; effluent discharged has been maintained at 550,000 – 750,000 gallons per day. The lower value is as a result of increased pumping of effluent to the reservoirs resulting in a water level drop in the effluent pond.

Figure B-1: Percolation ponds in their original condition when constructed in 1977:

Figure B-2: Schematic of Forced Channel and Flow path of Tapias' Effluent

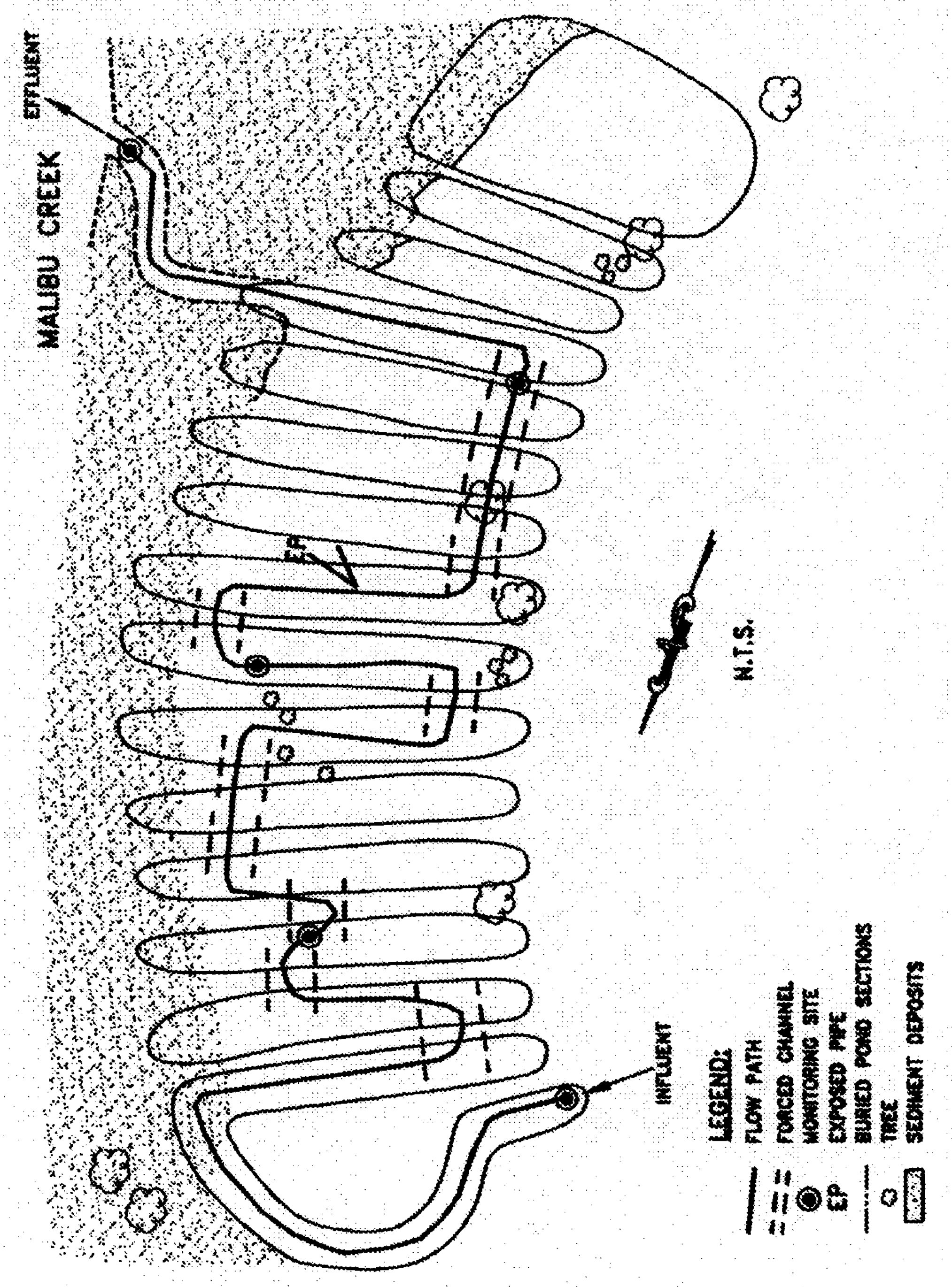


Figure B-3: Pond 1 - Inflow of Tapias' Tertiary Effluent

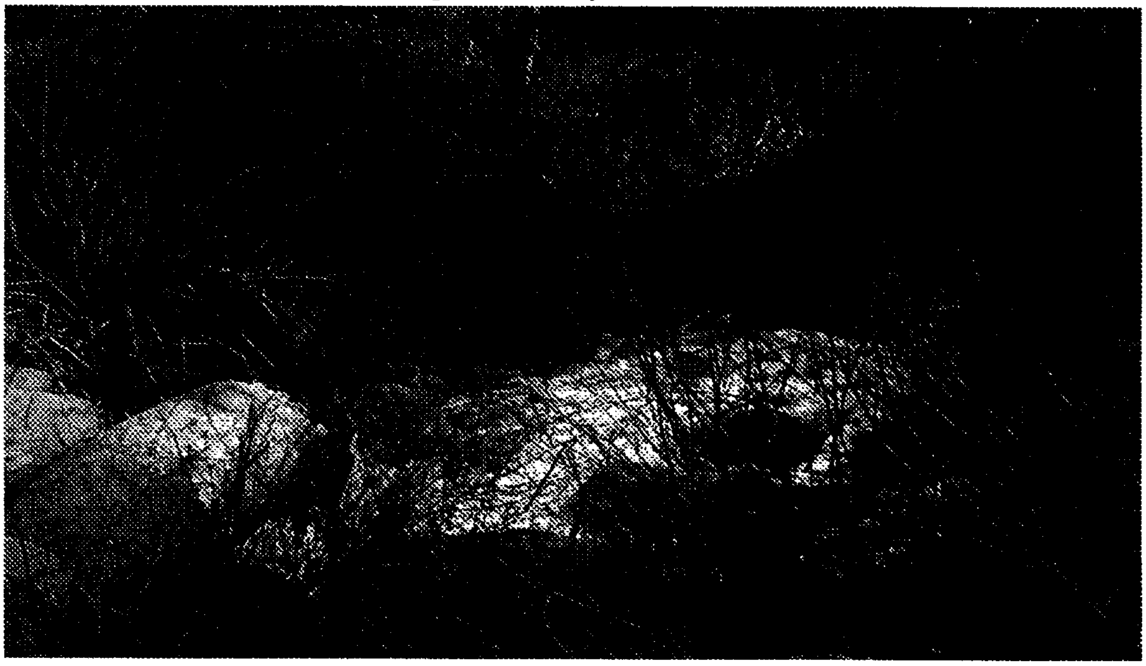


Figure B-4 East end of Pond 13- Empty

Figure B-5: Pond 13- Final Water Level after Flow Shut-off

Figure B-6: Methodology

	Phase I	Phase II	Phase III	Phase IV
Application:	Single batch	Semi-continuous	Continuous	Double batch
Duration:	2-3hrs	11.5 - 12 hrs	24hrs	varied
Loading Rate:	2.9-3.9 MGD	1MGD	0.6 - 0.7 MGD	2.5 - 3.5 MGD

Figure B-7: Leakage through drop gate of Effluent Pond.

Figure B-8: Phase I - Single batch application

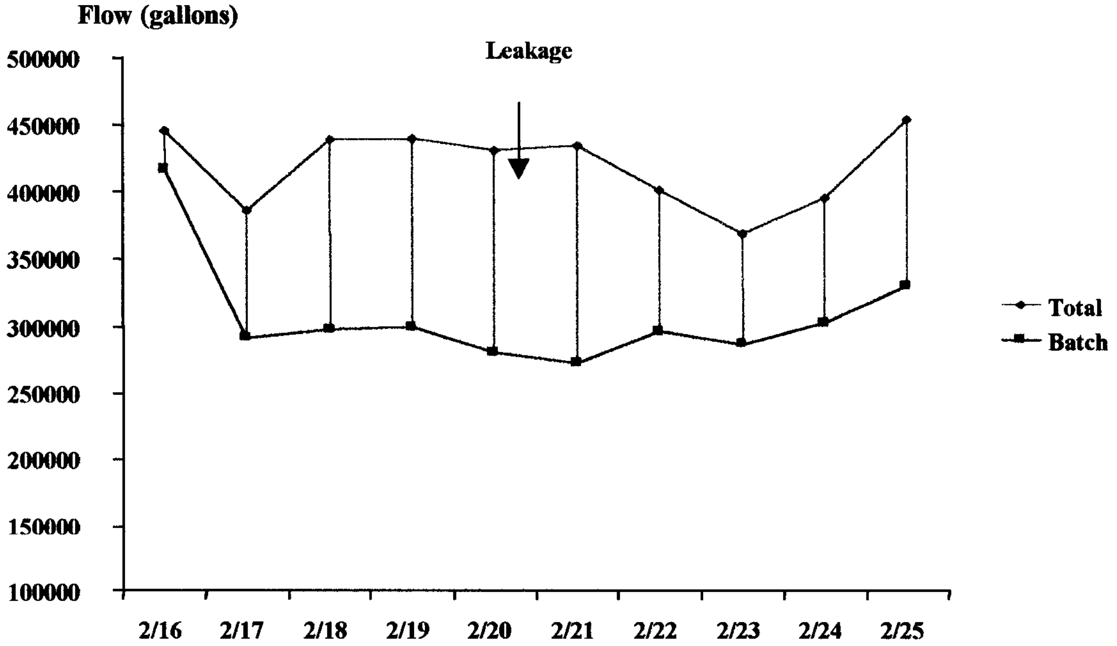


Figure B-9: Effect of Leakage of Effluent on Capacity of Ponds

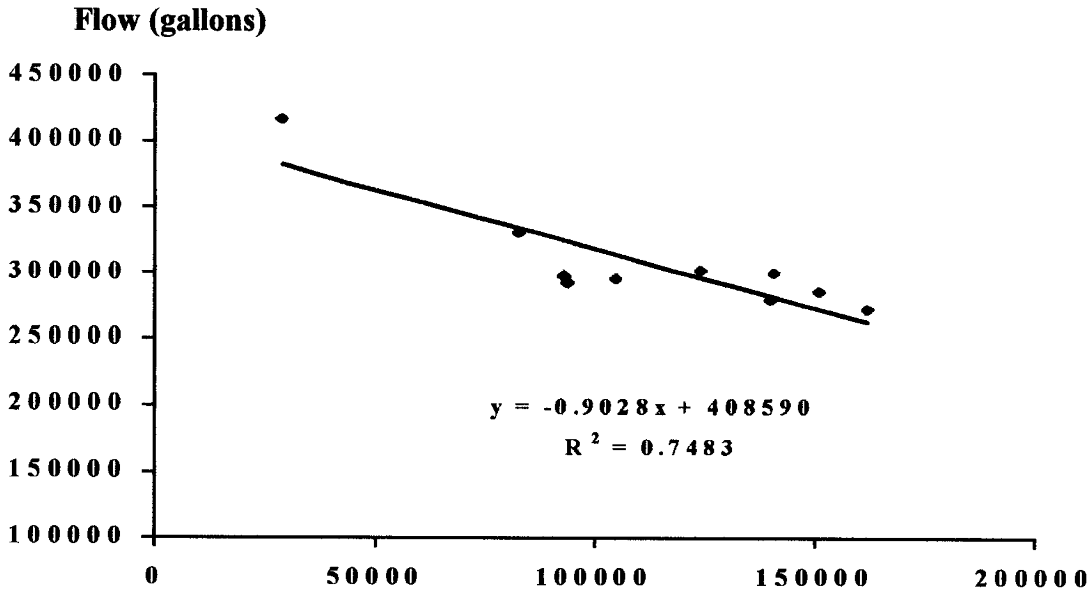


Figure B-10: Phase II – 12hr Continuous Flow

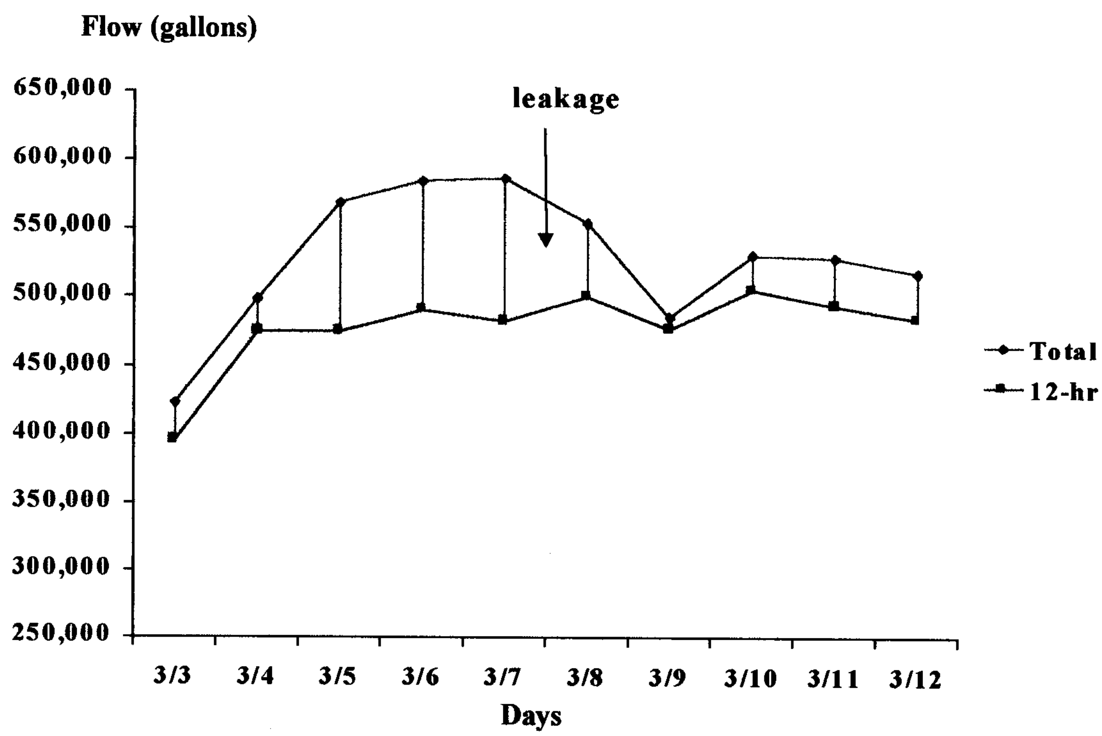


Figure B-11: Change of Infiltration Rate over Time

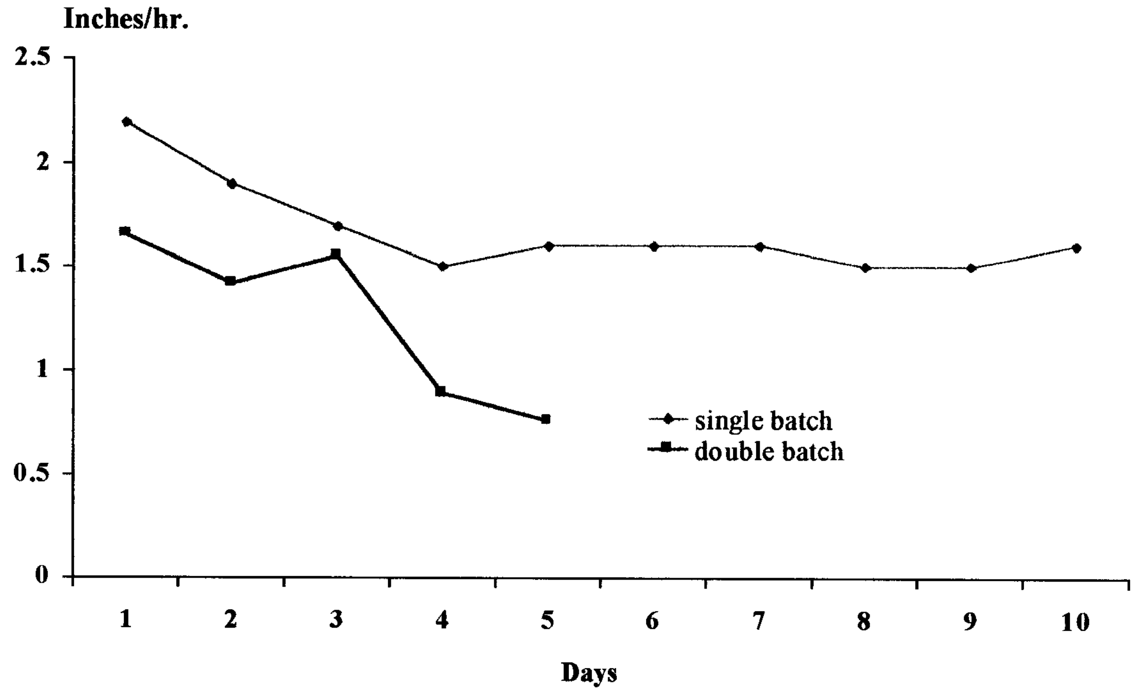


Figure B-12: Summary of effluent capacities for different phases

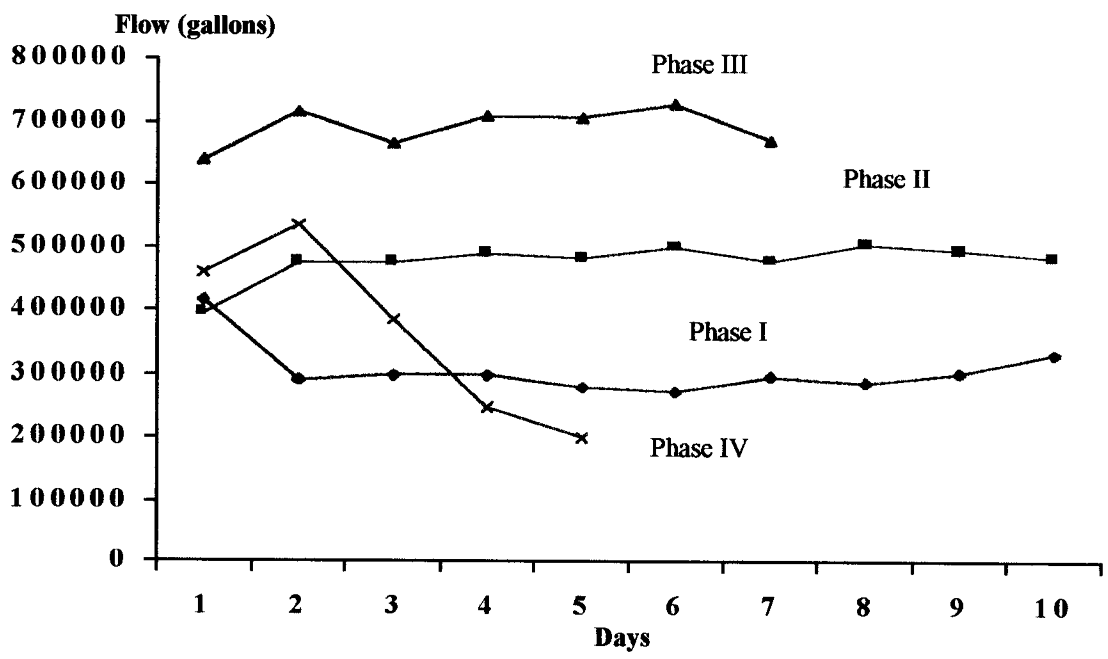


Table B-1: Summary of Results of Percolation Pond Capacity Monitoring – Phase I

Date	Temp (min) degC	Temp (max) degC	Rain (in.)	Total Flow - 10^3 gals	Batch Flow - 10^3 gals	Leakage 10^3 gals	Infiltration (in/hr)	Water Level (in.)
2/16	44	60	0	446	417	29	2.15	0
2/17	46	60	"	386	292	94	1.9	0
2/18	49	66	"	439	298	141	1.7	7
2/19	46	56	"	440	300	140	1.5	11
2/20	46	60	"	431	280	151	1.6	15
2/21	48	61	"	434	272	162	1.6	14
2/22	48	62	"	401	296	105	1.6	13.5
2/23	46	60	"	369	286	83	1.5	14.5
2/24	58	70	"	395	302	93	1.5	15.5
2/25	48	70	"	454	330	124	1.6	14.75

Table B-2: Summary of Results from the different Phases

Parameter	Phase I	Phase II	Phase III	Phase IIIa	Phase IV
Av. controlled discharge (cd) - gals	307,300	477,300	687,286	803,429	365,800
Standard deviation (cd)	41,532	30,310	29,182	31,251	140,830
Total discharge -gals	419,500	527,300	687,286	803,429	443,600
Average leak - gals	112,200	500,000	-	-	128,911
Standard deviation (leak)	39,796	34,641	-	-	37,028
% leak (of total discharge)	27	9.5	-	-	18
Extra operating hrs	none	3 hrs	none	none	none
No. of site visits per day	2	2	<1	<1	3
Water level build up	moderate	slow	none	none	fast