Central
Coast
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Studies

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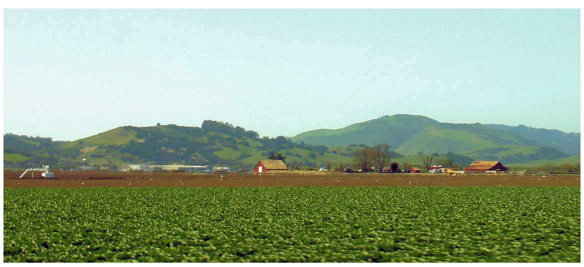
Successful treatment systems for removing nitrate concentrations from agricultural runoff



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Executive Summary

The goal of this project was to document practices to reduce nitrate concentrations from agricultural runoff and help growers comply with the Porter-Cologne Act while maintaining successful farming practices. As part of a graduate course, six students set out to review existing literature documenting the success of nitrate removal treatment systems that might be appropriate in an agricultural setting. Section 3 of this report describes the effectiveness of six treatment methods for reducing nitrate. Appendix A summarizes key information regarding load reduction potentials, constraints, and expected costs.

Documented health threats associated with nitrate contamination caused by agricultural runoff have prompted regulatory action requiring farmers to demonstrate discharge is not causing or contributing to contaminant exceedances in drinking water. But due to the lack of available knowledge, implementing nitrate reduction practices may incur high costs without delivering the desired results. Additionally, these costs for cleanup may be unnecessarily compounded, putting growers out of business. To help close this knowledge gap, several sources were investigated and six methods were selected to discuss their potential for reducing nitrate concentrations in agricultural runoff. For each method, discharge measurements, influent and effluent concentrations, system specifications, byproducts, approximate costs, benefits and disadvantages were recorded to help provide an overview of the research that has been conducted in this field and give growers a starting point when selecting the best approach for their farming practices.

Overall, a consensus was reached that research applying nitrate reduction methods to Central Coast farming conditions is enormously inadequate. Stakeholders will have to combine resources - including those not peer-reviewed - to develop a body of applied science documentation to help growers achieve realistic and effective goals. Not surprisingly, there was also a sense that we may be limiting ourselves if we expect to find one simple solution to solve a problem caused by many factors working together within a complex system. Within the next year, new expectations and deadlines for regulatory compliance will be established and growers will need to implement measures. Initiating local field-based studies will help examine the advantages and potential setbacks described in this report. We recommend exploring practices that can impede the flow of discharge and create optimum denitrification conditions. Additionally, since so many fields in this region already have artificial subsurface drainage networks, utilizing controlled drainage methods with denitrification bioreactors may be one of the most cost effective and efficient approaches as long as climate conditions are favorable and growers can prevent problems such as clogging and excessive moisture conditions that can harm the crop.

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Disclaimer:
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1 Introduction

To compete in the food market, growers rely on the application of fertilizers that support plant growth, but increased fertilizer use over the years has amplified the transport of nitrate from agricultural fields to surrounding water bodies. Nationwide surveys have detected nitrate levels in water below agricultural fields to be in the range of 5 to 100 mg/L, with regular sampling between 20 to 40 mg/L. For several years, studies of drainage flows in Illinois have shown more than 50% of nitrate found in surface waters surrounding cultivated fields was from added fertilizer not used by the crop (Bower 1990). High nitrate concentrations accelerate eutrophication causing loss of habitat and killing wildlife; in fact, nitrogen loading from cropland draining into the Mississippi River is known to be the primary source for hypoxia in the Gulf of Mexico (Carpenter et al. 1998, Goolsby and Battaglin 2000).

Human health concerns associated with nitrate contaminated water have also been extensively documented. During digestion, humans reduce nitrate to nitrite which can be fatal, particularly in infants, because it diminishes the transfer of oxygen in the blood (Malberg et al. 1978). Due to these health concerns, the USEPA established the maximum contaminant level for nitrate in drinking water as 10 mg/L; however, plans to reduce nonpoint source water pollution have not been successful (USEPA 2009, 2010). Row crop growers, UC Cooperative Extension, Resource Conservation Districts, and other researchers have demonstrated some modest nitrate reduction practices, which are implementable on farms in this region. The majority of the research we found that directly addresses practices for reducing waste discharge from agricultural fields has been conducted in Midwestern and Eastern United States where environmental conditions differ considerably from the Central Coast. In our region, semiarid climate conditions, perennial cropping systems, complex irrigation schedules, saltwater intrusion, large crop varieties, soils with low organic content and variable drainage properties, and economic and sociopolitical constraints together have complicated cultivation practices, making contaminant reduction a big challenge.

Continued location specific research to gain an in-depth understanding of the basin-scale mechanisms that control the fate and transport of nitrate to both surface and groundwater is necessary. When evaluating contamination, it is important to consider the cumulative effects from all sources within a watershed and the time period of transport to vulnerable water bodies. While denitrification rates vary, in general, groundwater in recharge areas with well-drained soils is vulnerable to nitrate contamination, and the use of artificial subsurface drainage systems in areas with poorly drained soils can accelerate nitrate transport to surface waters (Gilliam et al. 1978, Los Huertos et al. 2001). However, in conditions of limited denitrification, the transport of contaminants may be even faster than previous studies have shown because basin spatial variability has been oversimplified in these studies. To fully understand the leaching potential of agricultural chemicals, basin-scale heterogeneity must be considered. But while some researchers are working towards building models that can simulate these systems, accurately predicting denitrification rates in large fields is difficult because variation in these rates exists at small scales (Gilliam et al. 1978, Thomas et al. 1995, Carle et al. 2006). Since soil profile characteristics that delay water flow promote high denitrification rates, some argue that we should explore ways to impede flow while boosting denitrification (Christensen et al. 1967, Gilliam et al. 1978).

Regulatory constraints require that irrigation drainage management should satisfy both agricultural objectives as well as protect water resources. Due to the increasing costs of land and resources combined with a lack of available research and education about the efficiency of nitrate reduction methods, growers can be faced with having to implement costly regulatory practices that may not deliver the required results. Discovering low cost, effective best management practices (BMPs) to decrease nitrate levels is key for maintaining viable production while ensuring clean water.

In this report, six treatment systems are reviewed for their efficiency in reducing nitrate concentrations: electrodialysis (ED), reverse osmosis, bioretention, denitrifying woodchip bioreactors, bioretention with controlled drainage, and treatment wetlands. In addition to reading peer-reviewed studies that have been directly applied in an agricultural setting, we decided to also explore methods used in other applications that may have the potential to successfully reduce nitrate loads in runoff. Farmers need to discover new tools that will help them improve soil and water conservation, and cutting pollutants while maximizing yields may involve “thinking outside the box.”

ED and reverse osmosis have been primarily used for desalination purposes and in wastewater treatment. Biorentention systems have traditionally been used to mitigate urban storm water runoff, but these applications (including woodchip bioreactors) are now being used as denitrification methods. Drainage system design and operation modifications have been widely applied to agricultural practices, and more recently, the practice of woodchip filled trenches installed underneath controlled drainage structures have shown to be effective for decreasing nitrate concentrations in agricultural runoff. There is a lot of interest and research in treatment wetlands. Both surface and subsurface systems are designed to function like natural wetlands. They are very effective for filtering sediments and other contaminants from field discharges, but they also make available aesthetically appealing landscapes that provide habitat for natural wildlife.

2 Environmental and Regulatory Context

The federal Clean Water Act and state Porter-Cologne Water Quality Control Act define and regulate the quality of the waters of the state. As defined by the regional board basin plans, beneficial uses have been defined for all waters of the state. When the beneficial uses are not met, the water body is considered impaired and regulatory actions aimed to improve water quality are established. The causes of impairment vary with water body, but nutrients, pesticides, and sediment are the most common in central California. In particular, nitrate-N concentrations in freshwater rivers and creeks often exceed safe levels in many parts of the Central Coast region and irrigated agriculture is considered a primary source of elevated nitrate (Briggs et al. 2006).

The Salinas and Pajaro River Watersheds culminate 5,900 square miles and include over 250,000 acres of irrigated agriculture. These watersheds are home to the Elkhorn Slough and the Watsonville Sloughs complex, two of the most important wetland habitats in California. Additionally, the Salinas and Pajaro make up 79% of the land area draining into the Monterey Bay National Marine Sanctuary (MBNMS), the largest Marine Protected Area in the United States. The MBNMS boasts the greatest biodiversity in the temperate regions of the world, including more than 50 plant and animal species on government special status lists (MBNMS Action Plan IV: Agricultural and Rural Lands, 1999). The region's rich soils and year round mild coastal climate also sustain the intensively farmed valleys throughout the Salinas and Pajaro Watersheds, which generate over two billion dollars annually in agricultural products.

According to the Central Coast Regional Water Quality Control Board's Basin Plan, the Pajaro and Salinas River Watersheds support over 20 beneficial uses, including the following directly related to agricultural sources of pollution: Agricultural Supply (AGR), Ground Water Recharge (GWR), Freshwater Replenishment (FRSH), Water Contact Recreation (REC 1), Non Contact Water Recreation (REC 2), Commercial and Sport Fishing (COMM), Warm Fresh Water Habitat (WARM), Cold Fresh Water Habitat (COLD), Inland Saline Water Habitat (SAL), Estuarine Habitat (EST), Marine Habitat (MAR), Wildlife Habitat (WILD), Preservation of Biological Habitats Special Significance (BIOL), Rare, Threatened, or Endangered Species (RARE), Migration of Aquatic Organisms (MIGR), Spawning, Reproduction, and/or Early Development (SPWN) (Briggs et al. 2006).

Both watersheds are severely impaired for nutrients, sediments, and pesticides. In the Pajaro River, nitrate concentrations are typically 4-7 times higher than the drinking water standard and observed concentrations of nitrogen in agricultural drainage ditches account for more than 15% of the total stream nitrogen load during summer and fall low flow conditions throughout the Watershed (Los Huertos et al. 2001). In addition, elevated concentrations of nitrate (5 to 60 mg/L) and ortho-phosphate (0.5 to 2 mg/L) are found in numerous water bodies in the Salinas River Watershed (Anderson et al. 2003).

Agricultural sources of sediment, nutrients, and pesticides play a significant source of impairment for 29 TMDLs adopted or in development throughout the watersheds (CCRWQCB 2008). These TMDLs, adopted or in development, require the adoption of BMPs for pollution reduction and attainment of water quality standards. Additionally, the adoption of BMPs is critical for the successful implementation of the Agricultural Waiver.

The State Water Resources Control Board is responsible for protecting and restoring water quality by delegating authority to nine regional boards. The central coast region (Region 3) developed a conditional waiver of waste discharge requirements for irrigated lands to address water quality issues from farming. The waiver required irrigated agricultural dischargers (owners and operators) to enroll in a monitoring and reporting program to control waste discharges. The waiver expired in 2008, but was extended to 2011 because the renewal process has been highly contested (CCRWQCB 2010a).

During a public workshop in July 2010, the biggest objection from growers and their advocates was that staff recommendations had not been tailored to the specific needs of individual growers. The success of a plan for reducing nitrate levels depended on a variety of combined factors like the size and location of fields, the types of crops grown, cultivation and irrigation practices, whether a grower had already implemented nitrate reduction methods or not. Growers were concerned that the staff did not understand their unique circumstances and new regulation would put them out of business.

On November 19, 2010, the staff proposed to renew the current conditional waiver with some revisions. To address water quality problems while considering the issues raised by the public, the staff established three tiers of conditions based on the size of the farm, proximity to an impaired water body, the use of chemicals, and the type of crops cultivated. The tiered system is designed to require growers with the highest threat the greatest amount of discharge control, monitoring and reporting requirements, while those with the lowest threat will have the least amount of requirements. Compliance dates for demonstrating discharge will not cause or contribute to exceedances in drinking water standards vary between 2 to 10 years depending on the tier. Public comment on the draft order, the monitoring and reporting program and the subsequent environmental impact report must be submitted by January 3, 2011. And on March 17, 2011, the Water Board will hold a public meeting to consider the staff's recommendations and adoption of the renewed waiver (CCRWQCB 2010b).

3 Practices

3.1 Electrodialysis (ED) Method

Electrodialysis (ED) is a membrane process in which ions are removed from an aqueous solution using an electric current (Hell et al. 1998). It has been utilized for over 50 years as a way of purifying brackish waters for potable drinking water (Strathmann 2010). Today, ED is used in a host of applications including, but not limited to, municipal and industrial wastewater treatment and desalination.

Water to be treated is pumped through a series of alternating anionic (negatively charged) and cationic (positively charged) membranes organized in stacks. The membranes are fixed between an anode and cathode, and when electricity is applied, negative ions are moved towards the anode and positive ions are moved towards the cathode. The movement of ions is inhibited by similar charged membranes and clean water is separated from concentrated brine. The brine concentration varies depending on influent concentrations as well as membrane size and specificity.

In 1990 the Austrian Energy and Environment Group (now AE & E) commissioned a test plant to remove nitrate from ground water with the use of ED. The pilot plant was an Electrodialysis Reversal (EDR) system with two stacks in series and the results of the study were recorded in Hell et al. (1998). An EDR system was chosen over biological reduction because physical systems can be shutdown, arguably making them cheaper for seasonal operations. The hydraulic capacity of the plant was 1 $m^3\ h^{-1}$ (4.4 gpm) and the dimensions of the membranes were 0.5×0.1 m (1.6×0.3 ft). Nitrate (NO_3–N) influents of 15.4 mg NO_3/L were reduced to 5.9 mg NO_3/L with one stack in operation. For higher influent concentrations of 28.4 mg NO_3/L two stacks were required for similar results. Based on the conclusions of the pilot plant, the Austrian Energy and Environment Group opened a large scale ED plant in 1996. The plant was designed for a hydraulic loading of 144 $m^3\ h^{-1}$ (634 gpm). Nitrate reductions of 28.4 mg $N0_3$/L to 5.9 mg $N0_3$/L were possible, but because of client requirements the effluent was only lowered to 9.5 mg $N0_3$/L, a 66% reduction (Hell et al. 1998). Brine concentrations were found to be 2820 μS/cm, with major ion concentrations: 211 mg NO_3/L, 15 mg Na/L, 4.6 mg K/L, 270 mg Cl/L, 433 mg Ca/L and 124 mg Mg/L (Hell et al. 1998).

General Electric Power and Water is one of many companies that offer ED technology and infrastructure. In a technical paper GE reported that one of their industrial ED plant had NO_3–N removal rates of 80.4% (Prato and Parent 1993). The plant utilized two Aquamite X ED units and processed water at a rate of 15.8 $m^3\ h^{-1}$ (70 gpm). Nitrate reductions of 155 mg NO_3/L to 30 mg NO_3/L were recorded. Other ions such as sodium, calcium, magnesium, chloride, bicarbonate, sulfate and nitrite were also removed to varying degrees. Total Dissolved Solids were reduced markedly however, from 1753 mg/L to 534 mg/L, a 70% reduction.

ED has many advantages. Plants can be easily shutdown and restarted, the process can be fine tuned for specific ion removal, ED has a low chemical demand, and in addition to nitrate mitigation a reduction in water hardness can be achieved (Hell et al. 1998). Prato and Parent (1993) cited both self-cleaning membranes, and low operating and maintenance costs as other advantages. Disadvantages of ED include the price of infrastructure and the highly concentrated brine byproduct. Hell et al. (1998) proposed the brine be recycled and used as irrigation waters for highly tolerant crops, or that the brine be sent to waste water treatment facilities for denitrification.

3.2 Reverse osmosis method

Reverse osmosis can separate inorganic or organic constituents by forcing fluid through a semi-permeable membrane (Sourirajan and Agrawal 1969). Reverse osmosis is a transport process where permeation across a membrane occurs because of a concentration gradient and a pressurized system (Lee 1975). Although most methods of reverse osmosis have been developed for desalination, (Brian 1965, Fritzmann et al. 2006, Greenlee et al. 2009), the reverse osmosis process can also effectively remove NO_3-N from water (Schoeman and Steyn 2003).

Schoeman and Steyn (2003) used a reverse osmosis system in rural South Africa to remove nitrogen and desalinate ground water. Water was pumped into three 10,000 L (2,650 gal) tanks (feed tanks). The feed water was filtered through three sand filters, treated with sulfuric acid and an anti-scalant and then filtered again through a 5 μm filter. The water was then pumped through a reverse osmosis unit and finally treated with caustic soda to adjust the pH level. The resulting permeate was desalinated potable water with a low NO_3-N concentration and a concentrated brine waste (Schoeman and Steyn 2003).

At a rate of 16 gpm, the reverse osmosis facility was able to reduce the NO_3-N concentration of the influent from 42.5 mg/L to 0.9 mg/L. This method removed 0.0025 kg/day (0.0056 lbs/day) from 100,000 L/day (26,417 gal/day) of water. If operated at a larger scale, the reverse osmosis facility could remove 22.69 kg/day from 545,000 L/day (144,000 gal/day) at a rate of 6.31 L/s (100 gpm). The plant output was approximately 55,000 L/day (14,529 gal/day) of potable water. Roughly 50% of the feed water was discharged as a brine solution. The brine solution had a NO_3-N concentration of 69.17 mg/L. Low water recovery (50% as brine) produced wastewater that was dilute enough to be used as a water supply for animals. If concentrated brine is produced, it could be sent to a sewage treatment plant, or used for field irrigation/fertilization. Reuse of brine solutions is dependent on brine quality, nitrate concentration, chloride concentration, sodium adsorption ratio (SAR) and residual sodium carbonate (RSC) (Schoeman and Steyn 2003). Further research is needed on acceptable brine quality and its disposal or applications.

The reverse osmosis facility built by Schoeman and Steyn (2003) cost \$29,000 and \$0.0005/L (\$0.003/gal) to operate (2002 US dollars). The highest portions of the operating cost were electricity and the membranes. Reverse osmosis uses 3.5–7.5 kWh of electricity per 1000 L (254 gal) of water (Reverse Osmosis 2009). The cost of the membrane is dependent on size and production volume. A cost estimate for membranes is $0.05–$0.44/1000 L (254 gal) (2003 US dollars) (Avlonitis et al. 2003). Biofouling, scaling and membrane degradation can reduce the efficiency and lifespan of membranes (Avlonitis el al 2003). Maintenance and effective filtering would extend the lifespan of membranes and efficient/alternative energy use could minimize electricity costs. The size of the reverse osmosis facility is primarily dictated by the size of the holding tanks for water. Schoeman and Steyn (2003) pumped groundwater into small feed tanks and the permeate was stored in a 160,000 L (42,268 gallon) reservoir. The size of the facility could be variable if runoff water was captured, stored and then pumped into the system. Brine use/disposal and permeate use/storage could also dictate the size of the reverse osmosis facility. A creative solution, unique to individual agricultural applications, would need to be created to use, store and process the maximum amount of water with minimal storage.

3.3 Bioretention

Bioretention systems have been conventionally utilized to mitigate urban storm water runoff. Bioretention systems are small areas of land specifically redesigned or 'landscaped' to maximize infiltration and nutrient uptake. Recently a number of researchers have begun to investigate the feasibility of utilizing bioretention systems to treat agricultural runoff (Davis et al. 2001, Hsieh and Davis 2005, Henderson et al. 2007, Hsieh et al. 2007, Siegel 2008).

The systems are constructed by excavating a small patch of land, refilling it with a variety of media layers and planting it with native vegetation. Runoff entering the bioretention system must first pass through a vegetated buffer zone before collecting in a ponding area. As the water infiltrates through the vegetated soil, mulch and sand layers, the contaminants are treated through the physical and biological processes of evapotranspiration, plant uptake, biodegradation, filtration and adsorption. Effluent is then channeled into a drainage pipe or allowed to percolate naturally into the surrounding soil (Siegel 2008).

Bioretention systems are effective at reducing total suspended solids, organic matter, copper, lead, zinc, oil, phosphorus, ammonia, total Kjeldahl nitrogen (TKN) and ammonium (Davis et al. 2001, 2006). These systems commonly have low nitrate removal rates (<20%). However, introducing a saturated zone at the base of a traditional bioretention system could create an anoxic zone which fosters microbial denitrification. As the runoff travels through the saturated denitrification region it is supplied with a donor electron and reduced to nitrogen gas by anoxic heterotrophic or autotrophic denitrifying bacteria (Siegel 2008). Modified bioretention systems utilize an impermeable layer to prevent water from exiting through the bottom of the system, thus maintaining anoxic conditions required for bacterial denitrification (Figure 1). Shredded newspaper as an additional media, and observed nitrate mass removal rates of 4.83 ppm in column reactor studies (Davis 2007).

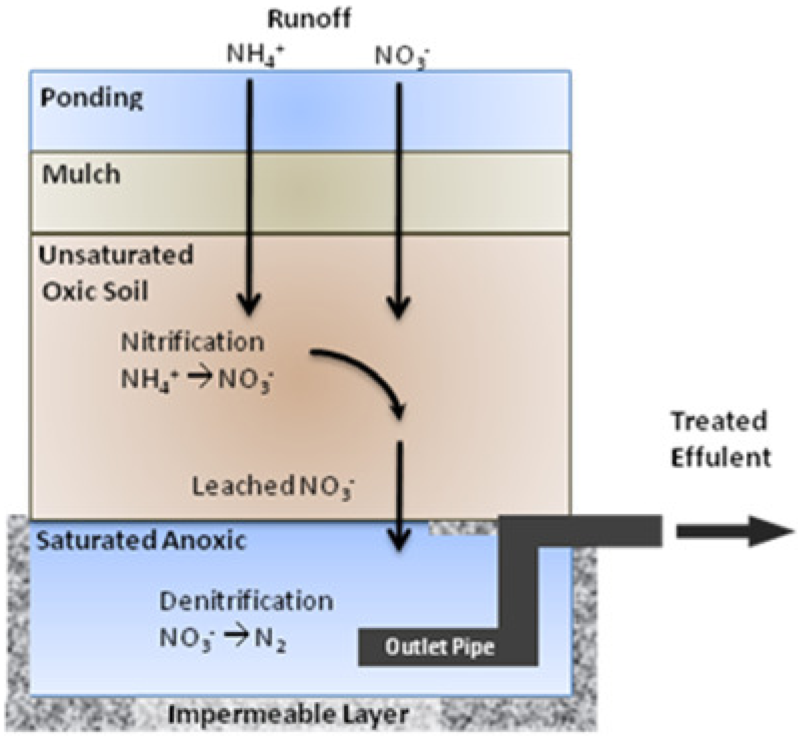

Figure 1. Diagram of bioretention cell modified for denitrification (Siegel 2008).

While laboratory results for nitrate removal in bioretention systems are promising, few field studies have been able to demonstrate a nitrate reduction (Davis 2007, Roy-Poirier et al. 2010). Carpenter and Hallam (Robertson et al. 2000, 2009) demonstrated the development of preferential flow paths within media of a bioretention cell/unit can severely retard nitrate removal efficiency. Development of standardized construction practices may be able to eliminate variation in nitrate removal results and enable lab results to be replicated in the field.

3.4 Denitrifying Bioreactors

Wood chip bioreactors rely on heterotrophic denitrification to remove nitrate from drainage water, the same natural process at work in wastewater treatment ponds and wetlands. Heterotrophic denitrification is a microbial process, which permanently removes nitrogen by converting nitrate to nitrogen gases using a carbon source. Bioreactors are box structures (Robertson et al. 2000) or trenches filled with woodchips and may be installed with tile drainage to intercept concentrated discharge or as denitrification walls, intercepting shallow groundwater.

Robertson et al. (2000) conducted agricultural field trials from 1993 to 2000 in Ontario, Canada, to treat nitrate in ground water. Field trials reduced influent NO_3 concentrations of 14 ppm to an average of 8 ppm, with complete nitrate removal occurring often. Denitrification rates were found to be temperature dependent, ranging from 5 ppm/d at temperatures of 2℃ to 5℃ to about 15 to 30 ppm/d at 10℃ to 20℃. However, influent concentrations and treatment volumes were low compared to typical Central Coast agricultural drainage concentrations and volumes. Results from septic tank field trials demonstrated the potential for wood chip bioreactors to reduce nitrate levels of 57 ppm to an average of 11 ppm.

Robertson (2000) assessed the lifespan of four samples of woodchips in the laboratory; two were fresh and two had been in continuous operation in subsurface denitrifying bioreactors for periods of 2 years and 7 years. Results indicated woodchips lost about 50% of their reactivity during their first year of operation, yet 2- and 7-year-old media maintained similar nitrate removal rates.

Greenan et al. (2009) conducted a laboratory study to determine the rate of nitrate removal by wood chip bioreactors under a range of flow rates similar to those of tile drained corn fields in Iowa. Nitrate was added at 50 ppm and mean nitrate concentrations in the effluent were found to be 0.0, 18.5, 24.2, and 35.3 ppm for the designated flow rates 2.9, 6.6, 8.7 and 13.6 cm/d respectively (corresponding to 100, 64, 52, and 30% efficiency of removal). It is clear that nitrate removal decreases as flow rates increases.

Woodchip bioreactors can be applied as 'end of pipe' treatments for reducing nitrate in drainage water. Bioreactor demonstration projects in Minnesota illustrate this approach (CSD 2010). For example, a woodchip bioreactor consisting of a trench filled with woodchips, 1.8m deep, 0.8m wide, 47m long, was installed at the edge of a farm field to treat drainage water from 27 acres. Diversion structures channel the drainage water through the wood chips and regulate how fast the water flows through the bioreactor. Bioreactors are also being used to help protect municipal water supply from nitrate contamination in St. Peter, Minnesota. Two wood chip bioreactors were installed at drain tile outlets located within the city's wellhead protection area (CSD 2010).

Schipper at al. (2010) estimated cost effectiveness of denitrifying bioreactors using a Canadian example described by Robertson et al. (2000). The bioreactor was 1.1m deep, 1.2m wide, 13m long, and removed an average of 11.3 kg N/year from an agricultural field drain. Assuming a conservative 20-year life expectancy for this bioreactor, cost of removal was estimated at US$ 2.39-15.17 kg-N. This compared favorably with estimates of other nitrate-N removal technologies such as treatment wetlands ($3.26/kg-N) and cover crops ($11.06/kg-N).

3.5 Biorentention with controlled drainage

The type of crops cultivated, the amount of fertilizer applied and tile system design have a great impact on the amount of nitrate present; however, soil profile characteristics that influence the rate of denitrification combined with drainage volume are believed to have the greatest influence on the amount of nitrate in water below the root zone and leaving the field through subsurface drainage networks (Gilliam et al. 1978, Fisher et al. 1999, Strock et al. 2007). By managing the watertable while enhancing denitrification, growers can maximize the efficiency of agricultural nitrate removal practices from tile drainage effluent water while maintaining profitable yields. Drainage control has been extremely effective in reducing nitrate export through tile lines by decreasing effluent volume (Gilliam et al. 1979, Thomas et al. 1995, Skaggs et al. 2005, Evans et al. 2007). At the same time, high concentrations of NO_3-N have been successfully treated in bioreactor beds (Blowes et al. 1994, Chun et al. 2009, 2010). Installing the woodchip filled trenches underneath controlled drainage structures can increase the efficiency of denitrifying bioreactors. Woli et al. (2010) concluded that drainage control structures combined with lined bioreactor beds have great potential for decreasing NO_3-N movement to streams due to enhanced denitrification.

Controlled drainage is the practice of varying water depth in the drainage outlet to reduce nitrate loads on tile-drained soils while maintaining sufficient drainage for successful crop production (Frankenberger et al. 2007). Drainage system design and operation modifications have been widely applied to agricultural practices in Eastern U.S. and Midwest corn and soybean cropping systems (Gilliam et al. 1979, Thomas et al. 1995, Fisher et al. 1999, Evans 2003, Evans and Skaggs 2004, Smeltz et al. 2005, Evans et al. 2007, Strock et al. 2007, Sui 2007, Skaggs and Youssef 2008).

Drainage occurs when the watertable rises, so to control water depth the drainage outlet is raised and lowered as illustrated in Figure 2 (Frankenberger et al. 2007). This method is appropriate for fields that require drainage, and runs more efficiently on patterned subsurface tile line systems rather than random designs. Each drainage control structure manages approximately 10-20 acres, and costs $500-$2000 USD depending on design and automation features. Growers should expect to also incur installation and structure management costs (Frankenberger et al. 2007).

Field studies have shown that if properly managed, water management practices do not harm crop yields; in fact, with sufficient rain, controlled drainage can improve production (Gilliam et al. 1979, Needelman et al. 2007, Strock et al. 2007). In a field study that compared controlled drainage with standard artificial subsurface drainage, the average corn yield was 19% greater and the average soybean yield was 64% greater because the water table management system conserved water for periods of low moisture and increased plant nitrate uptake. In the same study, the amount of NO_3-N potentially available to move via drainage channels to surface water bodies was reduced by 46% (Fisher et al. 1999). Reductions in nitrate load in drainage discharges have ranged between 15-75% depending on geographical location, climate, the type of soil, and cultivation practices (Gilliam et al. 1979, Thomas et al. 1995, Fisher et al. 1999, Evans 2003, Evans and Skaggs 2004, Smeltz et al. 2005, Evans et al. 2007, Frankenberger et al. 2007, Needelman et al. 2007, Strock et al. 2007, Sui 2007, Skaggs and Youssef 2008).

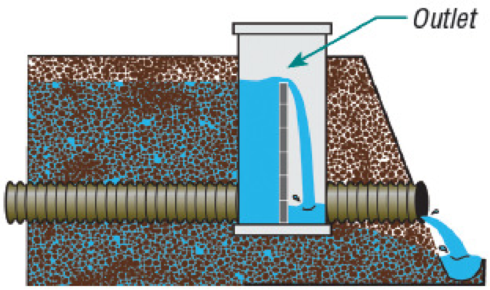

Figure 2a. The outlet is raised after harvest to reduce nitrate delivery.

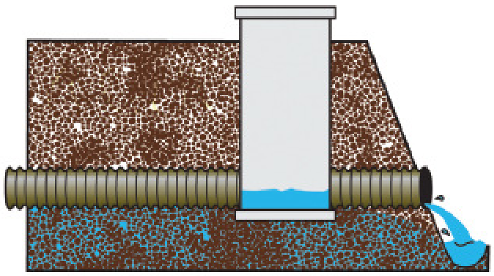

Figure 2b. The outlet is lowered a few weeks before planting and harvest to allow the field to drain more fully.

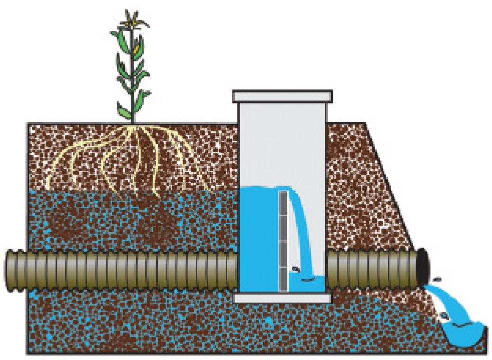

Figure 2c. The outlet is raised after planting to potentially store water for crops.

While drainage water management can cut pollutants in subsurface drainage flows, conserve water and boost yields all at a reasonably low cost with low maintenance, there are important considerations and limitations in the application of drainage control structures:

- Drainage control structures have not been widely applied and tested in perennial or winter annual cropping systems;
- The practice is suitable on fields with low permeability soil drainage properties;
- Flatter fields allow one control structure to maintain a large area; one structure can manage a field with 1 foot or less of elevation change; as the slope increases, more structures are needed and costs increase;
- Sufficient rainfall must occur for the water table to rise to the depth of the outlet setting;
- During the growing season there is a risk of excess water stress on the crop during prolonged wet periods;
- Narrow drain spacing may reduce the risk of yield loss during times of heavy rainfall;
- Yield benefits may not accrue in years when rainfall is not sufficient;
- Potential crop yield increases will be greater in regions where drains flow for long periods of time after planting (Frankenberger et al. 2007).

Water table management practices can work well in conjunction with other nitrate removal methods; in fact, drainage control structures should be viewed as one solution among many for a successful soil and water conservation project. The use of controlled drainage in conjunction with woodchip biofilters is being increasingly tested. Woli et al. (2010) compared free drainage (standard subsurface artificial drainage) and controlled drainage with bioreactors to study the methods' efficiency for reducing nitrate. The results indicated that cumulative N loads measured during a two year period reduced from 0.62 to 0.14 pounds per day (nearly 55%). While bioreactors were not installed in the free drainage field, and these results focused solely on controlled drainage associated with bioreactors, it should be noted that the free drainage system had much higher drainage loss compared to the controlled drainage system (from 10,885 to 2827 gallons per day - a 26% decrease in flow) which suggests more denitrification with the controlled drainage system. After installing flashboard riser-type water level control structures in outlet tile ditches to raise the water table during the winter, NO_3-N concentrations were cut in half with the use of bioreactor systems (Woli et al. 2010). These results indicate that managed drainage with denitrification beds has great potential for reducing tile nitrate export to both streams and groundwater. In combining these two practices, growers may have the opportunity to enhance denitrification, improve water quality, conserve water, and maximize profits through increased yields.

3.6 Treatment wetlands

Treatment wetlands are areas that have been engineered to utilize naturally occurring processes in wetlands to treat waste water (Vymazal 2007). There are multiple specific purposes of treatment wetlands, and multiple construction designs. Treatment wetlands can involve surface or subsurface flow. We summarized the basic designs and uses of each category, as well as a couple of relevant case studies in which treatment wetlands were used to reduce nitrate concentrations.

3.6.1 Subsurface Wetlands

Sub-surface flow constructed wetland systems have been used extensively in Europe and South Africa for nitrogen removal from domestic wastewater. Recent innovations in low impact development are incorporating sub-surface wetlands in water quality improvement and stormwater management plans. Designed to look and function as a natural wetland, sub-surface constructed wetlands effectively remove sediments and other pollutants found in run-off while providing a visually appealing landscape. They can be placed in existing irrigation ditches or dry ponds as part of a water quality improvement plan.

Sub-surface flow constructed wetland systems generally consist of a ditch or bed that has been sealed by an impermeable substance to block leakage and media to assist the growth of emergent plants (Lee et al. 2009). Typically, crushed rock, gravel, soils, sand or various combinations are used as media. Cattail (*Typha* spp.), reeds (*Phragmites* spp.), and Bulrush (*Scirpus* spp.) are examples of commonly used plants due to natural adaption to saturated conditions (Gaboutloeloe et al. 2009). Sub-surface flow systems can be further divided into horizontal and vertical flow systems according to the flow direction of the wastewater.

For complete nitrogen removal both nitrification and denitrification need to occur. Although horizontal sub-surface flow systems are most often used, vertical systems are increasingly utilized for waste water treatment. Recent research has looked at the feasibility of combining horizontal and vertical sub-surface flow systems to utilize the specific advantages of individual systems (Vymazal 2007). A multi-stage system study in Norway found a 73% reduction in nitrate-N (30ppm to 8 ppm) by regulating the vertical flow bed sizing and managing the carbon supply to enhance denitrification (Maehlum and Stlnacke 1999).

The cost of building a subsurface wetland varies from project to project and may not be financially viable for many sites. A technical paper on sub-surface gravel wetland construction identified costs in the realm of $22,000 per acre for a 5,400 ft^2 wetland designed to accommodate 3300 ft^3 of storm runoff (UNH 2010) . A sub-surface wetland project at Milan Army Ammunition Plant conducted by USAEC and Tennessee Valley Authority (TVA) estimated the capital costs of wetland treatment to be $0.0136/100 gal over a 10 year period. However, over a 30 year period, the cost was dramatically reduced to $0.0045/100 gal (Deuren et al. 2002).

Successful performance of subsurface wetlands depends on many factors such as the growth of macrophytes, wetland design, and operation and maintenance. Typical problems include: oxygen depletion, clogging by sludge sediment accumulation, destruction of wetland layers, clogging of the filter media, and short circuit flow (Lee et al. 2009).

3.6.2 Free water surface treatment wetlands

Surface water treatment wetlands (STWs), visually similar to natural marshes, are wetlands which contain areas of open water, aquatic macrophytes and microorganisms. It has been demonstrated in both the lab and the field that STWs can remove ammonia (NH^4) and nitrate (NO_3), as well as provide habitat for a wide variety of aquatic, terrestrial and avian wildlife (Vymazal 2007, Kadlec and Wallace 2009, Kadlec et al. 2010). Because of their ability to handle a large variation in inflow (i.e. storm runoff, seasonal variations in runoff) STWs have been an exclusive treatment system of choice for agricultural runoff (Kadlec and Wallace 2009) (Kadlec and Wallace 2009).

Often STWs are designed with sedimentation ponds upstream of the inlet. This reduces the amount of fine sediment settling in the pond and subsequently the maintenance of the STW (Kadlec and Wallace 2009). Through the physical deposition process, settling ponds can decrease the nutrient load in agricultural runoff (Beutel et al. 2009).

Laboratory and field tests of STWs all confirm the successful removal of nitrogen, but show a wide range of removal rates. Stewart et al. (2008) performed an experiment on a laboratory scale. The STW was designed in a 30 gal drum with a commercially available vegetated floating mat on the surface of the water, and a water pump recycling the water at a rate of 0.8 gal/ min. By holding the tank at 27 °C and periodically adding food grade molasses the tanks reduced the concentration of nitrate from 230 ppm to approximately 0 ppm in 48 hours. This was, however, a controlled laboratory experiment and field results are likely to vary widely. Placing the vegetated mats in a pond with the vegetated mats will more likely yield reductions of 759 mg/L per ft^2 of mat with aggressive circulation or 144 mg/day per square foot with moderate circulation (Stewart pers. comm. 2010).

Field tests of STWs have yielded large ranges in nitrogen removal abilities. Beutel et al. (2009)studied a 4 acre STW in the Yakima Basin, Washington state. This system was designed to treat large amounts (236gal/min) of elevated nitrate levels from agricultural runoff. Their system captured farm runoff and mixed it with river water to vastly decrease influent concentrations. Beutel et al. (2009) reported a nitrate reduction from 2.0 ppm to 0.2 ppm.

Kadlec et al. (2010) reported on two large STWs in Imperial Valley, CA. Similar to the Yakima Basin STW, these systems treated the combined runoff from many farms (3,090 and 417 gal/min, respectively) in two STWs which were 11.5 and 4.5 acres respectively. Nitrate reductions were observed to be from 2.55 to 1.84 ppm, and 0.95 to 0.77 ppm.

4 Summary and Concluding Remarks

According to the peer-reviewed studies summarized in the appendix, the potential for nitrate load reduction (normalized to 100 gpm) varied from 2 to 100 percent. The reverse osmosis method resulted in a very low load reduction (2%). The ED and subsurface treatment wetland studies showed potential for reducing nitrate loads (19% and 27% respectively) in high discharge, high concentration applications, but the costs associated with these methods may be too high for some growers. The surface treatment wetland method resulted in the highest reduction (100%), but this study was conducted in a highly controlled laboratory experiment. The nitrogen bioretention study and the first woodchip bioreactor study each yielded a 40% load reduction, but these applications involved low discharge and influent concentrations, and the former was in a laboratory setting. The second woodchip bioreactor study and the woodchips with controlled drainage method yielded 19% and 23% load reductions, but the former was a septic tank system application. Also, the results from these two studies may not be applicable to large-scale agricultural practices on the central coast due to considerable differences in soil moisture conditions.

A better understanding of complex hydrological processes and the technologies available can help design and implement improved water quality protection programs. We may discover the most efficient plans for reducing nitrate concentrations in our region will involve combining several methods that can reduce the application of chemicals, slow down and decrease discharge, and at the same time, enhance denitrification. While both ED and reverse osmosis have the potential for producing high quality results, large-scale operations and the required disposal of brine can be expensive in some agricultural settings. Depending on the quality of brine, the byproduct may be treated and reused for field irrigation and fertilization. Biorentention systems and treatment wetlands can be effective at removing a large variety of constituents, but both methods require allocating land for constructing the ponds which may also pose an economic challenge for growers.

On the other hand, denitrification bioreactors (woodchip bioreactors) may have the potential to provide Central Coast farmers with an effective, long term, passive treatment method to reduce nitrate from agricultural drainage. Low cost, ease of installation and relatively low maintenance are some of the benefits. If combined with already existing artificial subsurface systems, there is no need to take additional land out of production. It will be necessary to conduct further study to mitigate possible increased concentrations of dissolved organic carbon in receiving waters. And while there are research gaps associated with the maintenance and application using different drainage systems under varying moisture conditions and soil types, combining woodchip bioreactors with controlled drainage systems may serve the purpose of impeding flow while enhancing denitrification in fields with infiltration problems.

While many of these practices explored here have promise in treating agricultural runoff, on the ground field work is still needed to evaluate the efficiency of nitrate reduction using these methods. The studies summarized in this report focused on peer-reviewed articles, but this may have limited the search results since there are growers and their affiliated interest groups conducting field studies throughout the country that have been documented in technical reports and newsletters, but not yet peer-reviewed. Also, one third of the articles reviewed were not discussing approaches directly applicable to farming practices. A big part of the research involved laboratory rather than field-based studies, and the field-based studies exhibiting successful nitrate removal practices in agricultural settings were conducted outside of California. To determine applicability of these methods to local agricultural practices, high nitrate concentrations at high discharge rates need to be studied in local climate and soil conditions. Ideally, several pilot studies in the field could determine compatibility with local farming practices and economic sustainability.

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6 Appendix

Nitrate load reduction potential, constraints, and estimated costs

Parameters	Electrodialysis[A]	Reverse Osmosis[B]	Surface Treatment Wetland[C,D]	Subsurface Treatment Wetland[E]
Discharge (gpm)	70	16	1.7	30
Influent (ppm)	157	42.5	230	8
Effluent (ppm)	30	0.9	0	3.33
Land requirement (acres)	n/a	n/a	0.36[F]	0.0148[G]
Load influent (lbs/day)*	18.9	51.1	276.4	1.5
Load effluent (lbs/day)*	3.6	1.1	0	0.4
Load reduction (%)*	19%	2%	100%	27%
System specifications (general)	Electrodialysis unit, disinfection unit, dosing station	Wells, small building, water tanks, sand filters, RO plant, and storage reservoir	27°C H_2O, 30 gal tanks, 24 hrs	Gravel, sand removal, filter, macrophytes, slow moving water, 20–25°C (68–77°F), pH >7
Byproducts (waste)	Salty brine with high nitrate concentration	Brine and concentrated constituents	n/a	Habitat
Cost (approx)	Cost = Infrastructure + Electricity + Membranes + Electrolyte + Brine disposal + Maintenance	Cost[H] = Capital Cost + Membrane + Electricity + Labor + Filters + Water Quality Additives + Waste Disposal	n/a	Cost = land requirement + construction + maintenance ($22,500 per acre according to UNHSC 2008)

* Normalized to 100 gpm

Parameters	Nitrogen Bioretention Laboratory Study[I]	Woodchip Bioreactors[J,K]	Woodchip Bioreactors with Controlled Drainage[L]
Discharge (gpm)	0.00000944	0.26, 0.0021 gpm	1.96[M]
Influent (ppm)	7.993	5, 57 ppm	2.8-18.9[N]
Effluent (ppm)	3.1	2, 11 ppm	0.1-14.6[N]
Area (acres)	0.00235	0.2, 9[O]	62
Load influent (lbs/day)*	25	6, 69	0.62
Load effluent (lbs/day)*	9.9	2.4, 13	0.14
Load reduction (%)*	40%	40%, 19%	23%
System specifications (general)	Constant regulated flow, denitrification media, sand/soil, mulch, impermeable layer	Box structure or trench (sometimes lined), installed in tile drainage or drainage ditch (>10 year life span)	Controlled drainage structure for each flat field in addition to bioreactor specifications
Byproducts (waste)	none	N_2O, DOC	Possible concern of N_2O emission
Cost (approx)	Cost = materials + maintenance + labor	Cost = materials + labor ($2.39-$15.17 kg or $1.09-$6.90 lb-N)	Cost = ($500-$2000 control box) + labor + maintenance + bioreactor cost

* Normalized to 100 gpm

[A] Reference: Prato and Parent 2010 (Technical report)
[B] Reference: Schoeman and Steyn 2003
[C] Reference: Stewart et al. 2008
[D] Highly controlled laboratory experiment
[E] Reference: Maehllum and Stlnacke 1999
[F] Assuming each 30 gal tank has a footprint of 468 in^2, and each tank can treat 30 gal/day
[G] 2-200 acres per million gallons influent to be treated each day
[H] Capital Cost $29,900, Operating Cost $0.003/gal; 2002
[I] References: Siegel 2008; Hseih et al. 2007; David et al. 2006; Kim et al. 2003
[J] Two studies are included: low concentration values are from an Ag field study; high concentration values from a septic tank system
[K] Reference: Robertson et al. 2000
[L] Reference: Woli et al. 2010
[M] Measurements based on a 3-year average of controlled drainage flow (2007-2009)
[N] Tile monitoring ranges for Nitrate concentrations at inlet and outlet of the bioreactors 2008-2009
[O] To estimate subsurface land requirement, volume of bioreactors in study were normalized for 100 gpm, and depth of 'scaled up' bioreactors was assumed to be 1 meter.