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SOURCE CHARACTERIZATION, EVALUATION, AND
TREATMENT POTENTIAL OF
AGRICULTURAL FILTER STRIPS

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Rebecca Anne Larson

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SOURCE CHARACTERIZATION, EVALUATION, AND TREATMENT POTENTIAL OF AGRICULTURAL FILTER STRIPS

By

Rebecca Anne Larson

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ABSTRACT

SOURCE CHARACTERIZATION, EVALUATION, AND TREATMENT POTENTIAL OF AGRICULTURAL FILTER STRIPS

By

Rebecca Anne Larson

Diffuse source pollution produced by runoff from animal feeding operations contains high concentrations of pollutants that pose serious risks to surface and ground water. Agricultural filter strips are an economical treatment option for farmstead runoff but have not been investigated as to the water quality after infiltration into soil subsurfaces. A runoff source characterization provided water quality data for four farmstead runoff pollutant sources: animal manure, general use impervious areas, upright feed storage, and bunker feed storage. Results from these sources, including two composite samples, indicated that animal manure produced the greatest pollutant concentrations, but due to the small footprint, the feed sources were of more concern for maintenance practices. It was concluded that quantity and dilution of runoff were the most significant factors in determining the impact from pollutant sources.

Three field-scale agricultural filter strips were investigated to determine pollutant removal percentages for typical operation at a small and medium sized dairy. Ten sampling events at the MSU dairy, a 160 cow dairy with a 2.42 acre drainage area, were analyzed for surface and subsurface runoff quality on two adjacent filter strips. A third filter strip was located on a small 40 cow Michigan dairy which had a drainage area of approximately 0.5 acres. The small Michigan

dairy had greater removal percentages for the majority of the 17 water quality parameters, resulting from the addition of a bioretention basin, decreased loadings, and sand soils.

Thirty soil columns were investigated for treatment depth, soil type, and submergence. Columns received synthetic wastewater applications two times per week, and effluent was analyzed for 11 water quality parameters. Columns with a depth of 30 inches or greater produced effluent concentrations that did not pose groundwater concerns for most water quality parameters. Concentrations of BOD_5 were typically below 6 mg/L for columns greater than 12 inches. Nitrate concentrations were greater than 10 mg/L for all columns and posed potential to impede implementation of this technology if they cannot be reduced. Sand soils provided soil characteristics that increased pollutant removal as compared to soil columns with a sandy loam soil. Soil type and depth of treatment were determined to be significant factors in column performance.

Treatment for the field-scale and laboratory soil columns followed the same trends, although field treatment percentages were reduced. The small Michigan dairy filter strip had similar removal to the laboratory study as compared to the sandy loam columns and the MSU dairy filter strips, showing greater continuity in performance of sand soil subsurfaces due to increased porosity and a decrease in soil moisture holding capacity.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1: INTRODUCTION

Diffuse source pollution from animal feeding operations has the potential to contaminate ground and surface water. Animal manure, feed, and other animal farmstead additives and wastes are susceptible to transport during a precipitation or thaw event, resulting in non-point source pollution (US EPA 2003). Feedlot wastes have the potential to contaminate surface water due to runoff from impermeable surfaces or saturated soils and aquifer contamination due to leaching through permeable soils (Burkholder et al. 2007). Water quality from these types of operations has been examined as early as the 1970's. Literature shows feedlot runoff contains high oxygen demanding wastes, elevated nutrient concentrations, organic material, sediment, salts, viruses, bacteria, and other microorganisms (US EPA 1993). Reported pollutant concentrations (Table 1) reveal that runoff from beef cattle feedlots exceed those originating from dairy operations. However, both pose potential for contamination if not properly handled, treated and disposed.

Table 1: Runoff water quality parameters for dairy and beef feedlots

Parameter	Concentration (mg/L)					
TKN	20-180	300	30-400	1122	n/a	n/a
N	n/a	n/a	n/a	n/a	2640	580
TS	500-7100	3700	2800-8400	12777	57800	11230
COD	130-14000	4220	600-5000	14288	79600	7850
P	2-40	64.1	20-50	n/a	770	120
Farm Type	Dairy	Dairy	Dairy	Beef	Beef	Beef
Reference	Larson (2009)	Dickey and Vanderholm (1981)	Nye (1982)	Dickey and Vanderholm (1981)	Edwards et al. (1983)	Clark et al. (1975)

Human and environmental health concerns result from improper handling and treatment of these waste streams (Burkholder et al. 2007). Of particular concern is nitrogen and pathogens. Additionally, runoff containing solids, oxygen demanding waste, and excess nutrients contribute to anoxic conditions in waterways and impact aquatic communities and habitats (Burkholder et al. 2007). Excess nutrient concentrations have been reported as a cause of environmental concern throughout the world. Nitrate contamination from various wastewaters, which sources include animal production facilities, have been measured at elevated concentrations in numerous countries including the United States (Kirby et al. 2003). Assessment has shown that agricultural sources are a leading source of impaired waterways (US EPA 2004) and diffuse agricultural phosphorus sources are a leading contributor to this water pollution (Parry 1998). Further, barnyard and animal storage areas are the leading source of runoff containing phosphorus among agricultural operations (Hively et al. 2005). Eutrophication of waterways can occur with only small additions to phosphorus concentrations (Hart et al. 2004). Excess phosphorus concentrations result in algal blooms and decreased oxygen as it is typically the limiting nutrient for the processes producing these effects (Anderson et al. 2002). Decreased oxygen concentrations in waterways leads to fish kills and habitat destruction (Burkholder et al. 2007; Anderson et al. 2002). Metals within runoff and in the leachate from soils subject to land application have reached surface and groundwater. The release of dissolved Fe^{2+} into groundwater is one of the most prevalent groundwater problems worldwide (Lovley 1991). Agricultural practices have had direct effects on the concentration of NO_3^-, SO_4^-, Cl^-, P, C, and As within groundwater (Bohlke 2002). Twelve percent of groundwater wells in Michigan (of a sampling of 73 wells in 1997) indicated arsenic contamination greater than the US EPA maximum contaminant level of 50 ug/L (Kim et al. 2002).

Management practices for runoff can be costly and if not properly designed, installed, or operated are ineffective in reducing environmental concerns. Current treatment options include land application, conventional wastewater treatment, and runoff infiltration designs. Land application has limitations dependent upon available field area based on accepted agronomic application rates. Conventional treatment requires extensive capital and operational costs that are not economically feasible for many animal operations. Vegetative filter strips are being investigated as an economically feasible management option for treating farmstead area runoff. Agricultural vegetative filter strips are engineered treatment systems which direct flow over a vegetated soil. The vegetation and design reduces runoff contaminant concentrations by increasing sheet flow thereby increasing sedimentation and infiltration. Biological, physical, and chemical processes within the infiltration zone are the principal mechanisms to effectively reduce the loadings and improve water quality prior to reaching groundwater (Koelsch et al. 2006). Surface water quantity issues are addressed as increased infiltration restricts runoff flow to surface water.

Vegetated filter strips are proven to reduce the pollutant concentrations from feedlot runoff. However, literature has shown variability in treatment performance of the various pollutants (Koelsch et al. 2006). Detailed studies have examined the trapping effectiveness and surface outflow to determine the ability of filter strips to eliminate surface water discharge (details of these studies can be found in the literature review). However, research has not provided definitive results as to the effectiveness of filter strips to reduce contaminant loads prior to leachate reaching groundwater. To protect against groundwater contamination, continued research is critical to determine the comprehensive pollutant removal capacity of vegetated agricultural filter strips. Filter strips are currently functioning in the absence of supporting data on the fate of contaminants once they have infiltrated into the soil subsurface. Determining pollutant removal capacity of these engineered systems is critical to formulate design standards that can maintain a sustainable water cycle.

1.1 Objectives

Assessment of the treatment potential and evaluation of design recommendations for agricultural filter strips was organized into three critical research elements. Initially, preliminary research focused on characterizing on-farm runoff sources for quantity and quality concerns. Secondly, analysis of the field treatment processes to determine pollutant removal processes, and finally a laboratory evaluation designed to identify critical issues associated with design depth.

Below is a detailed explanation of the objectives for each of the three research elements.

1.1.1 Runoff Characterization

Collect and analyze precipitation data to determine the source characteristics from a representative dairy farm runoff. Included is the evaluation of the quantity and quality impacts from the heat check lot, upright silos, bunker silos, and general impervious roadway areas used for transport and mixing of farmstead operational inputs and outputs. Based on the characterization, recommend on-farm management practices to reduce the pollutant quantity and increase water quality.

1.1.2 Analysis of Field Treatment Systems

Determine the pollutant removal of agricultural filter strips in typical environmental and farmstead conditions. Specific objectives include the following:

- Assess the surface and subsurface water quality at two field sites.
- Assess current practice standards in regards to operation and maintenance procedures.
- Determine if agricultural filter strips are an effective agricultural treatment/management option as designed, with a particular emphasis on metal leaching into groundwater.

- Determine treatment consistency throughout season and rainfall events.

1.1.3 Laboratory Evaluation of Treatment System Design Components

Conduct soil column experimentation to assess the required soil depth to achieve adequate treatment of land applied agricultural runoff prior to infiltration to groundwater.

- Statistically determine the pollutant removal capacity of a volume of the overall soil column system for the various water quality parameters.
- Determine impact of soil depth and total soil volume to pollutant removal.
- Examine the influence of groundwater capillary rise on the depth of soil required for treatment of agricultural runoff.
- Find the degree of treatment variance between two defined soil types, sand and sandy loam, to determine if further detailed analysis for soil type is warranted.

CHAPTER 2: LITERATURE REVIEW

Previous work done on the three research areas provided a basis for experimental design. The following literature review sections contain the details from these studies relevant to the proposed research.

2.1 Runoff Characterization

Pre-treatment of waste is a critical for agricultural filter strips (Koelsch et al. 2006). An effective management plan can reduce runoff concentrations, reducing the environmental effects and the loading to treatment systems through source reduction, reduction in transport mechanisms, and/or removal/degradation of pollutants prior to reaching waterways (Azevedo, 1974; Sweeten, 1998; US EPA, 2003). Practices include covering pollutant sources prior to precipitation, sweeping impervious surfaces, and/or maintaining faces on feed bunkers. Although previous research has examined water quality data of agricultural feedlots as a whole, there has been no source investigation. Identifying contaminant sources and strength is critical in developing an effective management plan.

2.2 Analysis of Field Treatment Systems

Engineered filter strips have two main mechanisms for pollutant removal, sediment trapping and infiltration treatment processes. Sediment trapping is a result of vegetation and sheet flow, which reduces flow velocities and captures sediments and sediment bound pollutants. Sediment bound pollutants have greater removal rates than dissolved or soluble contaminants due to higher trapping efficiencies (Goel et al. 2004; Schmitt et al. 1999). However, infiltration is responsible for the majority of pollutant removal, in particular dissolved contaminants (Dosskey et al. 2007, Lee et al. 2003). Infiltration allows for pollutant soil assimilation, microbial degradation, and plant uptake. Removal rates by infiltration are determined by biological activity, adsorption, filtration, and oxidation, which are the primary mechanisms (Brown and Caldwell 2007). Microbial degradation rates are dependent upon environmental conditions including temperature, moisture, energy sources, and oxygen and nutrient availability (Donker et al. 1994). Temperature is typically directly related to microorganism's cell reaction rates and the environmental conditions of the cell habitat. Microorganism decomposition rates increase with increasing temperature up to approximately 45°C, after which the rate declines (Paul and Clark 1996). Oxygen and soil moisture also have significant effects on degradation rates. Aerobic conditions result in greater degradation rates as compared to anaerobic cells (Paul and Clark 1996). Moisture has an indirect effect on degradation rates as high levels decrease the oxygen content therefore leading to the slower rates associated with anaerobic microorganisms, but also controls the solubility and availability of nutrients required by microorganisms to maintain activity (Paul and Clark 1996). Removal of specific contaminants varies with environmental conditions.

Nitrogen removal, for example, is more effective in the soil subsurface than at the soil surface and is dependent upon soil type, hydrology, and biogeochemistry (Mayer et al. 2007). Nitrogen removal is accomplished primarily through various nitrification and denitrification processes in addition to plant uptake in the overland flow and soil infiltration. Although the processes are not generally understood, denitrification plays the dominant role in both (Corbitt 1998). The majority of phosphorus is fixed within the soil profile, although small amounts are removed via plant uptake (Corbitt 1998). In addition to metals within waste streams, an overload of biodegradable organic material can lead to incomplete removal within the soil profile and mobilization of iron, manganese, and other metals (McDaniel 2006), greater details in this process are discussed in the next section.

Filter strip design dimensions of width, length, and slope impact pollutant removal. An increase in filter strip width increases infiltration, reducing the volume and contaminant concentration and surface outflow (Schmitt et al. 1999) as they provide more area for infiltration. Nutrient removal, such as nitrogen, is more effective in wider strips (Mayer et al. 2007). Trapping, however, is not impacted by filter strip width as it is a function of the vegetation and slope (Jin and Romkins 2001). Literature values are available for minimum and maximum filter strips widths. There is a point in which increasing the filter strip width begins to impact the flow design, as it is difficult to maintain even distribution and sheet flow over very large widths, and does not result in increased removal.

Increased filter strip lengths can also reduce pollutant loads. The length of the filter strip has been shown to impact the removal of inorganic compounds, such as Cu, Fe, Zn, K, Na, and Ni (Edwards et al. 1997). A longer filter strip has also been shown to increase removal due to increased trapping (Lee et al. 2003). Sediment removal of over 90% resulted with a filter strip of 10 m length (Dillaha et al. 1988; Goal et al. 2004). Magette et al. (1989) investigated 4.6 m and 9.2 m long filter strips and found an increase in pollutant removal with an increase in length, it was also found that a 4.6 m filter strip was below the threshold to achieve any removal of some pollutants. Dickey and Vaderholm (1981) found that channelized systems require greater lengths than those designed for overland flow for equivalent removal performance. Many pollutants experienced an exponential reduction with increasing length, but NO_3^-, TKN and TOC did not undergo significant reductions after 3 m and NH_3, PO_4^-, and TP beyond 6 m (Srivastava et al. 1996).

Increased slopes result in reduced treatment effectiveness (Hay et al. 2006). Sediment trapping and transport is strongly dependent upon the slope of the filter strip. An increase in the slope leads to a reduction in the trapping efficiency and an increase in pollutant transport (Jin and Romkins 2001, Dillaha et al. 1988). However, the slopes must be great enough to maintain sheet flow.

Consequently, length, width, and slope are all critical components for sizing agricultural filter strips. Recommendations from Dickey and Vanderholm (1981) include a minimum width of 61 m and a length to accommodate runoff volumes for a 1-yr 24-hr storm calculated based on slopes and contact time. Others investigated ratios of drainage area to infiltration areas from 1:1 to 6:1 (Nienaber et al. 1974, Lorimor et al. 2003). NRCS designs are based on the infiltration of a 25-yr 24-hr storm and the length and width are based on the infiltration of these volumes using the equations provided in Appendix C.

Filter strip soils must provide adequate filtration to avoid flooding during wastewater application. Minimum hydraulic conductivities have been suggested by previous researchers from 0.27-0.5 in/hr (Schueler 1987). The soil textures that fall within this range include sand, loamy sand, sandy loam, loam, and silt loam (silt loam falls within the lower range only) (Rawls et al. 1982). Previous research done by Mokma (2008) added to the validity of these assumptions in which clay loam was excluded as it did not adequately treat waste water. Komor and Hansen (2003) speculated that poor performance and a greater impact to groundwater was due to greater hydraulic conductivities at a site with silt loam soil as compared to a second site with loamy soils. Saturated hydraulic conductivities are increased as compared to that of unsaturated conducitivties (Miyazaki 1993). This can lead in increased flow within land application of wastewater through soil profiles, decreasing the time for adsorption and increasing the transfer of pollutants within the soil.

Vegetation plays a role in the uptake of pollutants by impacting velocity and infiltration processes. In addition, vegetation develops dense mats of roots on the upper portions of soil profiles which can provide nutrient trapping and increases soil oxygen through respiration (Bhaskar, 2003). Various researchers have experimented with the selection of vegetation to increase pollutant removal and demonstrated that some species are contaminant specific (Schmitt et al. 1999). For example, nitrogen removal is dependent upon the vegetations' depth of root zone and ability to provide flow paths that favor microbial denitrification (Mayer et al. 2007). Goal et al. (2004) has shown that sod grasses have the greatest effect on soluble phosphorus removal and particulate nutrients in comparison to rye grass and mixed grasses. In addition, Goal et al. (2004) found no trends with grass type and NO_3^- removal. Some results have shown no change in the collective removal efficiencies between entire vegetation classes, such as forest vegetation and grasses (Dosskey et al. 2007). In terms of plant life cycles, perennial plants have been shown to trap sediments effectively and allow for greater infiltration and reduce erosion in comparison to annual plants (Lovell and Sullivan 2006). However, it has been shown that there is no difference within perennial species in terms of removal (Schmitt et al. 1999). Trapping efficiency is increased due to an increase in vegetation density (Lee et al. 2003). After a two years of growth there is no difference in infiltration due to age of vegetation (Dosskey et al. 2007). Multiple plant species allows for numerous soil root sizes, and various stalk and leaf sizes to have the greatest overall impact on infiltration and sedimentation (or trapping).

Although previous studies have investigated various filter strip design parameters including length, width, slope, and vegetation in an effort to determine design standards and maximize treatment efficiency, there have been no studies to date correlating loading to the depth of soil and groundwater table required to effectively treat runoff prior to reaching groundwater. Determination of this depth is critical in developing effective standards for the implementation of the practice.

2.3 Laboratory Evaluation of Treatment System Design Components

Primary soil assimilation mechanisms include biological oxidation, adsorption, filtration, and oxidation (Brown and Caldwell 2007). The soil profile provides the environmental conditions to support biological and biochemical activity (Haggblom and Milligan 2000). Application of wastewater increases the soil pore water thereby decreasing available oxygen within the soil. Under aerobic conditions, carbon sources are the electron donors with oxygen accepting the electrons (Tarradellas et al. 1997; Rittmann and McCarty 2001). After most of the oxygen is depleted, anaerobic and facultative microorganisms become dominant. The carbon source remains the same but the electron acceptor changes in order of energy potential (Haggblom and Miller 2000). As oxygen is removed or fixed within the system, other oxidants act as electron acceptors.

The diagenesis model (or electron tower) ranks the oxidants in order of free energy yield per mole of organic carbon oxidized, or O_2, NO_3^-, MnO_2, $Fe(OH)_3$, SO_4^{2-}, and methanogenesis (Froelich et al. 1979; Postma and Jacoksen 1996; Matocha et al. 2005). After the oxygen is utilized within the system, oxidants will be reduced in accordance to free energy yield. A reduction of metals causes previously immobile metals to mobilize and leach into groundwater. Mn solubility is increased when converted from Mn(IV) to Mn(II) as Mn(II) is typically released into solution (Norvell 1988). Fe(II) is more soluble than Fe(III) and is governed by pH, as reduced forms are more prevalent in soils with a lower pH.(Lindsay 1985). Solubility of metals can be decreased through an increase in pH (McGowen and Basta 2001). Mn oxides are present as surface coatings arranged in octrahedra sheets or tunnel structures and are typically associated with Fe oxides (Bartlett and Ross 2005). Acidic or saturated soils in combination with soil organic material can easily lead to Mn(IV) reduction (Bartlett and Ross 2005). Reduction mechanisms can be biological or physical/chemical in nature. Mn complexes are more available for microbial reduction than the Fe complexes (Lovley 1991). Biological iron reduction from Fe(III) to Fe(II) can follow numerous pathways including bacterial reduction, acting as a respiratory electron acceptor, and interactions with microbial end products (Paul and Clark 1996). Enzymatic conversion of Fe(III) to Fe(II) under anaerobic conditions is the main cause of iron reduction (Roden and Zachara 1996). Organisms reduce Fe(III) and typically Mn(IV) enzymatically, in addition Fe^{2+} reduces Mn(IV) nonenzymatically (Paul and Clark 1995). The affinity for Mn(II) to adsorb to manganese oxides results in excess bound Mn(II), so when Mn(IV) is reduced, soils release the excess bound Mn(II), further increasing the release of Mn(II) (Fendorf et al., 1993a, 1993b),. Mn(IV) and Fe(III) reduction rates are governed by the mineral surface area (Burdige et al. 1992; Roden and Zachara 1996; Larsen et al. 1998; Matocha et al. 2005). In addition, nitrogen species within the soil can also affect metal mobilization. The accumulation of NO_2^- results in the reduction of MnO_2 to Mn(II) (Vandenabeele, 1995) while NO_3^- typically inhibits iron reduction (Paul and Clark 1995). Although some relationships and mechanisms have been developed found, there are many exceptions that are still to be explained. For example, sulfate reduction is typically preceded by Fe oxide reduction according to the diagenesis model, there have been cases where pore water has shown a reduction in SO_4^{2-} prior to Fe oxides (Matocha et al. 2005). The relationship between the various water quality parameters is not completely known. Although Mn and Fe leachate are mainly aesthetically unpleasant in groundwater, metals further down the electron tower, such as arsenic, will leach after other oxidants have been exhausted and pose serious human health risks. Groundwater wells high in As concentrations also reported high concentrations of Mn(II) and Fe(II) (Kim et al. 2002). Although the processes involved have been investigated, there are still many factors to be determined, and in particular, how these processes will occur simultaneously (Holden and Fierer 2005).

An increase in soil depth is predicted to provide greater pollutant removal due to the increase in available soil surface area, an increase in the soil pore area for microbial activity, and an increase in available area for the vegetative root system (Bratieres 2008). Microorganisms are naturally present within the soil profile (Paul and Clark 1996), and a larger soil volume will in turn increase the soil microbial mass. However, it has been shown that microbial biomass is the highest at soil surfaces and decreases with an increasing depth (Paul and Clark 1996; Holden and Fierer 2005). The change in microbial activity may not be linear. Microbial degradation rates may also be affected due to depth. Oxygen at the soil surface is at atmospheric concentrations but drops significantly with increasing depth (Wood and Petriatis 1984), which in turn would decrease microbial degradation rates with increasing depth. Changes in microbial mass are also greater with increasing depth than the change reported in different soil types (Holden and Fierer 2005), so it is assumed that soil depth will have a greater impact on degradation and metal mobilization than soil type.

Adsorption of Mn and Fe is a minor component in Fe and Mn reactions within soil as these reactions are mainly driven by pH and oxidation reduction reactions (Shuman 2005). Therefore the increase in the soil cation exchange capacity (CEC) within the sandy loam due to increased fractions of silt, clays, and organic matter, is not predicted to have a large overall effect in metal leaching.

A lack of literature exists for the effect of capillary rise on pollutant removal capacity. An unpublished study by Mokma (2008) investigating pollutant assimilation of food processing waste in soil columns revealed soil saturation had risen 2-3 inches from the bottom of the column during deconstruction due to soil saturation within the bottom of the column due to poor drainage resulting in capillary rise. The increase in the soil water has a direct effect on the availability of oxygen within the soil and will lead to anaerobic soil conditions. However, it has been shown that biomass increases directly above the water table (Paul and Clark 1996). In addition, the microbial activity increases in the capillary fringe as the rising and falling of the water table redistribute necessary nutrients and microbial mass (Holden and Fierer 2005).

CHAPTER 3: METHODS AND MATERIALS

Experimental design was based on previous research. Details of the design and experimental operation for each research section is outlined below.

3.1 Runoff Characterization

Runoff for characterization of water quality was collected at the Michigan State University Dairy Teaching and Research Facility (MSU dairy). The MSU dairy is a fully operational 160 head dairy facility that was originally designed to transport runoff using a traditional urban storm water collection system. In 2008, the existing system was modified to collect and divert water from two approximately one acre in size areas into two 86,774 gal storage basins. Four source locations were sampled to investigate pollutant sources; the areas adjacent to the heat check lot, upright silos, bunker silos, and main roadway. The two storage basins were also sampled for the composite runoff water quality. The heat check lot is an outdoor cattle holding area and consequently, is typically high in animal waste. Both silo locations are feed storage areas. The upright silos are covered, but are prone to spillage and produce dry weather leachate. Bunker silos are partially uncovered feed storage and remain open to the elements making them particularly susceptible to environmental conditions, leaching, and runoff. The main roadway is used to transport and mix feed and animal waste, it provides data for multifunctional impervious feedlot areas. Storage basin 1 collects runoff from a 1.28 acre area containing the heat check lot and roadway areas. The heat check lot accounts for 9% of the total drainage surface area for storage basin 1 with the remaining area comprised of roadway surfaces. The second storage basin collects runoff from a 1.14 acre area where 23% of the area is bunker silos, 6% of the surface area is upright silos, and the remaining surface is roadway.

A comprehensive management plan, Appendix A, was given to the MSU dairy prior to sample collection.

3.1.1 Sample Collection Methods

Samples were collected during precipitation events that produced runoff volumes adequate for sampling. Clean plastic sample bottles were used for each new sample and collection devices were cleaned with a dilute bleach solution and rinsed with de-ionized water a minimum of three times to avoid cross contamination of samples. Grab samples were collected above sewer grates where large quantities of runoff accumulated, then were preserved if required, and stored until analyzed following proper quality control and quality assurance (QA/QC) protocols, Appendix B.

3.1.2 Laboratory Analysis

All samples were evaluated for the water quality parameters listed in Table 2. These parameters are typical water quality indicators used by environmental regulatory agencies. Nutrient removal was analyzed for nitrogen species and total phosphorus as these are the major nutrients of concern associated with agricultural practices. Oxygen requirements were measured via the 5-day biochemical oxygen demand (BOD_5) and chemical oxygen demand (COD). Manganese (Mn) and iron (Fe) concentrations were measured as indicator species for metal leaching and reduction potential. Arsenic was also included as it is currently a significant groundwater contamination concern in Michigan. Sedimentation, filtration and loading limits were investigated using solids data. Samples shaded in Table 2 were preserved and transported to the Michigan Department of Natural Resources and Environment State Environmental Laboratory for analysis. The remaining parameters were analyzed at the MSU Ecological Engineering Laboratory. All laboratory analyses were subject to detailed QA/QC procedures, see Appendix B.

Table 2: Water Quality Parameters for Source Characterization and Field Treatment Systems

Parameter	Method	Detection Limit	Hold Time
Alkalinity	USEPA 310.1 (1)	10 mg/L $CaCO_3$	24 hours
Arsenic	SW-846 method 6020 (2)	1 µg/L	6 months
BOD_5 (mg/L)	USEPA 405.1 (1)	2 mg/L	Analyze Immediately
Chloride	SW-846 method 325.2 (2)	1mg/L	28 days
Conductivity	Conductivity Cell	1 µS/cm	28 days
Iron	SW-846 method 6010B (2)	0.02 mg/L	6 months
Manganese	SW-846 method 6020 (2)	5 µg/L	6 months
NH_3	USEPA 350.3 (1)	0.02 mg/L NH_3-N	28 days
NO_2	USEPA 354.1 (1)	0.002 mg/L NO_2-N	24-48 hours
NO_3	USEPA 353.3 (1)	0.3 mg/L NO_3-N	24 hours
pH			Analyze immediately
TKN	USEPA 351.1 (1)	1 mg/L	28 days
TOC	SW-846 method 415.2 (2)	0.5 mg/L	28 days
Total and Soluble COD	USEPA 410.4 (1)	1 mg/L	28 days
TP	USEPA 365.1 (1)	0.02 mg/L P	28 days
TS	USEPA 160.3 (1)	0.1 mg/L	7 days
TSS	USEPA 160.2 (1)	0.1 mg/L	7 days

(1) (US EPA 2009a)
(2) (US EPA 1996)

3.1.3 Precipitation Data

In addition to water quality data, precipitation data was measured using a Campbell Scientific TE-525 rain gage produced by Texas Instruments. Precipitation data provided an estimate as to the intensity of a rainfall, identification of rainfall return period and precipitation duration and, in conjunction with the recorded sample time, an estimate of the water accumulated on the ground at the time the samples were taken for analysis of covariance when appropriate.

3.1.4 Data Analysis

Statistical analysis was conducted using ANCOVA in SAS, with rainfall and season as covariates when appropriate, to determine the statistical significance of location on each measured water quality parameter. Assumptions that residuals are normally distributed and the variances are homogenous were evaluated using normal probability plots and side-by-side box plots to ensure their validity. Covariates were selected to an attempt to reduce the experimental wide error; they were used within the statistical model when ANOVA indicated that they increased model significance. When ANCOVA or the ANOVA was significant, difference of least squares means was used to compare the treatment means and their interactions.

3.2 Analysis of Field Treatment System

The second phase of research was a full scale implementation and analysis of vegetated filter strips at two site locations, the MSU dairy and the small MI dairy, details for this second site are discussed on page 24. Each site was designed in compliance with the specifications of the NRCS Technical Guide titled "Wastewater Treatment Strip 635," Appendix C. Sample collection and analysis of runoff water quality pre and post treatment provided data for assessment of treatment.

The design at the MSU dairy is composed of two filter strips, each 400 feet long and 40 feet wide with a 4% slope. Side slopes of 12.5% along the length of the filter strip created the channel which was backfilled with the sandy loam soil native to the site. Vegetation was planted as a mixed grass species containing 37% Tuscany II Tall Fescue, 28% Smooth Bromegrass, 20% Graze N Gro Annual Ryegrass, and 12% Chiefton Reed Canarygrass. After a two year growing period, the Annual Ryegrass was the dominant species, with the three remaining species onsite but at lower densities. Five rock checks extended across the width of the filter strip (with a depth and width of two feet) at the flow entrance and every 100 feet downslope to redistribute flow. Storm drains divert runoff from two locations, one from 1.14 acres surrounding the feed sources and a second from 1.28 acres which included the heat check lot and roadways, into two separate 86,774 gallon concrete basins, Figure 1. Grab samples were collected from each of the two storage and sedimentation basins for baseline data.

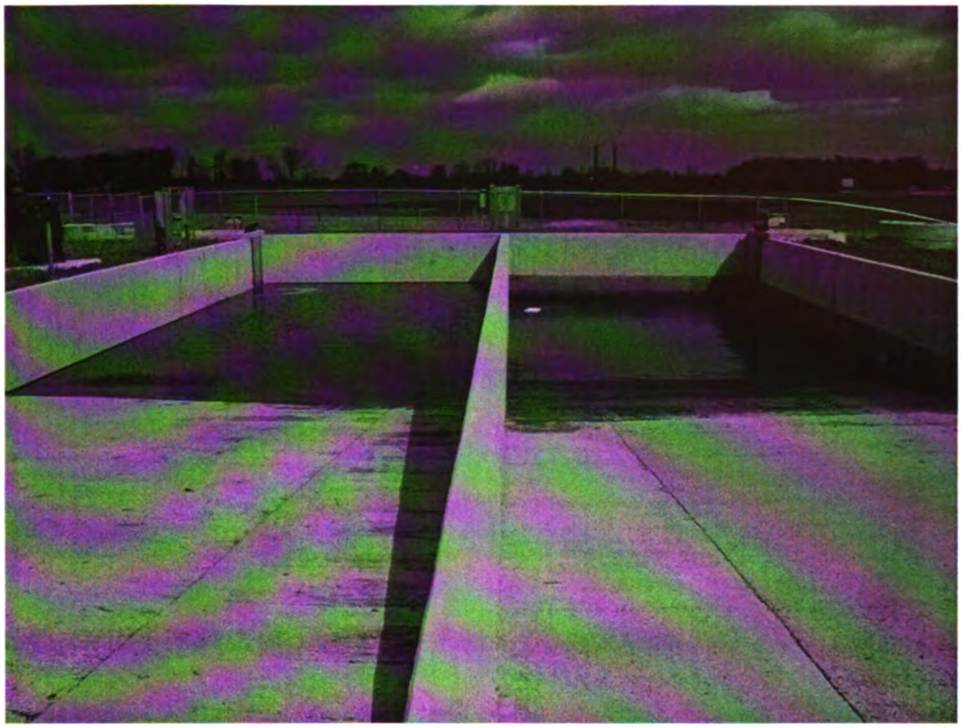

Figure 1: MSU Dairy Concrete Storage and Sedimentation Basins

From each storage and sedimentation basin, wastewater is transported to the two small concrete distribution basins at the top of each filter strip, each ~5,000 gallons using a pump system active by level sensors (flow rate ~350 gal/min). Wastewater exits the distribution basins via four vertical slots, each 1 in wide and 24 in tall, which empties into a rock check to evenly disperse flow across the width of the filter strip, Figure 2.

Figure 2: Filter Strip Influent Flow Dispersion

Nine collection boxes were installed in the rock check of each filter strip at the MSU dairy for surface samples. Subsurface drainage tile was installed 9 to 15 inches below the surface 25, 50, and 150 feet downslope to collect infiltrate that has passed through the soil profile. The tile drained to a sample well for collection using a sampling pole affixed with a clean sample container (washed with a 10% bleach solution and tripe rinsed with deionized water between samples). Ten sampling events were collected over a 2 year period.

At the second site, the small MI dairy was designed to treat runoff from a 40 cow dairy from an approximately ¼ acre drainage area. Dairy feedlot and manure storage runoff were diverted via overland flow to a small concrete basin. Effluent from the concrete basin flowed over a weir to a bioretention basin for storage of runoff volumes up to a 25-yr 24-hr storm for the ¼ acre area. The subsurface of the bioretention basin was lined with an impermeable geomembrane with a subsurface collection tile located above the membrane to transport effluent that leached through the soil to the filter strip via gravity. The filter strip was 110 ft long, 40 ft wide, with a 0.5% slope and sandy soil present at the site prior to installation. Rock checks were located at the top of the filter strip and 50 feet. Six surface collection boxes were installed in the two rock checks for surface water collection. Subsurface samples were collected using 1.5 ft and 2.5 ft collection wells made from corrugated pipe buried 3 ft and 13 ft downslope of the first rock check. This subsurface collection method was different from the MSU dairy site as this site had a sandy soil which increased hydraulic conductivity and decreased the length the runoff traveled downslope before infiltrating.

3.2.1 Sample Collection Methods

Grab samples were collected within 24 hours after a rainfall event for all sample locations. Influent data was collected from the two concrete storage basins for the MSU dairy site and the concrete sedimentation basin and bioretention basin at the small MI dairy. After baseline samples were collected at the MSU dairy the pumps were manually activated and one location from each rock check was sampled for surface water quality and all subsurface sampling locations were sampled if effluent was present. Figure 3 provides a diagram of the sampling locations.

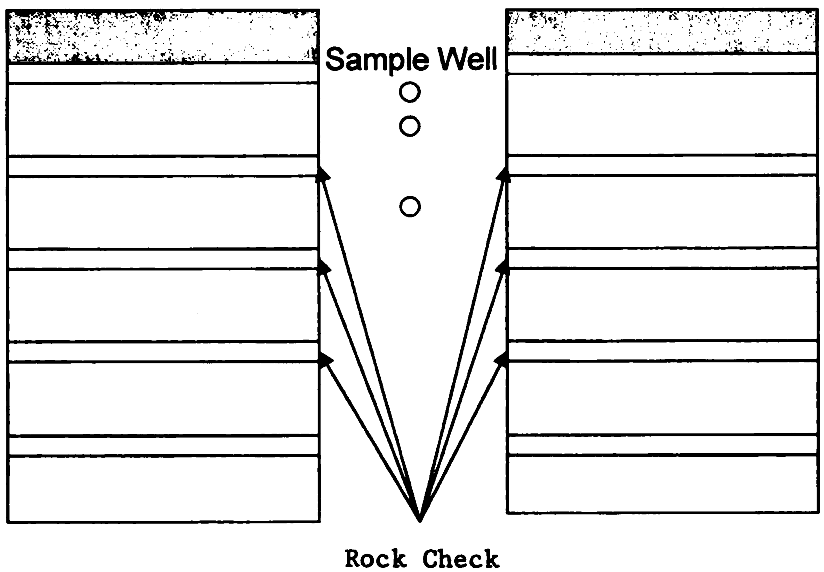

Figure 3: MSU Dairy Filter Strip Sampling Locations

Typical operation at the MSU dairy site relies on pump activation due to level floats at the top of the basin to activate the pumps and 1 foot from the bottom of the basin to turn the pumps off. If a storm event is not large enough to initiate pumping on the high float level switch, pumps must then be manually operated within 72 hours of the storm event.

At the small MI dairy site one sample from each rock check was again sampled and effluent was collected from the 4 samples wells if effluent was present, Figure 4 is a diagram of the sampling locations.

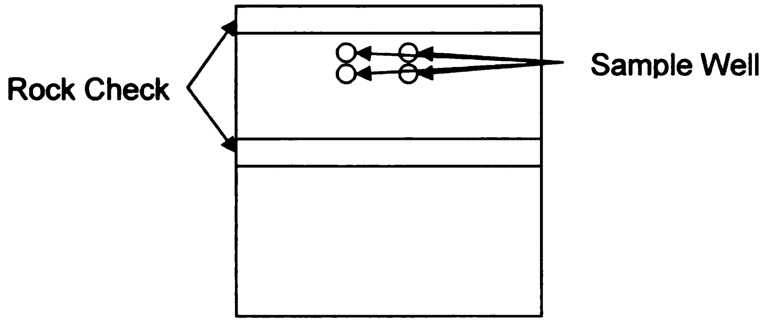

Figure 4: Small MI Dairy Filter Strip Sampling Locations

Rainfall on the filter strip surface area was assumed to be insignificant in dilution of samples. At the MSU filter strip site, water was applied after rainstorm events and rainfall on the filter strip was therefore not a factor in dilution of the applied runoff. Although the small MI dairy filter strip was a gravity fed system, there was a delay from the transport of water from the source location to the filter strip, and the area of the filter strip to which the water was actually applied (within the first 15 ft) was negligible compared to the drainage area (< 1%).

All samples were transported immediately to the MSU laboratories and preserved if necessary. Samples for the State Environmental Laboratory were preserved and transported in a cooler within a two week period. Ten sampling events were obtained from the MSU dairy and five sampling events from the small MI dairy.

3.2.2 Laboratory Analysis

Water quality evaluation was determined by the identical parameters to those of the source characterization listed in Table 2. Analysis procedures are also identical to those of the source characterization, Section 3.1.2.

3.2.3 Precipitation Data

Precipitation data was collected at the MSU dairy using a rain gage. Precipitation for the small MI dairy was obtained from a rain gage in Charlotte, MI, approximately 10 miles from the farm.. This data was critical for comparison of runoff volumes and filter strip performance.

3.2.4 Data Analysis

Data was evaluated for general trends including removal percentages for each water quality parameter. Because conditions varied for each storm, reliable replication was not possible which prevented statistically significant results for analysis.

3.3 Laboratory Evaluation of Treatment System Design Components

Soil columns with surface vegetation were designed, constructed, and operated to evaluate the objectives for the laboratory research. The columns provided data to correlate pollutant removal to soil depth, soil type, and simulated groundwater effects. Wastewater was applied to column surfaces and allowed to leach, producing effluent that could then be analyzed to evaluate pollutant removal.

3.3.1 Soil Column Experimental Design

Soil treatment columns were evaluated for three treatment depths, two soil types, and submerged or not submerged conditions. The three column lengths were 12 inches, 30 inches, and 48 inches. The 12 in column was selected for direct comparison to field data obtained at this depth. An increase in soil depth for the remaining two columns allowed for the investigation of soil depth to pollutant removal. The two soil types were sand and sandy loam, selected from the list of soils in Section 2.2. Sand soil provided data for a soil with the greatest hydraulic conductivity and sandy loam a lesser hydraulic conductivity as a comparison of soils within those accepted for the technology. These soil types also corresponded with the soil types at each field site. Groundwater simulation, or capillary rise effects, were investigated by submerging the bottom end of a set of identical sets of columns for each design depth to mimic the interface between the soil and groundwater, commonly termed the vadose zone. When submerged, the design prevented air from entering the bottom of the column and allowed for capillary rise within the soil column system.

Vegetation, hydraulic load and organic load were held constant throughout testing. Research has shown that vegetation pollutant removal performance varies by individual pollutant and vegetation type, so a mixed grass species was selected for use in all columns to maximize overall pollutant removal. A combination of 37% Tuscany II Tall Fescue, 28% Smooth Bromegrass, 20%

Graze N Gro Annual Ryegrass, and 12% Chiefton Reed Canarygrass was selected, identical to the selection for the MSU dairy. The mixed species provided the necessary variation required for pollutant removal mechanisms associated with vegetation (trapping, uptake, and root size). This vegetation will also provide adequate food for grazing in typical farmstead operations. A constant hydraulic load for wastewater application was determined using a BOD concentration of 225 mg/L. This BOD concentration was selected from preliminary source characterization data for typical dairy runoff loadings with adequate management (Larson 2009). Source characterization research has shown that this concentration is achievable, although concentrations have been found that exceeded this number by an order of magnitude. An organic load of 75 lbs/acre/day was used in the simulated wastewater, as was determined in evaluating data from a prior study conducted by Mokma (2008), which is currently in review for publication. The study results showed metal leaching from a column depth of 36 in from a loading of 75 lbs BOD/acre/day but not from a loading of 50 lbs BOD/acre/day, so the higher loading was selected to produce leachate.

A power analysis in SAS, a statistical computing program, was completed to determine the required number of replications to predict a statistical difference within the treatment effects. As the analysis is relatively variable due to interpretation, only depth and soil type were included to determine power. Unpublished data from Mokma (2008) provided the necessary variance for soil type and depth required for the analysis. Statistical power analysis indicated that 3 replications are required to produce a power of 0.94 for column depth, Figure 5. An increase to 4 replications did not increase the power significantly for column depth, so 3 replications were deemed appropriate. Soil type did not establish a significant power even after 6 replications. As more replications did not produce a significantly greater power for soil type, and the amount of resources necessary to predict a highly significant result are not feasible, replications were limited to 3.

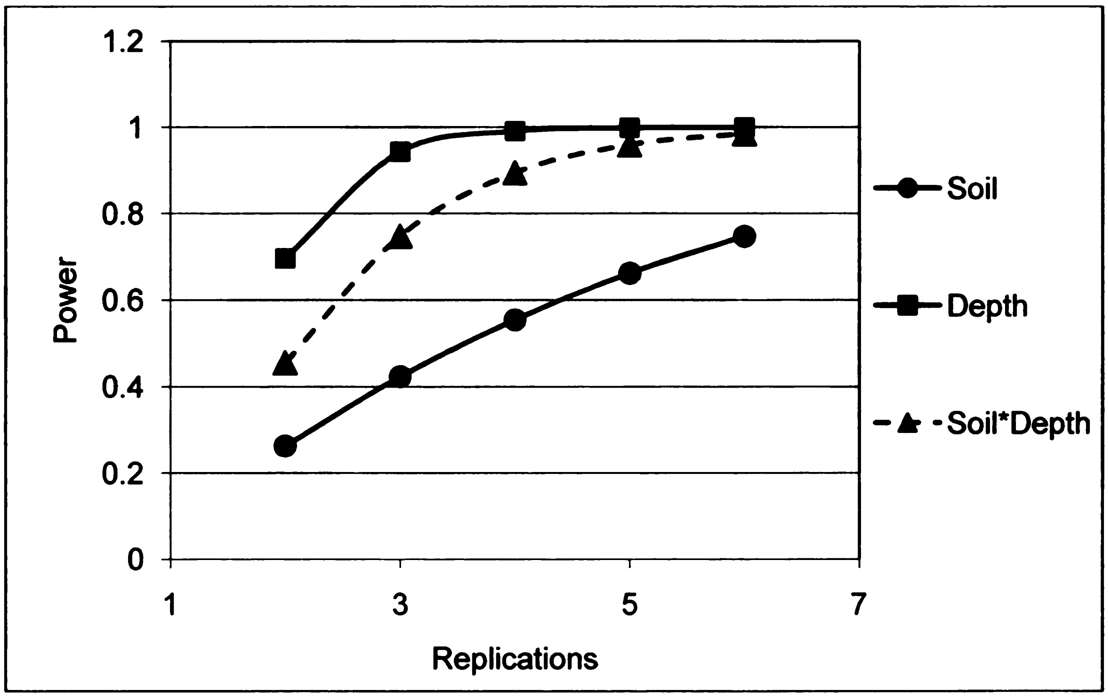

Figure 5: Filter Strip Power Analysis

The three treatments each had the following levels, depth – 3, soil – 2, groundwater – 2, and would require 12 columns for each replication, or 36 columns total. The interaction of capillary rise and soil type were not outlined in the objectives, so direct comparison of the two was not required. Therefore, a groundwater simulation was conducted for sandy loam soils only at each treatment depth. This reduction allowed for determination of the study objectives with a total of 30 soil columns, Table 3. Columns were assigned randomly to experimental conditions to minimize experimental wide error.

Table 3: Soil Column Treatment Assignment

Application	Soil Type	Length (in)	Submergence	Column #'s
WW	Sand	12	Air	12, 25, 26
WW	Sand	30	Air	1, 7, 13, 20
WW	Sand	48	Air	3, 19, 24
WW	Sandy Loam	12	Air	10, 18, 23
WW	Sandy Loam	30	Air	4, 5
WW	Sandy Loam	48	Air	14, 15, 17
Water	Sand	30	Air	30
Water	Sandy Loam	30	Air	22, 29
WW	Sandy Loam	12	Water	2, 11, 21
WW	Sandy Loam	30	Water	6, 16, 28
WW	Sandy Loam	48	Water	8, 9, 27

3.3.2 Soil Column Structural Design

Soil columns, Figure 6, were constructed from 6 inch drainage pipe. The lengths of the columns correspond to the actual soil depth plus two additional inches on the top of the columns for application of wastewater and another two inches on the bottom that was packed with washed pea gravel to prevent soil from settling and leaching from the column. The pea gravel rests on a fine fiberglass screen attached to the bottom of the column to allow for free flow of leachate.

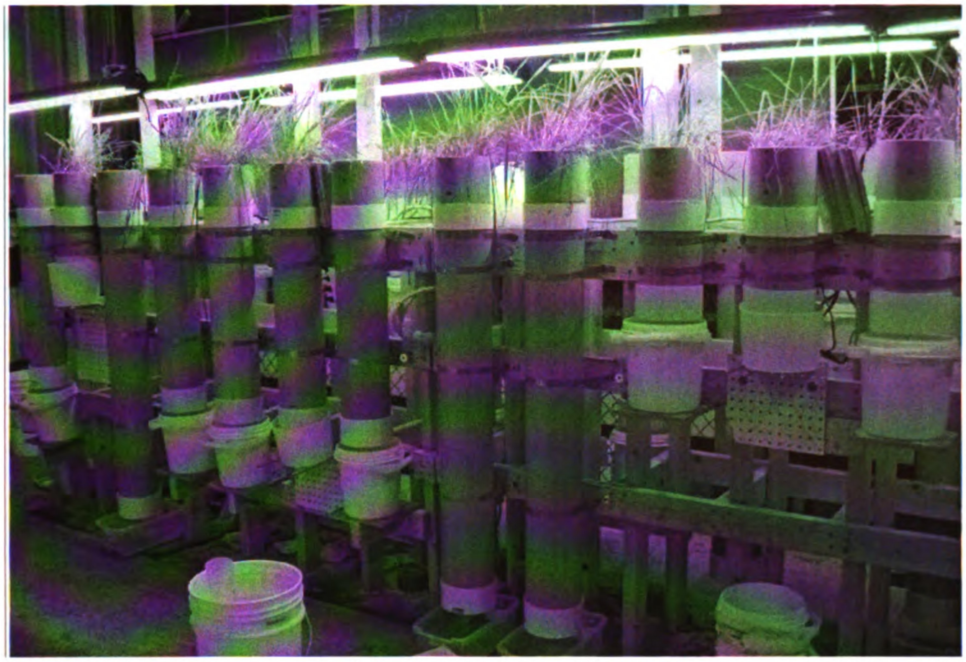

Figure 6: Soil Column Construction

Buckets located directly beneath the columns collected effluent. Wastewater was fed by hand in two doses one followed directly by the second, as the 2 inch free board excess could not hold the entire 1.4 L in one application (discussed further in section 3.3.4). A single batch of simulated wastewater was prepared at the time of application to ensure uniform loading among columns. During feeding the wastewater was mixed prior to application on each column to maintain even dispersion of pollutants.

3.3.3 Waste Water Composition

A synthetic wastewater was used to provide the carbon and nutrients required by the microbial biomass. BOD concentrations were achieved by adding D-glucose (dextrose) to dechlorinated tap water. The estimated oxygen demand of glucose for BOD is 70% of the theoretical O_2 (Gray 2005) as calculated using Equation 1 and 2. Seventy percent of this oxygen demand is then used to estimate the BOD_u for glucose, Equation 3.

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6H_2O \qquad Eqn.\ 1$$

$$1g\,C_6H_{12}O_6\left(\frac{192\frac{g\,O_2}{mol}}{180\frac{g\,C_6H_{12}O_6}{mol}}\right) = 1.07\,g\,O_2 \qquad Eqn.\ 2$$

$$(1.07\,g\,O_2)(0.70) = 0.75\,g\,BOD_u \qquad Eqn.\ 3$$

Consequently, 1 g of glucose produces 0.75 g of BOD. To achieve the desired organic loading of 75 lbs BOD/acre 2 times per week, the simulated waste water was designed with an average BOD concentration of 225 mg/L. A hydraulic load of 1.4 L/day with a glucose concentration of 300 mg/L will be applied 2 days a week to achieve a loading of 75 lbs BOD5/acre twice a week, Equation 5.

$$\left(\frac{1g\,C_6H_{12}O_6}{0.75\,g\,BOD_5}\right)\left(\frac{225\,mgBOD_5}{L}\right) = 300C_6H_{12}O_6\frac{mg}{L} \qquad Eqn.4$$

$$\frac{\left(\frac{\frac{75\,lbs\,BOD_5}{acre}}{day}\right)\left(2\pi(3\,in)^2\right)}{300C_6H_{12}O_6\frac{mg}{L}} = \frac{1.4\,L}{day} \qquad Eqn.5$$

The synthetic wastewater was prepared according to Trulear and Characklis (1982) to provide the essential micro and macro nutrients for the microbial biomass. Table 4 details the nutrient solution composition which has been proportionally adjusted according to BOD concentrations. One substitution from the nutrient solution is MnO_2 to replace $MnCl_2$. MnO_2 adds manganese to the soil columns as Mn(IV), an immobile form of manganese, as an objective of the research is to determine reduction from the immobile form to the mobile form, Mn(II). Constant agitation during application is required to distribute this chemical as in this form it is insoluble.

Table 4: Nutrient Solution Constituent Concentrations for Synthetic Wastewater

Constituent	Trulear and Characklis (1982) Concentrations (mg/L)	Filter Strip Soil Column Concentration (mg/L)
$C_6H_{12}O_6$	10	300
$FeCl_3$	0.045	1.35
MnO_2	0.005	0.15
$ZnSO_4 \cdot 7\,H_2O$	0.008	0.24
$CuCl_2 \cdot 2\,H_2O$	0.005	0.15
$CoCl_2 \cdot 6\,H_2O$	0.007	0.21
$(NH_4)_6\,Mo_7\,O_{24} \cdot 4\,H_2O$	0.005	0.15
$Na_2B_4O_7 \cdot 10\,H_2O$	0.003	0.09
$Na_3C_6H_5O_7 \cdot 2\,H_2O$	0.408	12.24
$NaH_2PO_4 \cdot H_2O$	0.575	17.25
$(NH_4)_2\,SO_4$	0.367	11.01
$NH_4\,Cl$	3.417	102.51
$CaCl_2$	0.308	9.24
$MgCl_2 \cdot 6\,H_2O$	0.565	16.95

3.3.4 Soil Column Operation

Columns were fed simulated wastewater for 7 months twice per week, on day 1 (Monday) and day 4 (Thursday) to allow for drying between applications. Control columns were fed 1.4 L of dechlorinated tap water coinciding with wastewater application. Declorination was achieved using a chemical chelating agent commonly used for fish tanks. The dechlorinating agent was added then mixed, and then chemicals added as described in the synthetic wastewater section above. Column influent and effluent was collected bi-weekly following wastewater application on day 4. Twice a week 10-12 hours after feeding (as was determined to be the time required for columns to leach the entire wastewater volume) the effluent was measured for volume and ambient air temperatures recorded. Air temperature was assumed to be the soil temperature as size permitted columns to equilibrate. Prior to wastewater application, water was removed from all submerged columns for effluent collection. After effluent collection and the leaching of all the wastewater volume, the soil columns were then re-submerged. Samples were prepared for laboratory analysis according to the QA/QC procedures outlined section 3.1.2 and in Appendix B.

3.3.5 Water Quality Parameters

Nutrient removal evaluation required lab analysis for nitrogen and phosphorus. Nitrogen measurements included TKN, ammonia, nitrate, and nitrite to assess the full nitrogen cycling as well as the impact on the other various soil-water biogeochemical processes. Oxygen requirements were measured via the BOD_5 and COD. Metals were analyzed to evaluate the reduction potential and metal leaching. Manganese was present within the system as Mn(IV) and Iron as Fe(III). The reducing conditions result in conversion from these insoluble forms to soluble forms, Mn(II) and Fe(II). Measurement of the influent and effluent allowed for determination of the redox conditions within the columns and the loading conditions that result in leaching of metals. For a detailed list of the parameters measured and the methods for their collection and analysis see Table 5.

Table 5: Water Quality Analysis Parameters and Methods For Soil Columns

Parameter	Method	Detection Limit	Hold Time
Alkalinity (mg/L $CACO_3$)	USEPA 310.1 (1)	10 mg/L CaCO3	24 hours
BOD_5 (mg/L)	USEPA 405.1 (1)	2 mg/L	Analyze Immediately
Iron	SW-846 method 6010B (2)	0.02 mg/L	6 months
Manganese	SW-846 6020 (2)	5 µg/L	6 months
NH_3 (mg/L)	USEPA 350.3 (1)	0.02 mg/L NH3-N	28 days
NO_2 (mg/L)	USEPA 354.1 (1)	0.002 mg/L NO2-N	24-48 hours
NO_3 (mg/L)	USEPA 353.3 (1)	0.3 mg/L NO3-N	28 days
pH			Analyze immediately
TKN (mg/L)	USEPA 351.1 (1)	1 mg/L	28 days
COD (mg/L)	USEPA 410.4 (1)	1 mg/L	28 days
TP (mg/L P)	USEPA 365.1 (1)	0.02 mg/L P	28 days

(1) (US EPA 2009)
(2) (US EPA 1996)

3.3.6 Soil Column Deconstruction

Before deconstructing, a soil column flow rates study was conducted over two feeding periods. Volumes were recorded every 3 minutes for short columns and every 15 min for the longer columns increasing to every half an hour after an hour. The volumes were then divided by the time interval to obtain an average flow rate for each time segment. After effluent sampling, one of each replicate column was deconstructed and the soil was sampled every six inches to determine the fate of metals within the columns. Soils samples were digested and analyzed for Mn, Fe, and COD at the State of Michigan Environmental Laboratory.

3.3.7 Data Analysis

Statistical analysis was conducted using ANOVA in SAS, with time as a repeated measure, to determine the statistical significance of depth, soil, and submergence on each measured water quality parameter. Assumptions that residuals are normally distributed and the variances are homogenous were evaluated using normal probability plots and side-by-side box plots to ensure their validity, and adjusted using grouping and data transformations when necessary. When the ANOVA was significant, difference of least squares means was used to compare the treatment means and their interactions. Statistical results in addition to treatment averages and percent reduction allowed for evaluation of research objectives.

CHAPTER 4: RESULTS AND DISCUSSION

Studies were carried out based on the designs outlined in the previous section. Results for each of the three sections, comparisons between field and column performance, and design implications are reported in this chapter.

4.1 Runoff Characterization

Runoff results for 9 storm events from July 2008 through May 2009 were analyzed at six sampling locations. Average concentrations at each sampling location for 17 water quality parameters are in Table 6. Storage basin 1 primarily receive wastewaters from the heat check lot and roadway, while storage basin 2 collects runoff diverted from the bunker and upright silos.

Table 6: Feedlot Runoff Water Quality Parameter Average Concentrations

Location	pH	Alkalinity (mg/L)	PO_4^- (mg/L)	COD (mg/L)	BOD5 (mg/L)	Ammonia (mg/L)	NO_2^- (mg/L)	NO_3^- (mg/L)	TKN (mg N/L)	SO_4^- (mg/L)
Bunker Silo	5.87	277	17	2320	900	18	0.14	14	115	12
Heat Check Lot	8.35	940	17	3180	930	58	0.12	13	355	112
Roadway	6.37	168	11	1380	410	10	0.08	5	54	21
Upright Silo	6.61	193	16	1910	730	28	0.46	7	101	20
Storage Basin 1	7.06	445	8	790	240	23	0.08	8	54	11
Storage Basin 2	5.21	136	20	2510	1180	36	0.22	4	74	7

Location	TS (mg/L)	VS (mg/L)	TSS (mg/L)	VSS (mg/L)	Mn (µg/L)	Fe (µg/L)	TOC (mg/L)	Conductance (umhos/cm)	Cl (mg/L)	As (µg/L)
Bunker Silo	2490	1310	410	190	418	3950	990	1342	39	4.9
Heat Check Lot	4910	2060	1030	800	491	2530	1250	6730	526	4.3
Roadway	1880	1340	370	250	216	1180	440	836	39	1.7
Upright Silo	1540	1210	780	540	221	3650	570	1124	24	11.8
Storage Basin 1	2490	900	450	270	216	3200	210	1226	151	2.3
Storage Basin 2	2970	910	310	250	411	5560	790	1384	26	3.0

Statistical analysis of water quality parameters (with a minimum of 8 complete data sets from the 9 total sampling events) was generated using SAS software (SAS 2008) to determine statistical differences in the mean source concentrations. Statistical models were fit using ANCOVA to determine if covariates of season and rainfall reduced the overall error within the model. If covariates did not reduce the experimental wide error for each water quality parameter assessed, the covariates were eliminated. If ANCOVA or ANOVA was determined to be statistically significant for each parameter then the model was evaluated for comparison of the treatment means for main effects for location using differences of least squares means. Analysis results for each parameter are outlined below.

Runoff from animal waste and feed contain high concentrations of COD, as can be seen by the values for the heat check lot and the silo locations, Figure 7. Average values for animal waste were near 3,000 mg/L while feed average concentrations were approximately 2,000 mg/L. Although there were large differences within the averages of these sources, there was not a statistical significance between the two COD sources over the length of the study. However, there was a statistical difference between COD concentrations from the heat check lot to that of the roadway and storage basin 1. This indicates that although the concentrations from the heat check lot were high, the dilution from the roadway runoff was significant enough to impact the composite storage basin concentrations. The second storage basin has a greater COD concentration as the source locations, bunker and upright silos, have significant COD contributions. Analysis of BOD_5 produced similar results although had only 5 complete data sets.

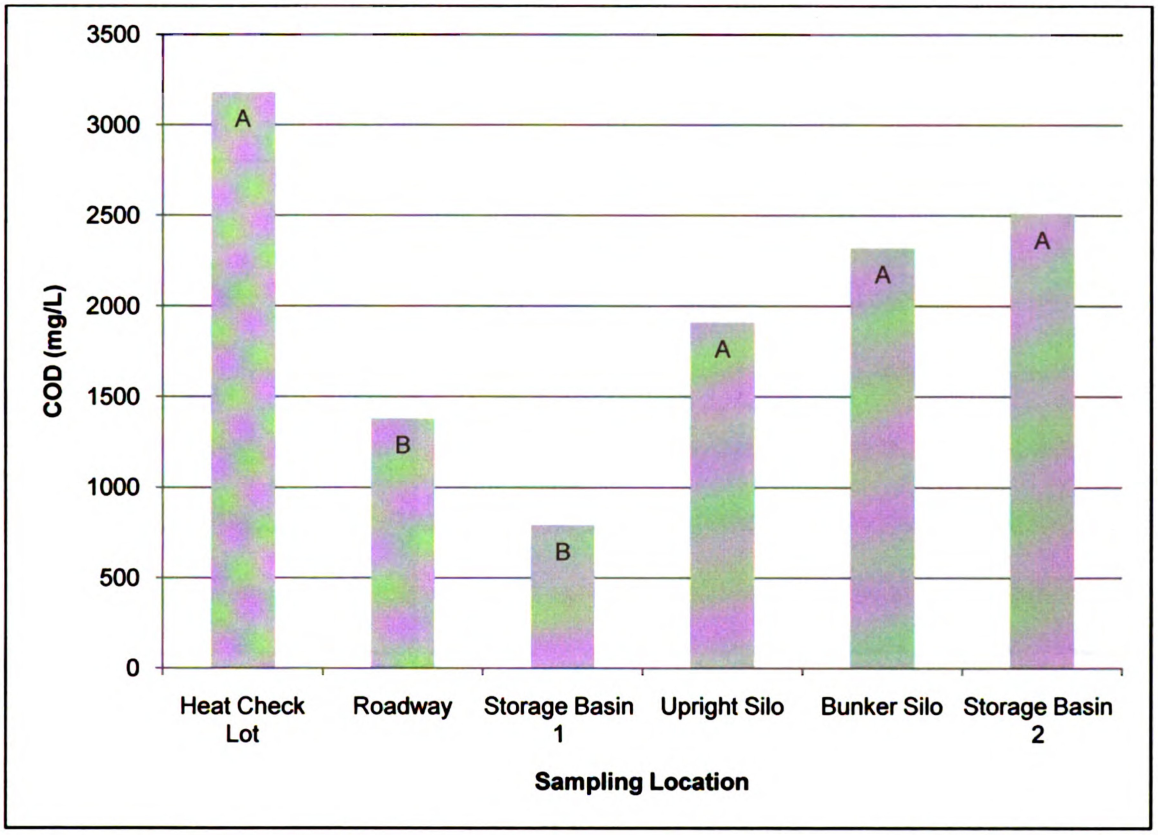

Figure 7: Average COD Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Phosphorus mean values for runoff sources are similar in concentration and produced statistically significant differences for the mean values for storage basin 1 and 2, Figure 8. Again, storage basin 1 is statistically different from the heat check lot taking on the phosphorus characteristics of the roadway, indicating phosphorus is also dependent upon dilution and runoff quantity.

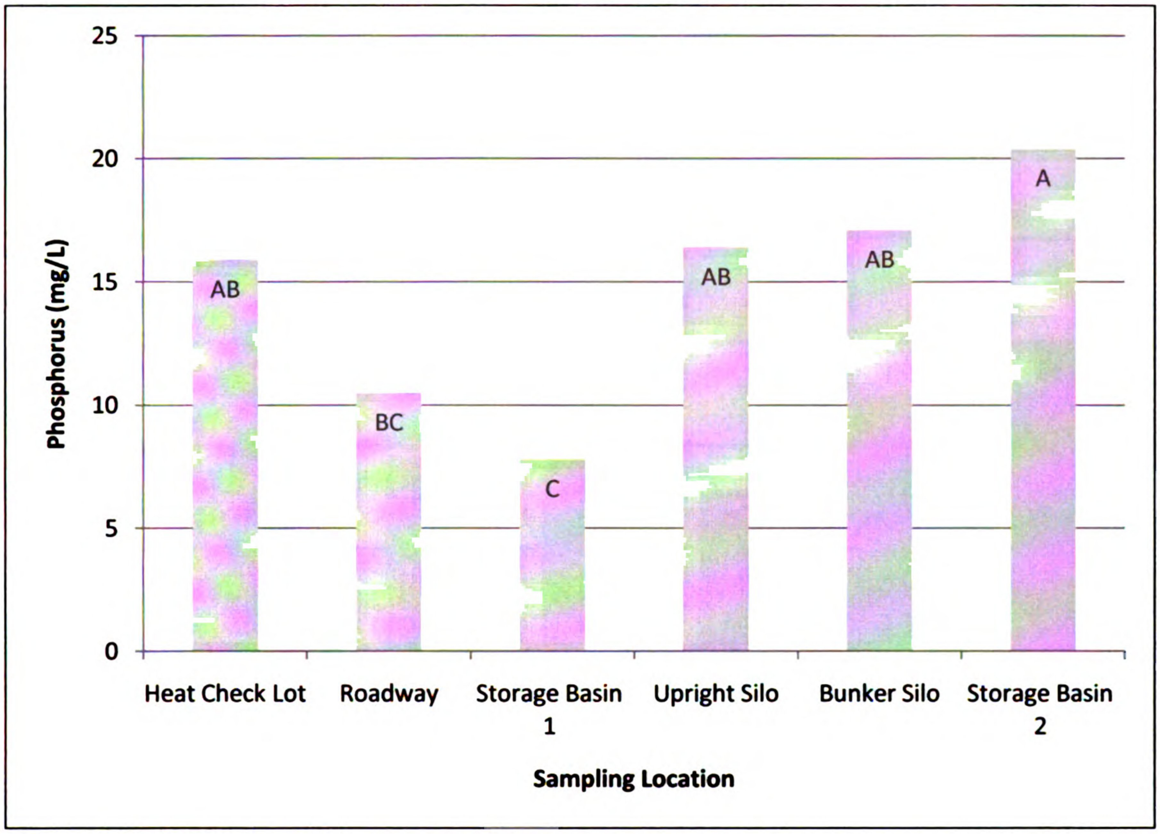

Figure 8: Average Phosphorus Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

The heat check lot produces significantly higher total solids concentrations than all other sources, Figure 9. Notched grooves for animal footing and safety within the heat check lot concrete reduces the impact of scraping and allows build-up of animal waste, the likely source of the solids concentrations. A more effective cleaning technique or diversion of rainwater from manure is required to reduce solids concentrations from this source. However, in this case the farm may not benefit from additional maintenance as the composite sample again takes on the characteristics of the roadway runoff in regards to solids concentrations.

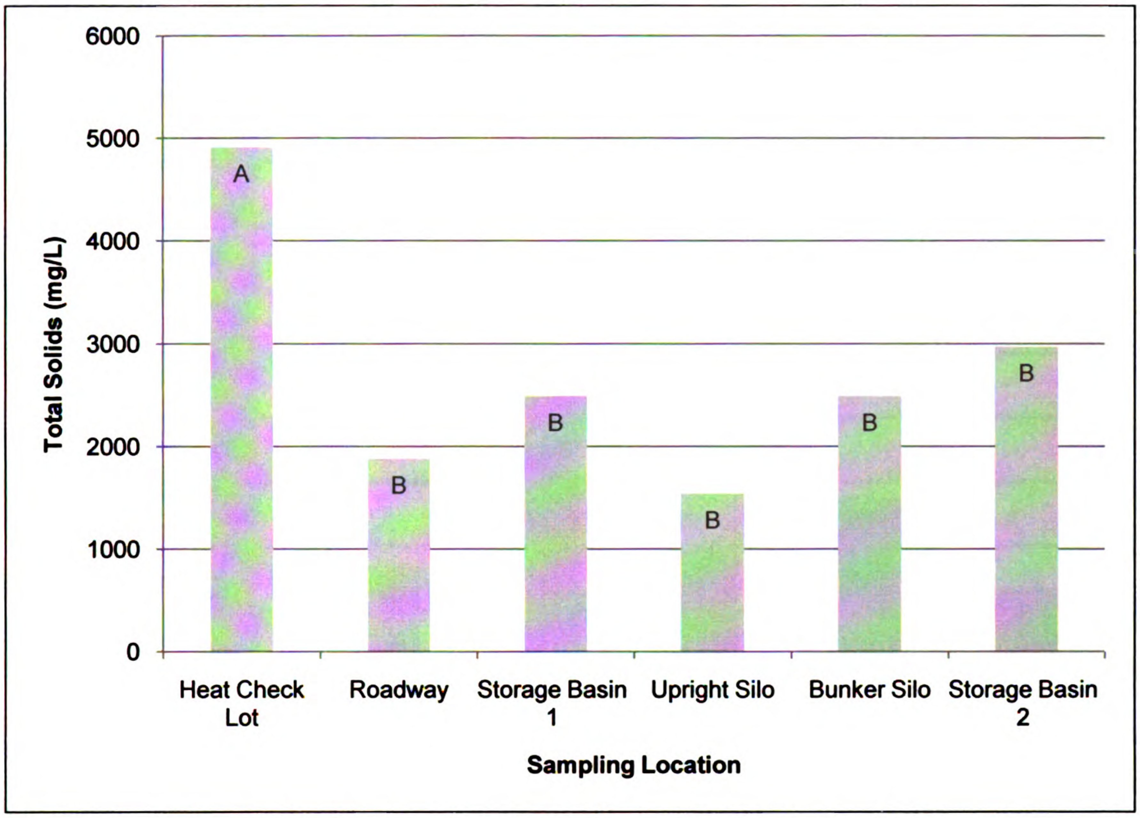

Figure 9: Average Total Solids Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Alkalinity concentrations in the heat check lot are significantly greater than all other source locations, Figure 10. This supports the findings that the heat check lot is largely affected by the manure concentration as liquid dairy manure has an alkalinity of over 4,000 mg $CaCO_3$/L (Debusk et al. 2007). All other source locations and basins did not produce statistically different mean concentrations.

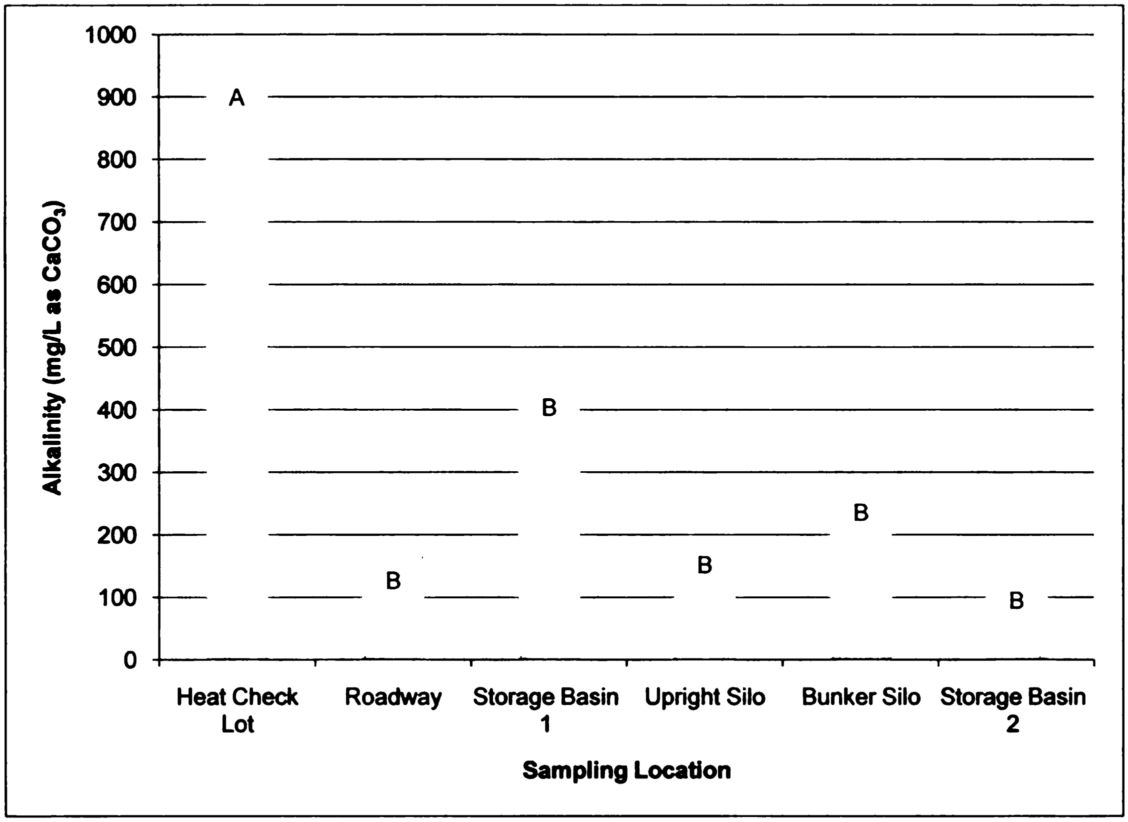

Figure 10: Average Alkalinity Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Heat check lot values produced an alkaline average pH value of 8.35 due to manure concentrations. Acidic pH concentrations were produced from feed sources, Figure 11. Unlike previous parameters, storage basin 1 was significantly different from both the heat check lot and the roadway. The two parameters combined to produce a composite concentration that was between the two mean values from the source locations, and was not impacted to the degree other parameters were from runoff quantity and dilution. Storage basin 2 is significantly different from the upright silos but not from the bunker silos, revealing the composite pH is affected directly by the low acidity from the bunker silos. This low pH poses potential problems to biological treatment, however eliminates *E. Coli* at a pH below 5 (which was common in storage basin 2) but did not fully eliminate all *Coli forms.*

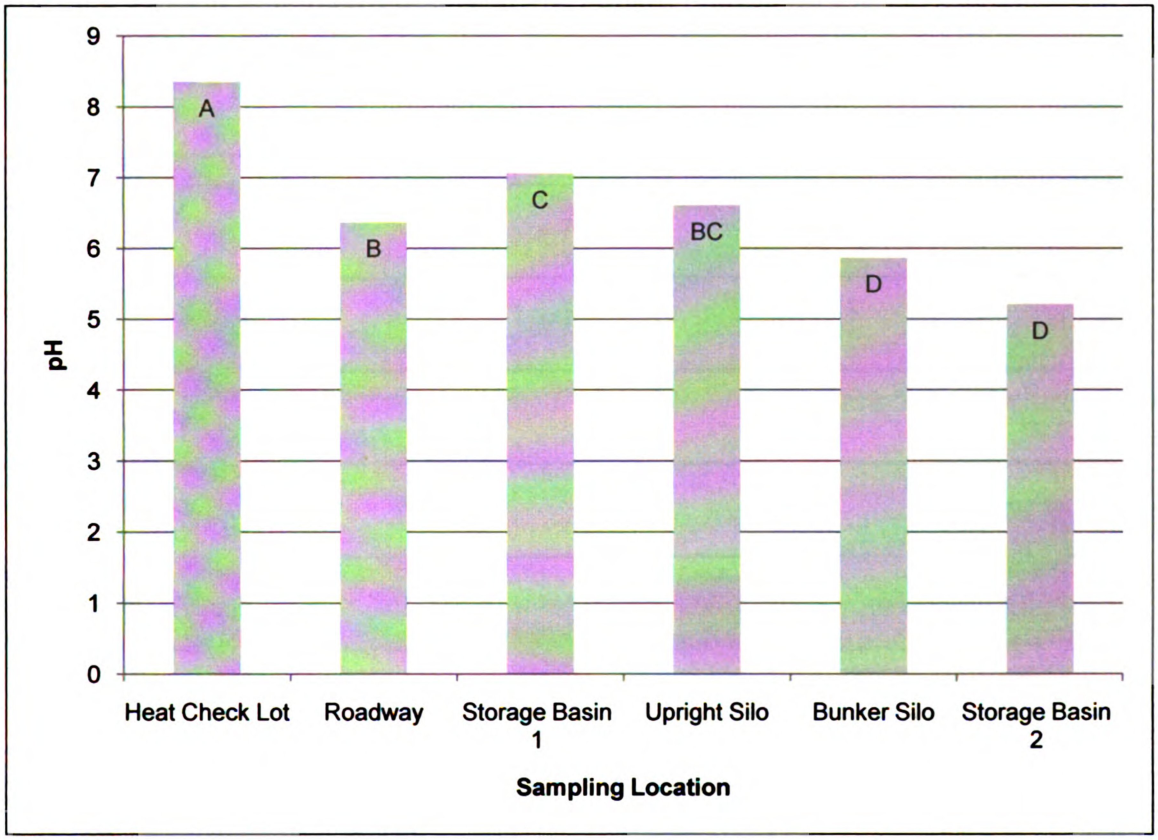

Figure 11: Average pH Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Ammonia and TKN concentrations were greatest in the heat check lot, with organically bound nitrogen and ammonium as the majority of the total nitrogen in the system. Statistically, the heat check lot mean for ammonia and TKN is significantly different from the roadway and storage basin 1, with two times the concentration of ammonia than any other source, but again not contributing significantly to the composite sample, Figure 12. Storage basin 2 in this case is governed by the upright silo as the concentrations within the basin are statistically similar to those from the upright silo. The bunker silos do not produce as high of concentrations of ammonia in runoff as the upright silos, but do not reduce the composite concentrations within the basin.

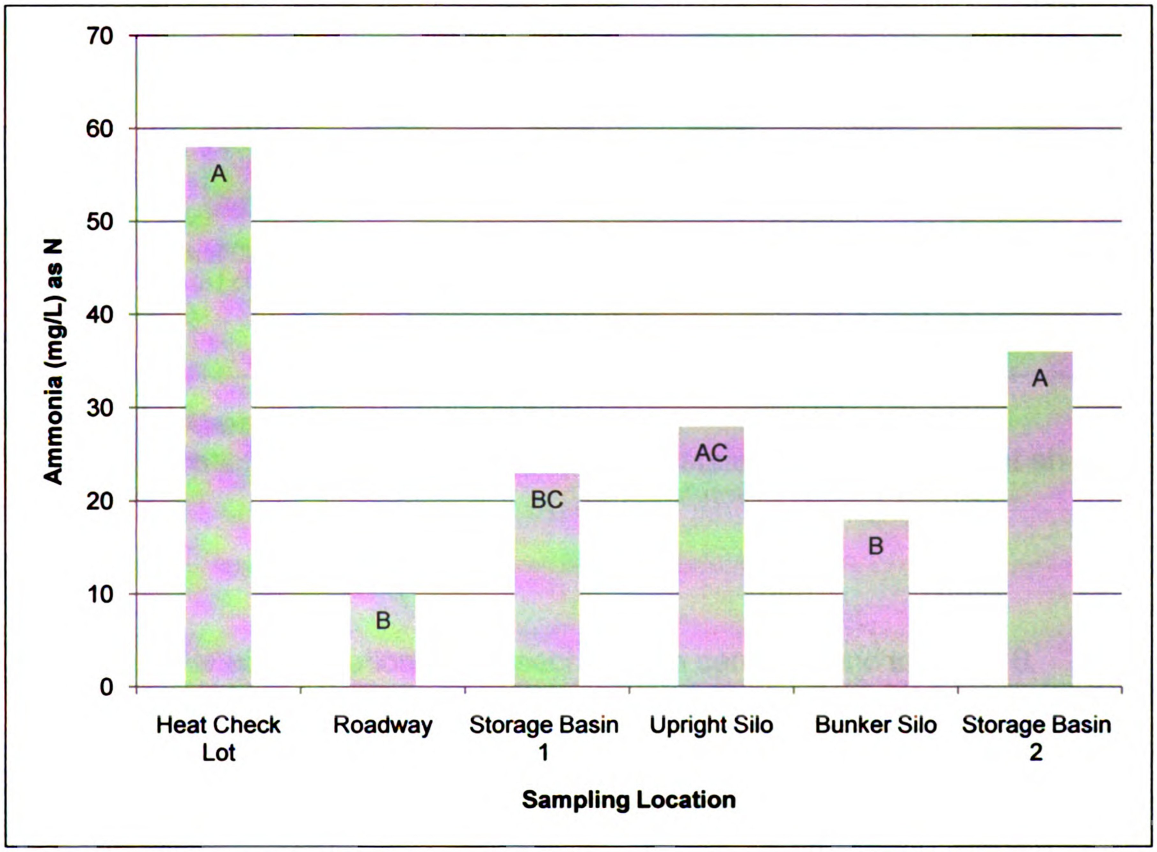

Figure 12: Average Ammonia Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Storage basin 2 concentrations for TKN are governed by the bunker silo concentrations as they are statistically similar, Figure 13.

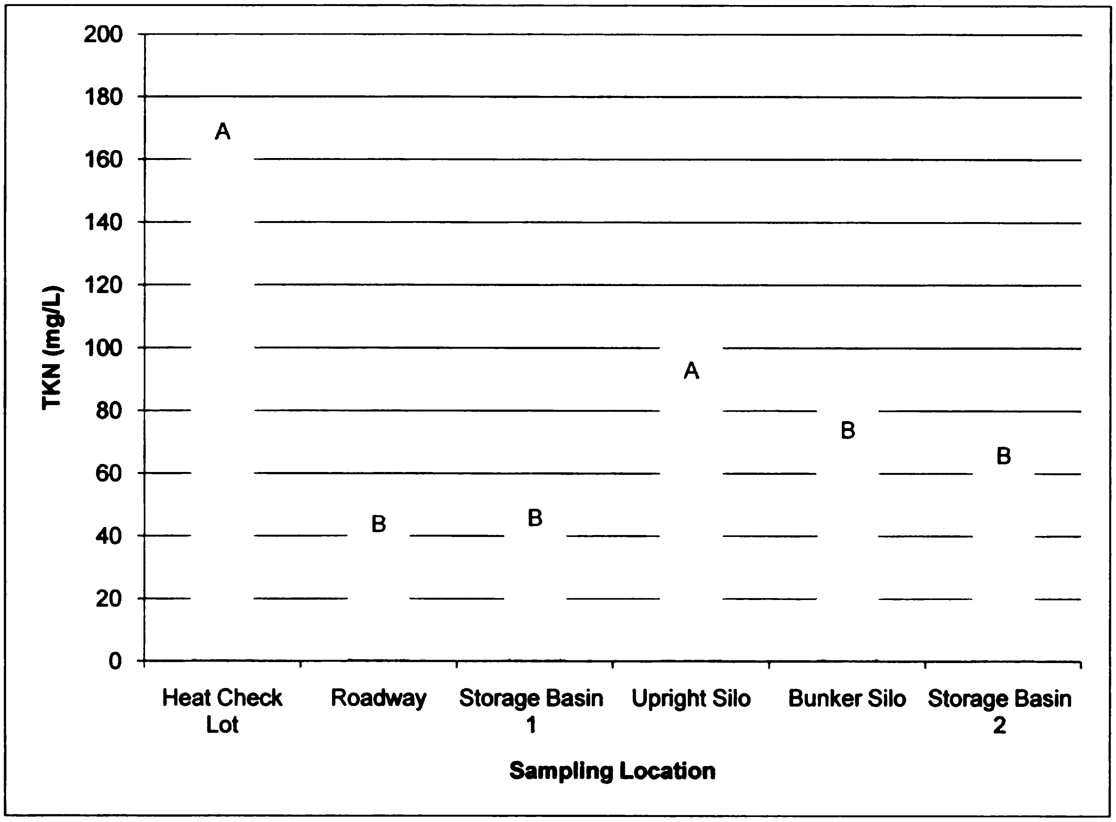

Figure 13: Average TKN Concentrations.
Sampling locations with the same letter are not statistically significant at an alpha value = 0.05.

Average metal concentrations from each source can be found in Table 6. Manganese concentrations were two times as great from the heat check lot and bunker silo than other locations. Arsenic concentrations are below water standards for all sources except the upright silos which have an average concentration of 11.8 ug/L, above the 10 ug/L US EPA drinking water maximum (US EPA 2009b).

4.1.1. Summary and Management Strategy

In summary, the heat check lot produces the largest concentrations for nearly all water quality parameters (Table 6) suggesting that animal waste on farmstead operations is the leading source for water quality issues. However, when examining composite samples in the storage and sedimentation basins that take into account water quantity in addition to water quality, feed sources are a greater concern at this farm. The footprint of the heat check lot is 5000 sq ft, bunker silos are 3100 sq ft, and upright silos 11,350 sq ft. The remaining farmstead area is impervious roadways. The heat check lot is 9% of the storage basin 1 drainage area. In regard to basin 2, the bunker silo is 23% and the upright silos 6% of the drainage area. If farmstead manure locations are at or below 9% of the drainage area and the remaining area has low concentrations similar to that of the roadway area, then management requirements for animal waste sources are low. The greater footprint of the silos in addition to the high pollutant concentrations make the feed sources a greater focus for farmstead maintenance resources.

On-farm management practices should focus on feed sources to limit the impact of runoff on water quality. In addition, runoff from upright silos contains higher pollutant loads in the fall in comparison to the spring due to filling practices. The fall months produced concentrations that are 10 times greater than those measured in the spring (data not shown). Note that the concentrations listed in Table 6 are the averages for all seasons. Properly loading silage, including harvesting at the correct temperature and moisture, rapid filling, and proper compaction, can improve silage quality and reduce leachate production (Saxe, 2007). Bunker silo runoff in general produces greater pollutant loads than upright silos (if the loading of the upright silos is managed properly). To minimize this impact, it is important to cover bunker silage prior to precipitation events, sweep impervious areas around feed sources, and maintain feed faces.

On typical farm operations when manure may be a larger source of concern, additional steps should be taken in combination to those listed above for feed sources. If possible, manure should be covered and/or berms or curbs provided to limit transport. Increasing vegetation in drainage areas with overland flow can also decrease transport of pollutants to treatment systems. Installation of gutters for diversion of clean water will reduce the volume required for treatment, therefore reducing the size and cost of implementation. Lastly, care should be taken to eliminate dry weather leachate and ensure farmstead operations other than runoff are not increasing the load to treatment systems.

4.2 Analysis of Field Treatment Systems

Application of farmstead runoff and treatment evaluation of 3 full-scale agricultural filter strips was conducted from 9/08/2009 to 6/24/2010. Ten sampling events were completed at 2 filter strips located at the MSU dairy ranging from 0.05-1.24 inches of rainfall, an additional 6 sampling events were investigated at the small MI dairy ranging from 0.04-1.71 inches of rainfall.

Surface water and subsurface effluent were measured to determine percent removal for 17 water quality parameters. All raw filter strip data can be found in Appendix D.

4.2.1 BOD

BOD removal for all three filter strips was relatively equal for surface water and subsurface effluent, Table 7 & 8. The greater differences within the removal percentages at the MSU site were negligible as variation between samplings was large as indicated by the standard deviation. Influent loading to filter strip 1 ranged between 35-270 lbs BOD_5/acre/rain event and loading to filter strip 2 between 140-910 lbs BOD_5/acre/rain event. Note the surface removal percentages for the small MI dairy represent reduction percentages from the bioretention basin.

Table 7: BOD_5 Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	37%	15%	54%	24%	-4%	14%	5%	-20%
1 ft	28%	19%	45%	2%	-6%	36%	20%	-31%

Table 8: BOD_5 Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	72%	4%	76%	68%
1.5 ft	89%	1%	91%	88%
2.5 ft	79%	5%	87%	76%

Influent concentrations were similar for the small MI dairy site and for filter strip 2 at the MSU site, both with average concentrations of 1300 mg/L BOD_5, whereas filter strip 1 at the MSU site had a much lower influent average of 230 mg/L BOD_5. This resulted in loadings of The difference in influent concentrations is due to dilution of wastewater at filter strip 1 and build-up of contaminants on impervious surfaces at the other 2 locations. Final BOD_5 subsurface effluent concentrations were similar for filter strip 1 and the small MI dairy filter strip at ~150 mg/L. Filter strip 2 had an average subsurface effluent of almost 2500 mg/L BOD_5, and increase in concentration. The sandy soil at the small MI dairy site which results in a reduction in the soil water holding capacity and an increase in oxygen diffusion is hypothesized as the cause for the increase in oxygen content and the greater pollutant reduction.

4.2.2 COD

COD removal at the MSU dairy site was even less than that for BOD_5, Table 9. Average influent concentrations were again similar for the MSU filter Strip 2 and the small MI dairy filter strip at 4700 mg/L COD and 4400 mg/L respectively. The first MSU filter strip had reduced influent COD concentrations at ~450 mg/L. Higher COD concentrations as compared to BOD_5 concentrations indicate the presence of recalcitrant carbon, most likely cellulose or lignin materials (Nielsen 2003).

Table 9: COD Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	8%	30%	70%	-19%	-38%	54%	29%	-109%
1 ft	18%	17%	49%	1%	4%	33%	57%	-30%

The COD subsurface effluent at the small MI dairy site sustained removal percentages above 70% for all sampling events, and performed much more consistently with reduced standard deviations, Table 10. Again, the greater removal rates are thought to be due to the sandy soil and increased porosity which increases oxygenation and diffusion rates, the decrease in soil moisture holding capacity, and the reduced influent flow rates.

Table 10: COD Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	59%	31%	87%	-1%
1.5 ft	85%	9%	96%	70%
2.5 ft	86%	8%	97%	78%

Effluent COD concentrations at the MSU dairy site have a linear trend when plotted as a function of influent COD concentrations, resulting in increased effluent concentrations when influent concentrations increase. The small MI dairy data does not follow this same linear trend. In the case of the sand soil, the effluent COD concentrations remain relatively constant for all COD influent concentrations, Figure 14.

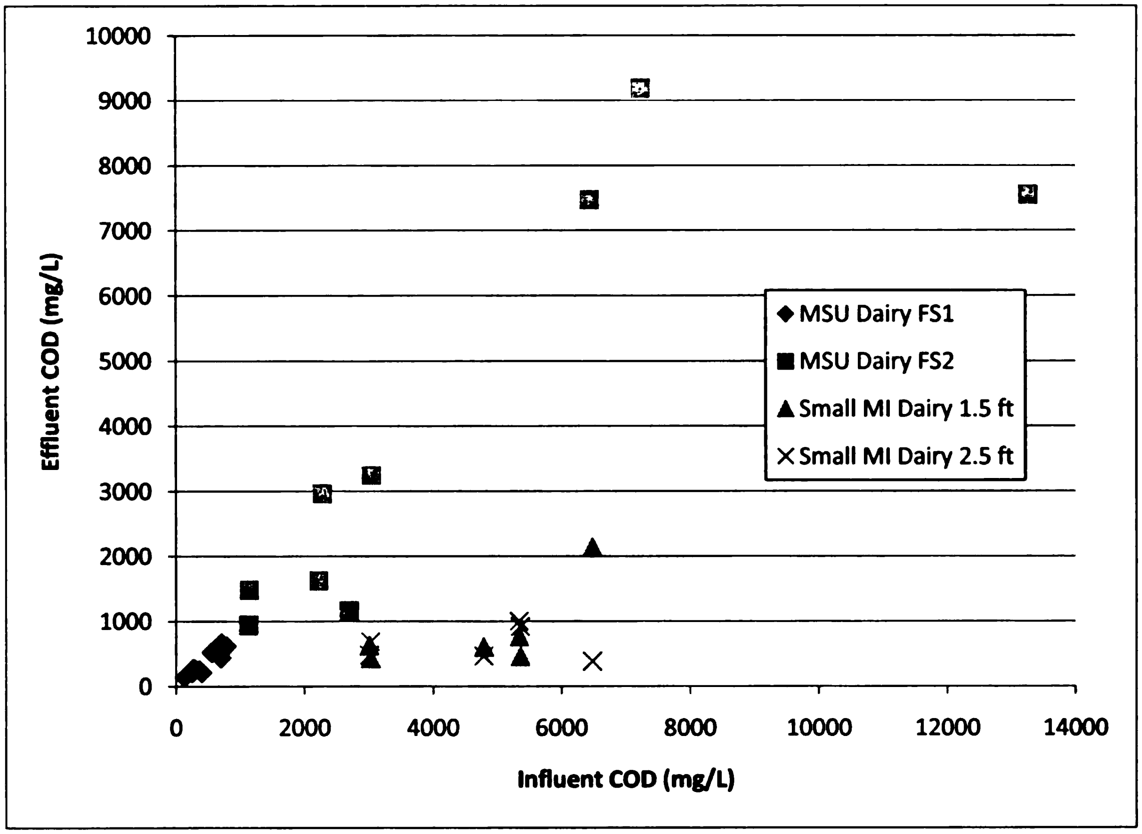

Figure 14: COD Effluent Concentrations as a Function of Influent COD Concentrations

The trends within this data indicate that there is a something rate limiting at the MSU site resulting in the increased effluent concentrations.

The small MI dairy had greater removal rates for soluble COD, Table 11 & 12, as compared to the MSU filter strips, consistent with the BOD_5 and COD removal. Soluble pollutant concentrations are more difficult to remove as indicated by the literature review in previous sections and the small MI dairy removal percentages for COD and soluble COD. The MSU dairy filter strips performed poorly and inconsistently for both COD and soluble COD with removal percentages below 30% for all locations.

Table 11: Soluble COD Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-2%	35%	48%	-70%	-21%	24%	-2%	-52%
1 ft	26%	15%	46%	7%	15%	35%	63%	-29%

Table 12: Soluble COD Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	23%	24%	45%	-2%
1.5 ft	62%	25%	76%	34%
2.5 ft	59%	7%	65%	51%

4.2.3 Nitrogen

TKN concentrations are the sum of organically bound nitrogen, ammonium, and ammonia. Nitrogen that originates as organically bound nitrogen must undergo ammonification to convert organic nitrogen to inorganic forms. These inorganic forms can then undergo the process of nitrification in aerobic conditions followed by denitrification under anaerobic conditions to exit the treatment system as nitrogen gas. Reductions in TKN were not constant for the MSU dairy locations, but were consistently above 80% at the site, Table 13 & 14.

Table 13: TKN Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	18%	14%	35%	5%	-7%	36%	23%	-54%
1 ft	32%	16%	52%	10%	20%	30%	61%	-19%

Table 14: TKN Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	51%	11%	62%	34%
1.5 ft	87%	2%	89%	85%
2.5 ft	83%	10%	94%	67%

Again the effluent TKN concentrations at the MSU dairy farm are dependent upon the influent concentrations, as represented by a general linear trend. The small MI dairy maintains similar effluent concentrations regardless of influent concentrations, Figure 15. These characteristics hold true for COD and TKN as shown, but also hold true for Ammonia, TOC, arsenic, and solids (data not shown).

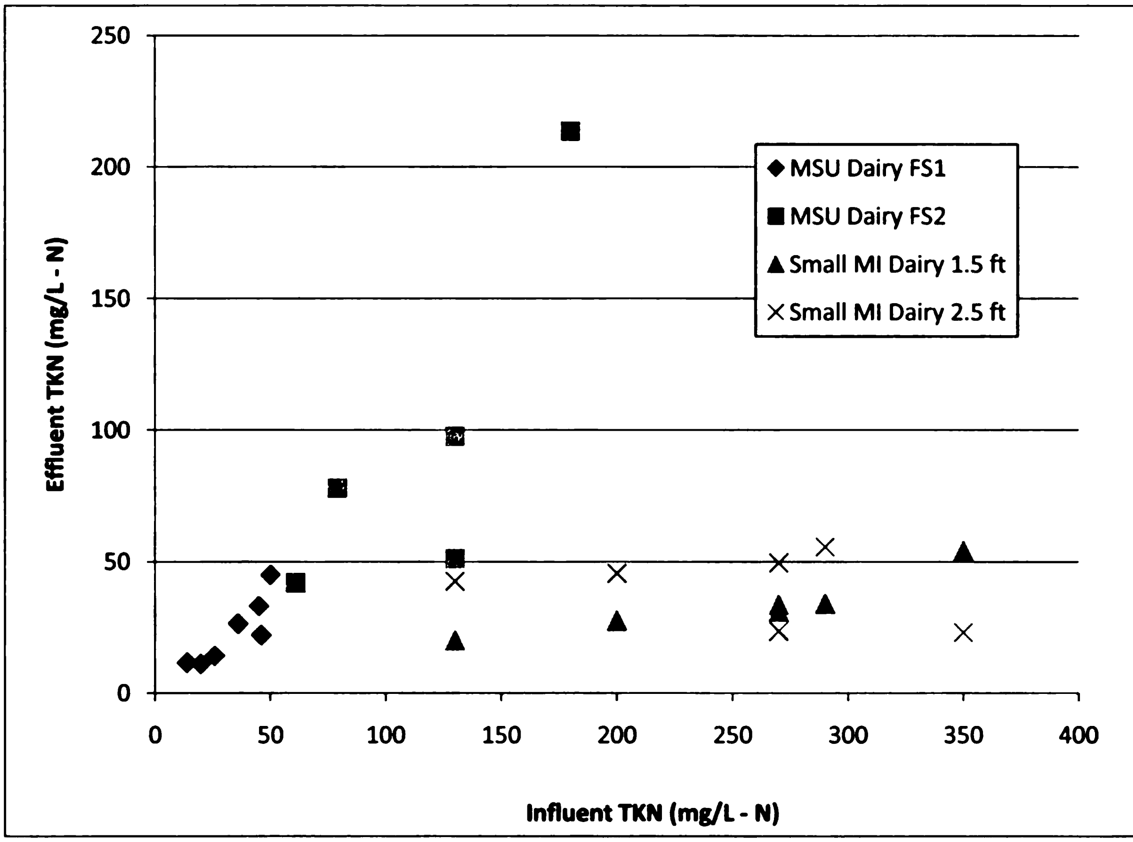

Figure 15: Effluent TKN Concentrations as a Function of Influent TKN Concentrations

Initially, 40% of the TKN within the MSU treatment systems is in the form of ammonia, this fraction increases over the surface of the soil then reduces again as the runoff infiltrates through the soil profile. This indicates ammonification is not occurring at the same rate as conversion of ammonia to other nitrogen forms. The small MI dairy also has 40% of the total TKN as ammonia, and increases steadily as the runoff moves over the surface and infiltrates. Reductions within the first foot of the soil indicate there may be sufficient oxygen for at least a portion of the ammonia to go through the nitrification process. Limited reduction in filter strip 2 in comparison to filter strip 1 can be a source of the decreased pH in filter strip 2 and the high levels of ammonia over 20 mg/L, which are reported as inhibitory for NO_2-N oxidation, all reducing nitrification rates and leading to nitrite build-up (Tchobanoglous et al. 2003).

Table 15: Ammonia Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-2%	32%	33%	-56%	-52%	55%	11%	-139%
1 ft	30%	37%	78%	-29%	22%	22%	49%	-17%

The small MI dairy site saw a greater removal of ammonia within the system, on the surface and within the soil, Table 16. This supports the hypothesis that the sandy soil (and the associated mechanisms which decrease water content) and reduction in influent flow in comparison to the MSU site increases the available oxygen therefore increasing nitrification. At a depth of 2.5 ft however, there is a slight decrease in the removal as average ammonia concentrations climb from 14 mg/L-N at 1 ft to 26 mg/L-N at 2.5 ft indicating oxygen may become limiting at an increasing depth.

Table 16: Ammonia Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	41%	13%	62%	31%
1.5 ft	84%	6%	94%	76%
2.5 ft	73%	13%	93%	57%

Nitrite concentrations increased for all filter strips examined within the study resulting in negative removal percentages, Table 17 & 18. Increases in nitrite concentrations indicate conversion of ammonia to nitrite, but the increases confirm that the conversion from nitrite to nitrate is not occurring at the same rate. The high level of accumulation indicate oxygen limiting conditions at the MSU dairy site as *Nitrobacter* is more effected by low dissolved oxygen concentrations, resulting in increases nitrite concentrations (Tchobanoglous et al. 2003). The greater increase at the second filter strip site is likely due to the low pH levels which inhibit denitrification (Sahrawat 2008; Yue-Mei et al. 2008; Tchobanoglous et al. 2003

Table 17: Nitrite Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-950%	2301%	93%	-5645%	-1494%	2661%	80%	-4567%
1 ft	-649%	1530%	59%	-4107%	-1203%	1347%	-63%	-2689%

Table 18: Nitrite Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	89%	19%	100%	67%
1.5 ft	-20%	19%	-6%	-33%
2.5 ft	-16%	118%	68%	-100%

Nitrate concentrations show a large variability, Table 19 & 20. Final average nitrate concentrations after infiltration are 11 mg/L-N for filter strip 1, 45 mg/L-N for filter strip 2 at the MSU site and 25 mg/L-N at the small MI dairy. These pose human health concerns and exceed the US EPA drinking water standard of 10 mg/L (US EPA 2009b).

Table 19: Nitrate Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	1%	73%	78%	-86%	-208%	446%	53%	-1108%
1 ft	26%	37%	62%	-17%	-164%	178%	63%	-413%

Table 20: Nitrate Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	-105%	212%	29%	-350%
1.5 ft	4%	83%	75%	-131%
2.5 ft	36%	43%	78%	-32%

Nitrification and denitrification are occurring within the soil as removal of the sum of all nitrogen species in all forms from runoff after infiltrating the soil profile varies from 25% to 80% for the full-scale system. Nitrate concentrations pose potential problems for groundwater contamination and may impact the implementation of this practice if improvements on removal cannot be made.

4.2.4 Phosphorus

Average influent phosphorus concentrations are 11 mg/L for filter strip 1, 24 mg/L for filter strip 2, and 94 mg/L for the small MI dairy. The greater concentrations at the small MI dairy site are much larger than reported literature values (Table 1) and are due to the large sources of animal waste and lack of containment or maintenance to control runoff from these sources. Removal rates for phosphorus are low for the MSU dairy, Table 21. The average concentrations for the subsurface effluent are the same as the influent concentrations at the MSU dairy site. As assimilation in soil is the main mechanism for phosphorus removal, it was theorized that removal for this parameter would be negligible over time.

Table 21: Phosphorus Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-89%	169%	11%	-422%	-75%	76%	-16%	-185%
1 ft	-5%	52%	35%	-126%	-25%	71%	48%	-158%

The small MI dairy has significantly greater removal with average subsurface runoff concentrations of 19 mg/L, Table 22. The assimilative capacity of this location will become exhausted over time as was seen with the increasing phosphorus concentration trend in the infiltrate of the MSU dairy filter strips over only a year of sampling (Appendix D).

Table 22: Phosphorus Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	62%	27%	88%	11%
1.5 ft	88%	12%	100%	66%
2.5 ft	80%	21%	100%	55%

4.2.5 Solids

Removal of total solids at the MSU site did not occur. Solids within the surface runoff and the subsurface samples were typically greater than the sampled basins, Table 23. Solids in the basin were allowed to settle prior to sampling which reduced the solids concentration within the basin samples. But, when the pumps were activated, this stirred the sediment located within the basin. There was however, solids settling within the storage basins as the solids had to be removed numerous times throughout the research period. Surface samples also collected sediment from solids which have settled from previous applications and within the rock checks where the surface samples were collected.

Table 23: TS Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-88%	88%	4%	-209%	-26%	8%	-18%	-36%
1 ft	-27%	31%	3%	-62%	-38%	29%	-1%	-79%

The small MI dairy had increased removal for total solids within the soil profile as the soil acted like a filter, Table 24. This may be attributed to the small settling basin that was not mixed and a bioretention basin where the runoff is forced to flow through the soil profile in this unit, 41% of the total solids are removed prior to filter strip application.

Table 24: TS Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	41%	42%	83%	-21%
1.5 ft	67%	14%	84%	50%
2.5 ft	67%	19%	88%	40%

Average influent concentrations for VS are 320 mg/L for filter strip 1, 2420 mg/L for filter strip 2, and 2490 mg/L for the small MI dairy site, indicating nearly 50% of the total solids are volatile indicating a large volume of organic material.

Removal for volatile solids follows the same trends as that for total solids, Table 25 & 26.

Table 25: VS Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	0%	19%	25%	-19%	-27%	21%	-3%	-58%
1 ft	-183%	237%	7%	-521%	-13%	21%	15%	-31%

Again greater removal is realized in the small MI dairy as 60% of the VS are removed after the bioretention basin.

Table 26: VS Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	60%	20%	81%	36%
1.5 ft	78%	14%	92%	61%
2.5 ft	82%	7%	92%	72%

Total suspended solids account for less than 20% of the total solids for all filter strips. TSS and VSS behave in a similar manner as TS and VS, Tables 27 – 30. However, TSS and VSS removal increased with increasing depth. Due to the great increase in the surface TSS and VSS, even the first MSU filter strip subsurface samples (which have negative removals) represent a decrease in the TSS and VSS concentrators.

Table 27: TSS Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-219%	313%	40%	-763%	-31%	74%	74%	-129%
1 ft	-22%	41%	22%	-89%	26%	17%	49%	13%

Table 28: TSS Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	78%	20%	96%	57%
1.5 ft	89%	10%	98%	75%
2.5 ft	92%	6%	98%	84%

Table 29: VSS Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-61%	111%	63%	-200%	-26%	85%	77%	-122%
1 ft	-17%	47%	27%	-83%	10%	42%	40%	-50%

Table 30: VSS Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	72%	24%	94%	48%
1.5 ft	92%	11%	98%	78%
2.5 ft	92%	8%	98%	82%

Solids removal was significantly improved by the addition of the bioretention area at the small MI dairy. This is also reflected in phosphorus removal as phosphorus is commonly sediment bound, and the greater sediment removal results in a significant reduction in phosphorus concentrations. Forty percent of the TS and 78% of the TSS were removed by the bioretention basin which is reflected in the 62% decrease in phosphorus after the bioretention basin.

4.2.6 Alkalinity/pH

The average alkalinity values for the influent and effluent are 200 mg/L and 250 mg/L, respectively for filter strip 1. Although the alkalinity for filter strip 1 increases (as shown by the negative removal), the standard deviation and average values indicate that there was only a slight increase in alkalinity, Table 31. The second filter strip at the MSU site has larger influent concentrations at 225 mg/L and a more significant increase indicated by the negative removal.

Table 31: Alkalinity Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-11%	23%	14%	-46%	-136%	196%	-1%	-513%
1 ft	-32%	52%	8%	-152%	-277%	403%	0%	-962%

A decrease in alkalinity for the small MI dairy site, Table 32, is indicative of the increased nitrification and denitrification processes. Nitrification decreases alkalinity while denitrification increases alkalinity by half of the nitrification process resulting in a net decrease (Tchobanoglous et al. 2003).

Table 32: Alkalinity Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	30%	21%	51%	2%
1.5 ft	54%	23%	74%	11%
2.5 ft	60%	12%	71%	39%

Filter strip 1 at the MSU site had very little change in pH between sampling locations as both the average influent and effluent pH values were 6.7, Table 33. Filter strip 2 had a slight average increase from 5.5 to 6.0. However, the second filter strip commonly had acidic pH values between 4 and 5. These low pH values pose problems to biological treatment and caused burning of vegetation at the top of the filter strip. Limiting the dry weather leachate in the spring by effectively managing the upright silage filling processes can reduce the problem of low pH values.

Table 33: pH Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-2%	2%	0%	-5%	-6%	7%	2%	-20%
1 ft	1%	2%	5%	-3%	-9%	12%	0%	-37%

A slight decrease from an average pH of 8.1 to 7.3 occurred at the small MI dairy. These concentrations pose no issues for treatment practices and require no management or treatment operational changes.

Table 34: pH Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	9%	3%	15%	5%
1.5 ft	10%	5%	14%	4%
2.5 ft	10%	7%	22%	5%

4.2.7 Metals

Manganese concentrations from the influent concentrations are increased drastically in the subsurface effluent as the Mn within the soil profile leached into the subsurface samples, Table 35. The average influent and subsurface effluent values for filter strip 1 at the MSU site are 200 ug/L and 665 ug/L over a 3x increase, and 555 ug/L and 2550 ug/L for filter strip 2, nearly a 5x increase.

Table 35: Mn Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-118%	155%	78%	-372%	-158%	190%	56%	-417%
1 ft	-343%	814%	81%	-2311%	-375%	304%	-12%	-783%

The small MI dairy follows the same trend, Table 36, but to a lesser degree as sand soils have lower initial Mn concentrations and are theorized to have increased oxygen availability (due to decreased moisture holding capacity) as discussed previously. The average Mn influent concentration is 1370 ug/L and the effluent is 3650 ug/L, a 2.5x increase in Mn concentrations in the subsurface effluent. Greater average effluent concentrations are theorized due to the higher average influent of soluble Mn concentrations in comparison to the MSU Dairy.

Table 36: Mn Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	13%	33%	50%	-35%
1.5 ft	-27%	67%	42%	-147%
2.5 ft	-8%	76%	81%	-104%

Average Fe concentrations did not increase in the subsurface samples, Table 37. Fe influent concentrations at the MSU dairy are 1670 ug/L and 6720 ug/L for filter strip 1 and 2 respectively. These averages decrease to 1430 ug/L and 4830 ug/L for subsurface samples. This is in accordance with the electron tower as Mn is continuing to serve as the electron donor and Fe will not leach within the soil profile until Mn is exhausted.

Table 37: Fe Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-347%	594%	54%	-1534%	-114%	230%	49%	-510%
1 ft	-2%	68%	65%	-139%	27%	22%	64%	2%

The general decrease in Fe at the small MI dairy follows the same trend as the MSU site, Table 38.

Table 38: Fe Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	30%	53%	72%	-62%
1.5 ft	43%	27%	81%	22%
2.5 ft	58%	30%	86%	10%

Arsenic concentrations for filter strip 1 and 2 have influent values 1.5 ug/L and 3.7 ug/L and subsurface effluent values of 2.8 ug/L and 8.4 ug/L. Although there are slight increases, the averages are still below the US EPA drinking water standards of 10 ug/L (US EPA 2009b). The increase in the subsurface samples is due to the increase in the surface samples prior to infiltrating the soil profile, average As concentrations at the surface are 6.2 ug/L and 10.2 ug/L, which indicates that the soil profile reduces the As concentrations within the soil profile.

Table 39: As Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-297%	298%	-11%	-737%	-153%	83%	-57%	-253%
1 ft	-76%	80%	-11%	-225%	-119%	53%	-19%	-185%

Influent, surface and subsurface concentrations for the small MI dairy are 26.7 ug/L, 35.3 ug/L, and 16.3 ug/L, respectively. The increase in As concentrations at the surface level and the decrease in the subsurface is in agreement with the MSU site, as the soil profile is assimilating a portion of the As. However, unlike the MSU site the final concentration of As is of concern as it is above the EPA drinking water standards. The increase is thought to be due to the addition of excess plate cooler water to the settling basin as high concentrations in groundwater have been reported in Michigan and in this area (MDEQ 2006; Myoung-Jin 2002).

Table 40: As Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	15%	23%	48%	-13%
1.5 ft	52%	8%	62%	41%
2.5 ft	59%	16%	88%	45%

4.2.8 TOC

Decreases in organic carbon occurred for all filter strip subsurface samples, Table 41 and 42.

Table 41: TOC Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	7%	24%	39%	-27%	-21%	24%	15%	-42%
1 ft	16%	20%	41%	-21%	8%	30%	60%	-21%

Total removal is greater within the small MI dairy filter strip, again attributed mainly to the soil type and increased oxygen concentrations.

Table 42: TOC Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	58%	13%	71%	35%
1.5 ft	84%	4%	87%	76%
2.5 ft	82%	11%	94%	62%

4.2.9 Chloride

Removal rates for chloride were negative, resulting in a net increase in average concentrations.

Table 43: Chloride Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-35%	71%	16%	-176%	-28%	37%	16%	-71%
1 ft	-135%	300%	20%	-856%	-11%	28%	26%	-41%

At the small MI dairy the final effluent concentrations for Cl^- decrease as the effluent moves through the soil. Removal is achieved mostly within the bioretention basin. Chloride concentrations remained below the US EPA secondary maximum contaminant limit of 250 mg/L for all subsurface filter strip samples.

Table 44: Chloride Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	47%	18%	64%	20%
1.5 ft	57%	17%	73%	28%
2.5 ft	63%	9%	70%	49%

4.2.10 Conductivity

Conductivity can be representative of salts and soluble nutrients. This follows the same trend for chloride indicating an increase in salts within the soil profile and an increase in soluble nutrients as wastewater enters the soil. For filter strip 2 there is a general increase in conductivity over time even with generally consistent influent concentrations, indicating a build-up of salts and soluble nutrients within the soil treatment system. However, data from filter strip 1 and the small MI dairy do not follow this same trend.

Table 45: Conductivity Percent Removal - MSU Dairy Filter Strip

	Filter Strip 1				Filter Strip 2			
	Percent Removal				Percent Removal			
	Average	Std Dev	Max	Min	Average	Std Dev	Max	Min
Surface	-17%	39%	9%	-97%	-26%	26%	3%	-53%
1 ft	-28%	43%	10%	-112%	-15%	33%	23%	-61%

The small MI dairy showed a distinct reduction in conductivity indicating a decrease in salts as is supported by chloride removal.

Table 46: Conductivity Percent Removal – Small MI Dairy Filter Strip

	Percent Removal			
	Average	Std Dev	Max	Min
Surface	41%	14%	51%	17%
1.5 ft	59%	15%	69%	31%
2.5 ft	61%	7%	67%	49%

4.2.11 Cold Weather Performance

Cold weather performance was evaluated at the MSU Dairy site only, and was found to have no effect on performance. However, performance at the MSU site was poor in warm weather, so temperature may have a greater effect on a more efficient site. The main issues associated with cold weather performance are due to frozen ground leading to surface runoff commonly associated with land application of waste in the winter. However, this application has a number of differences that would limit the issues commonly faced in land application. The settling basin within these systems can collect the runoff as it melts (as temperatures rise as compared with land application during cold winter months), and application will not occur until the gravity or pump system has thawed, requiring warmer temperatures that will inevitably increase the thaw in the filter strip subsurface. The gravity fed system has a pipe feed above the freeze point within the soil subsurface, requiring the ground to thaw to that depth prior to any runoff application in addition to the large bioretention subsurface that the water must flow through prior to reaching the treatment strip which would also require the subsurface soil to thaw. The MSU pump based system has a much larger capacity and the ability to manually operate pumps to determine the optimal time for application. Although there is a more limited concern for cold weather surface discharge, investigation at a more efficient site will be required to assess the microbial performance at reduced temperatures as their activity is known to decrease with decreasing temperatures.

4.3 Laboratory Evaluation of Treatment System Design Components

Soil column effluent was assessed to determine the pollutant removal capabilities of the column experimental treatments. It should be noted that soil columns were investigated for 7 months, but did not reach a steady-state or equilibrium. This is important as removal and processes involved in removal are dynamic, and changes in one water quality parameter concentration may invoke changes in other parameters or mechanisms associated with removal within the soil column.

A number of soil and soil column characteristics were measured to evaluate the processes associated with removal of the water quality parameters discussed within this section. Physical characteristics include soil mechanical properties, bulk density and porosity, flow rates, and soil constituent concentrations. Flow rates for the 12 inch columns were initially more than 3 times the 30 inch columns and more than 10 times the 48 inch columns, Figure 16 and 17, not the differences in scale on the y-axis.

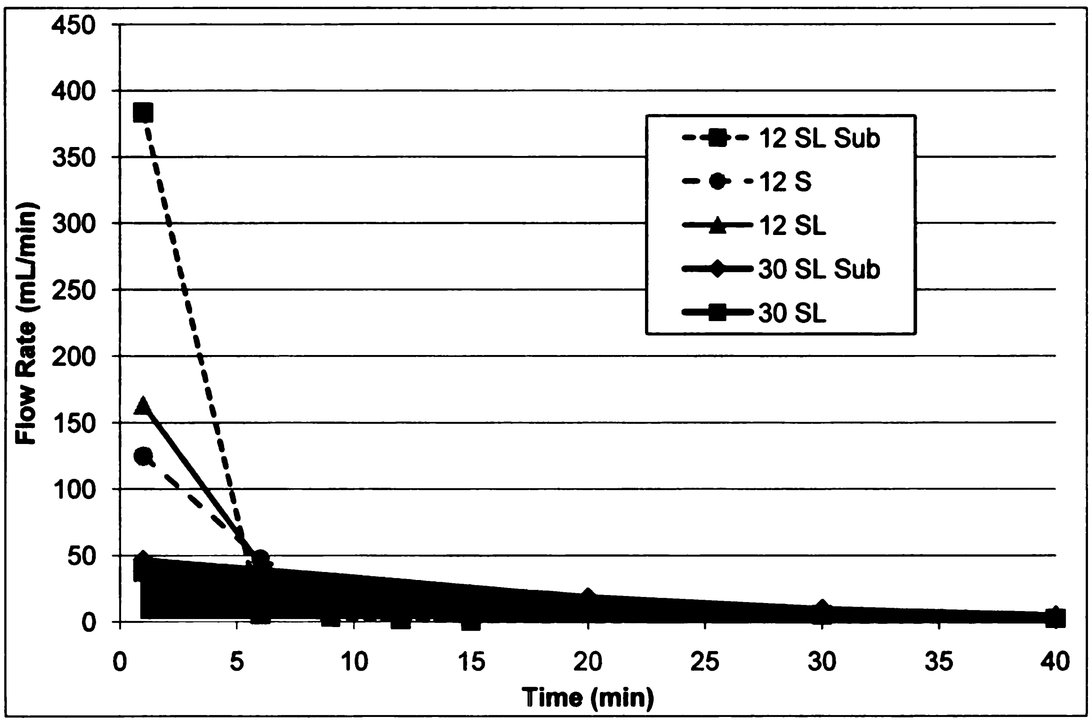

Figure 16: Soil Column Flow Rates – all 12 inch and 30 inch sandy loam columns.

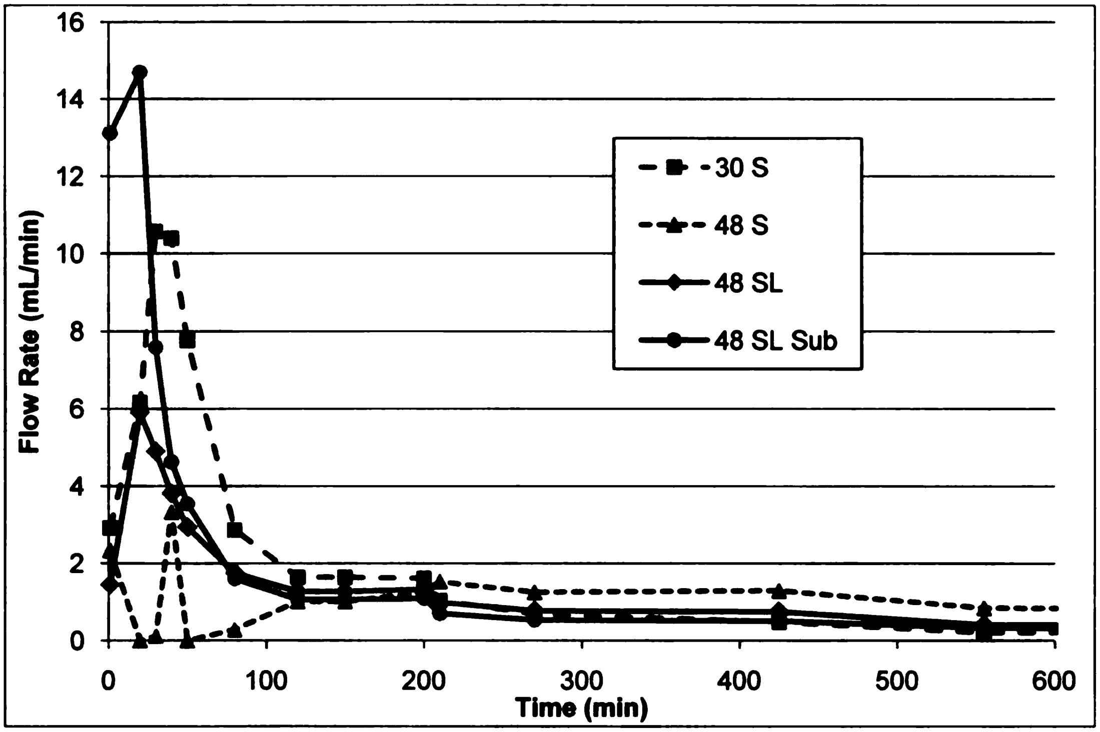

Figure 17: Soil Columns Flow Rates – 30 inch sand and all 48 inch columns

The 12 inch soil columns leached the entire wastewater volume within 20 minutes of application. Greater depths show a lag time in the increase in flow rates, and take over 10 hours to leach the entire volume of wastewater. An increase in depth decreases the flow rate from the column, increasing residence time resulting in greater contact time with the soil surface for increased adsorption and increased time for microbial metabolization. This increase is not linear with an increase in depth, indicating that the water not only has to travel through a greater depth of soil, but it also travels as a reduced rate of flow through the column. Soil characteristics dominate not only the hydraulic conductivities and flow rates, but also control the oxygen diffusion rates. Soil characteristics for the sand and sandy loam are in Table 47 below.

Table 47: Sand and Sandy Loam Soil Column Characteristics

Parameter	Sand	Sandy Loam
pH	8.8	6.9
P (ppm)	3	98
K (ppm)	8	133
Ca (ppm)	632	966
Mg (ppm)	198	198
Zn (ppm)	3.1	4.9
Mn (ppm)	4.4	13.9
Cu (ppm)	0.8	13.9
Fe (ppm)	8.1	44.7
Organic Matter (%)	0.3	2.3
Chloride (ppm)	61	59
Total N (ppm)	n.d.	0.10
Nitrate-N (ppm)	0.6	11.0
Ammonium-N (ppm)	0.5	1.4
Sand (%)	93.5	69.8
Silt (%)	2.8	25.9
Clay (%)	3.7	4.3

Bulk densities for the sand and sandy loam soils were determined experimentally to be 1.55 and 1.65, respectively. Bulk density can then be used to calculate soil porosities, which were 42% and 38%. The decrease in soil porosity in the sandy loam is due to the reduced size of the clay and silt particles filling the void space of the sand particles. Sandy loam soils typically have a greater porosity than sand soils, but in this case the compaction of the sandy loam clays has a significant effect on the bulk density reducing overall porosity and pore size. It has been shown that oxygen diffusion rates are increased due to increases in air-filled porosity and decreases in soil water content and bulk density (Feng et al. 2002). This relationship indicates that the oxygen diffusion rates are less in the sandy loam soils compared to that of the sand soils as they have greater bulk densities. In addition, as water fills the air voids within the soil, this also reduces the oxygen diffusion rates. This has implications for the columns of lesser depths as the ratio of wastewater volume to porosity volume is increased, Table 48, therefore increasing the overall soil pore column water content and reducing oxygen diffusion rates. Soils with decreased porosity require greater suction to remove soil pore water (Rose 2004), therefore sandy loam soils will retain more water when exposed to the same conditions as sand soils, a main factor in determining soil oxygen concentration. Soil moisture was measured every six inches in depth prior to deconstruction. Although the data was not precise enough to show the progression of the wetting front, it did show that sand soils had significantly decreased soil moisture after only a short time period as compared to the sandy loam columns. This again would have a positive effect on oxygen diffusion for sand columns. Average effluent volumes from the columns were also measured after each wastewater application, for columns of the same depth the sand soils had 15-25% more final effluent volume than the equivalent sandy loam soil columns. This is further evidence that the sandy loam soils retain more moisture than their sand counterparts.

Table 48: Soil Column Volume and Porosity

Depth (in)	Soil Volume (in^3)	Porosity Volume (mL)		Ratio of Wastewater Volume to Pore Space Volume	
		Sand	Sandy Loam	Sand	Sandy Loam
12	339	2335	2113	0.60	0.66
30	848	5838	5282	0.24	0.27
48	1357	9341	8451	0.15	0.17

Submerged columns also showed an increase in flow rates as compared to their non-submerged counterparts. When the columns are submerged, soil water rises within the capillary fringe due to pressure differences (Rose 2004), again increasing the soil water content and decreasing the soil oxygen content. The physical characteristics indicate that the columns of greater depth, those with sand soils, and those that are not submerged will have increased oxygen diffusion rates. In addition, columns of shorter depths have increased flow rates decreasing residence time and the associated soil mechanisms dependent upon contact time.

4.3.1 BOD

The synthetic wastewater was designed to have a BOD concentration of 225 mg/L. Actual BOD_5 influent averaged 172 mg/L with a standard deviation of 19 mg/L, but it should be noted that during analysis the results were commonly over the test range and therefore could not be included in the average resulting in a lower average than was actually realized. In addition, the design of the experiment used ultimate BOD concentrations, and 5-day BOD concentrations are always only a portion of the ultimate BOD. Column leachate was analyzed for BOD_5, and as mentioned prior, multiple set-ups are required to cover a large range for each sample. The low end of the test detection limit was designed to be 6 mg/L BOD_5, of which many of the samples were commonly below. Column replicates were used for analysis of statistical design for all parameters, but graphs were made based on the averages of the three replicate columns for each treatment combination. A graph of BOD_5 concentrations can be found in Figure 18, all columns that did not produce any readings over 6 mg/L BOD were not included.

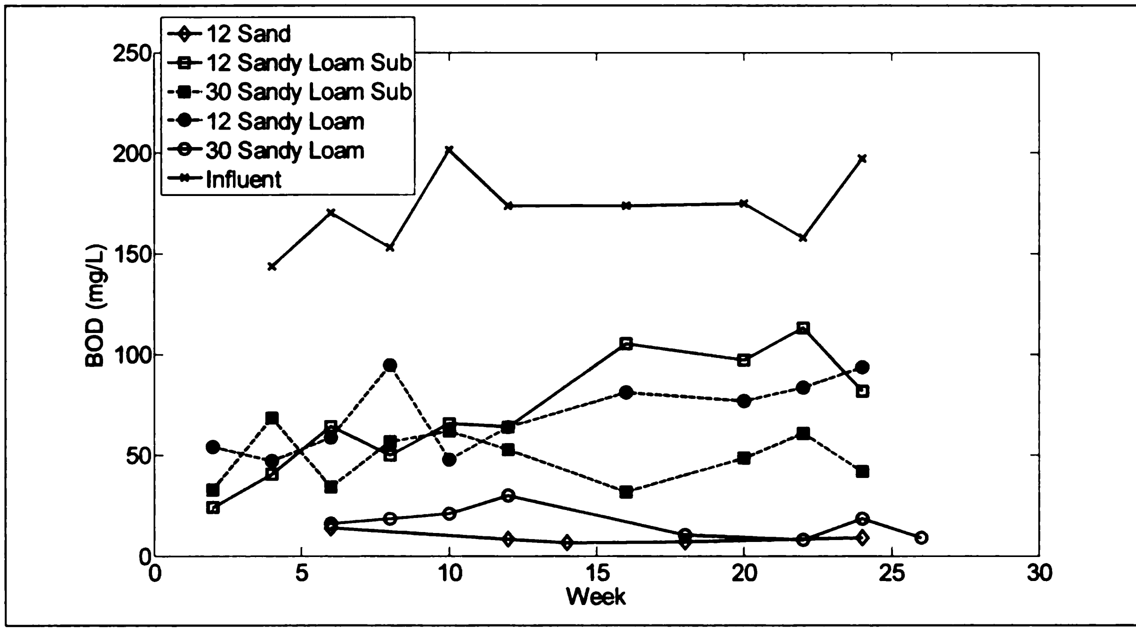

Figure 18: Soil Column BOD_5 Concentrations.

Columns 48 inches long, regardless of soil type or submergence criteria did not produce effluent BOD_5 concentrations over the detection limit of 6 mg/L. The 30 inch depth columns with sandy soil and the control columns of both soil types also did not produce effluent concentrations over 6 mg/L. Removal percentages for the 12 inch sandy loam columns, regardless of submergence, are in the mid-50% range. This increases for the 30 inch sandy loam submerged columns to an average of 70% removal. The remaining columns average removal percentages are 90-99%. Consequently, the statistical model had significance for depth and soil but not for submergence because there was not a significant difference in

BOD_5 concentrations for those columns that were and were not submerged, Table 49.

Table 49: Soil Column BOD Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	13.4	11.79	0.0043
Depth	2	12.8	6.71	0.0101
Soil*Depth	2	12.8	6.31	0.0124

A significant difference in effluent concentrations resulted for the 12 inch columns and 48 inch columns, Table 50, confirming the greatly reduced performance for the 12 inch columns. The shorter columns had decreased oxygen availability during the 20 min the wastewater traveled through the columns due to the greater ratio of wastewater volume to porosity.

Statistical analysis also identified a significant difference between BOD_5 effluent concentrations in columns with sand and sandy loam. Those with sand had greater removal than the equivalent column with a sandy loam soil for all columns, Table 50. This is most likely due to the greater porosity and decreased soil moisture within the sand which increases oxygen diffusion rates contributing to the removal of oxygen demand in these columns.

The soil*depth interaction effects revealed that columns with sandy loam soils had significant differences at all depths for BOD_5 concentrations. The 12 inch sandy loams columns (submerged and non-submerged) performed the poorest with an average removal percentage of 56%, increasing to 70% for the 30 inch sandy loam columns, and finally 94% for the 48 inch sandy loam columns. Again, the greater depths have a more favorable wastewater to porosity ration, increasing the available oxygen within the greater depths.

Table 50: Soil Column BOD Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Depth (in)	Soil	Depth (in)	Estimate	Standard Error	DF	t Value	Pr > |t|
Depth		12		48	35.1619	9.7862	8.25	3.59	0.0067
Soil	Sand		Sandy Loam		-28.2909	8.2397	13.4	-3.43	0.0043
Soil*Depth	Sand	12	Sandy Loam	12	-64.376	17.9825	7.78	-3.58	0.0075
Soil*Depth	Sand	30	Sandy Loam	12	-66.6267	16.0061	20.6	-4.16	0.0005
Soil*Depth	Sand	48	Sandy Loam	12	-65.8154	10.0546	8.8	-6.55	0.0001
Soil*Depth	Sand	48	Sandy Loam	30	-22.7544	8.5357	14.7	-2.67	0.0178
Soil*Depth	Sandy Loam	12	Sandy Loam	30	43.0611	10.3102	11.1	4.18	0.0015
Soil*Depth	Sandy Loam	12	Sandy Loam	48	68.8844	9.6512	7.91	7.14	0.0001
Soil*Depth	Sandy Loam	30	Sandy Loam	48	25.8233	8.0567	14.5	3.21	0.0061

Other statistical comparisons of significance in the sand columns and the sandy loam columns can be found above, but the remaining effects all follow the same general trend, sand soils have greater reduction in BOD_5 than sandy loam soils. Impact on treatment system design indicates that a depth of 48 inches, regardless of soil type, is sufficient for treatment of BOD_5 at these loadings or below. Sites with sandy soils require only 30 inches of depth for treatment to concentrations below 6 mg/L. Both of these conditions results in BOD_5 reading of 6 mg/L or less and do not pose a danger to groundwater resources. Surface discharge concentrations as determined by the Clean Water Act are 30 mg/L

BOD (US EPA 2007), which were met by all columns greater than 12 inches in depth.

4.3.2 COD

The synthetic wastewater applied to the soil columns had an average influent COD concentration of 325 mg/L and the water applied to the control columns had an influent concentration of COD 30 mg/L. Treatment performance for COD for sand and sandy loam is presented in Figure 19 & 20, respectively. The sand control columns performed similarly to the 30 and 48 inch sand columns, indicating that these sand columns did not produce a greater COD effluent with wastewater application than with the application of water. The 12 inch sand column, although follows the same trends as the other sand columns, was determined to be statistically significantly different from the 48 inch column only.

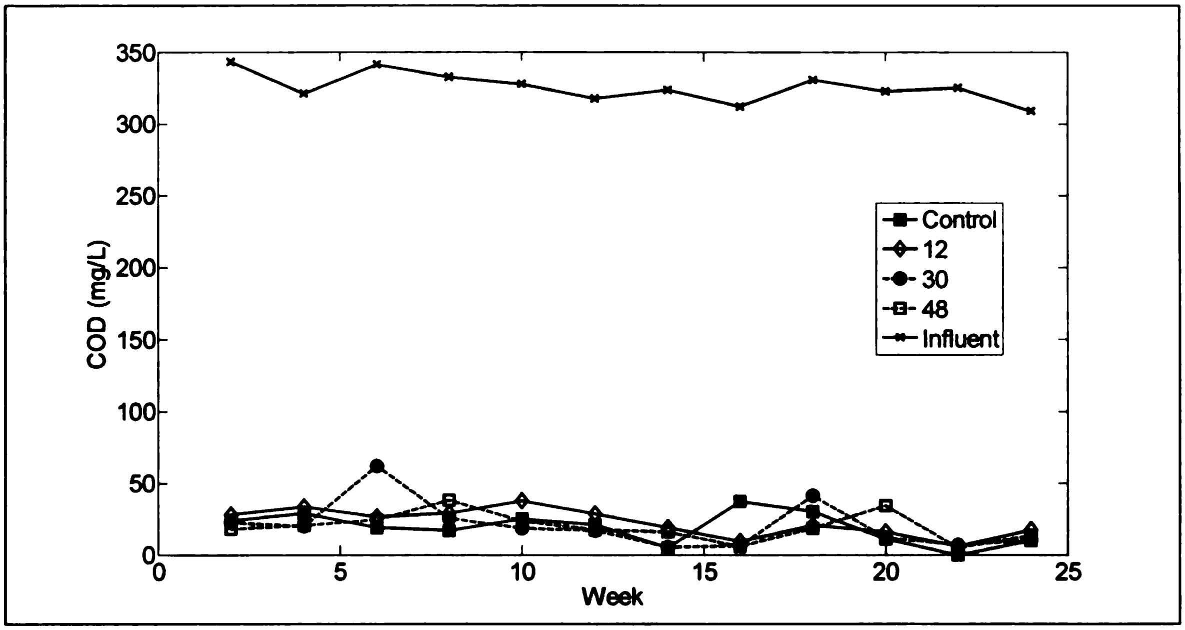

Figure 19: Sand Soil Column COD Concentrations.

The spike in the 30 inch sand column at week 8 was the result of one of the column readings used to calculate the average being significantly greater, 176 mg/L, than all of the other readings. This is most likely due to an error in laboratory analysis as at no other time did the effluent for this column exceed 28 mg/L. Although here readings became slightly more erratic as experimentation continued, effluent concentration remained within the same general range.

Examining the graph for sandy loam columns (Figure 16), the 12 inch columns, regardless of submergence, performed more poorly than all other columns. The control column responded similarly to the 30 and 48 inch sandy loam columns that were not submerged and the 48 inch column that was submerged.

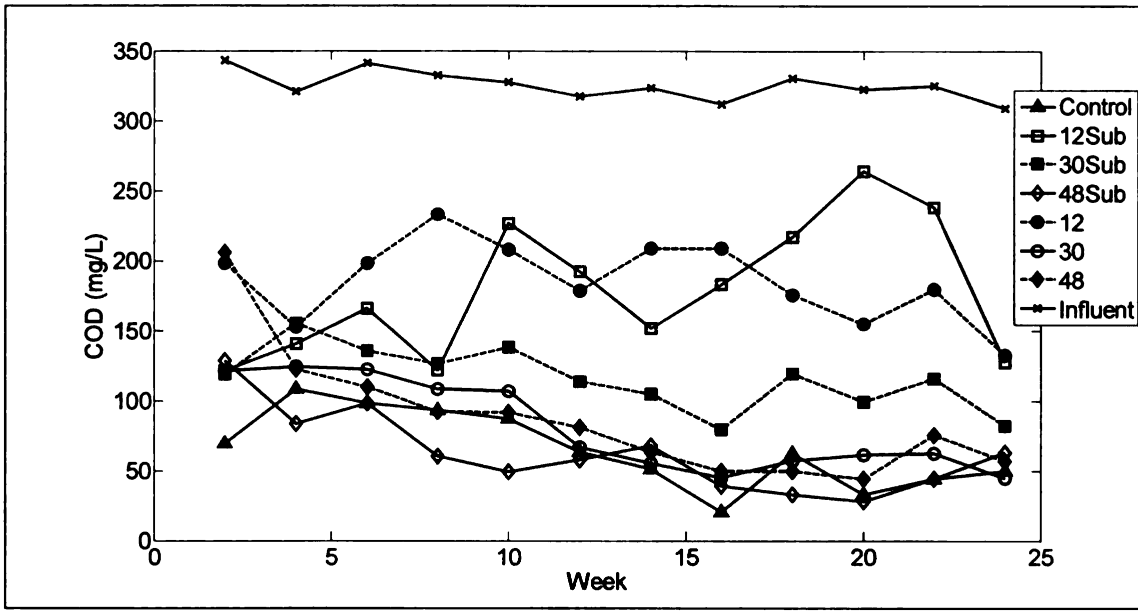

Figure 20: Sandy Loam Soil Column COD Concentrations.

The statistical model for COD was significant for depth, soil, submergence, time (as a repeated measure) and the various interactions of these main effects, Table 51.

Table 51: Soil Column COD Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	21.8	102.11	<.0001
Depth	2	24.2	34.64	<.0001
Soil*Depth	2	15.8	18.08	<.0001
Sub	1	26.5	18.39	0.0002
Depth*Sub	2	17.3	19.29	<.0001
Time	11	30.3	9.88	<.0001
Sub*Time	11	27.9	3.88	0.0018
Soil*Time	11	29.2	7.17	<.0001

As with BOD_5, there is a significant difference in the soil columns with a sand soil to those with sandy loam, Table 52. The sandy loam columns did not perform as well as the sand soils concerning COD removal, once again due to the available oxygen, see discussion in section 4.3. Columns at a depth of 48 inches had significantly lower effluent COD concentrations than 12 inch and 30 inch soil columns, thought to be due to the ratio of wastewater volumes to porosity volumes and decreased flow rates both of which result in the 48 inch columns never becoming completely saturated leaving oxygen in the remaining pore space (verified by soil moisture measurements). Overall the submerged columns performed significantly differently than the non-submerged columns in terms of COD effluent concentrations.

Further statistical analysis involves interaction of the main effects. There is a significant difference in the all depths of the sand soils with all other depths in the sandy loam soils, meaning that even at a depth of 48 inches there was a significant difference between the sand and the sandy loam columns in terms of COD effluent. This is again due to the soil differences mentioned above that hold true for all depths. Average removal percentages for the sand columns outperform all sandy loam columns. Even the12 inch sand column had a significantly lower effluent concentration than the 48 inch sandy loam column, once again indicating the increased oxygen diffusion to sand soils and a reduction in pore-water. Pollutant removal within the columns that were submerged increased with depth; the 12 inch columns had and average COD removal of 45%, 30 inch columns 65% and 48 inch columns 81%, again due to pore-water ratios and available oxygen.

Table 52: Soil Column COD Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Sub	Depth (in)	Soil	Sub	Depth (in)	Estimate	Std Error	DF	t Value	Pr > |t|
Soil	Sand			Sandy Loam			-144.01	14.251	21.8	-10.11	<.0001
Depth			12			48	67.0388	13.662	11.6	4.91	0.0004
Depth			30			48	47.9626	6.6127	52.8	7.25	<.0001
Soil* Depth	Sand		12	Sand		30	-93.9179	29.501	15.9	-3.18	0.0058
Soil* Depth	Sand		12	Sandy Loam		12	-288.38	33.211	10.3	-8.68	<.0001
Soil* Depth	Sand		12	Sandy Loam		30	-156.31	28.574	13.7	-5.47	<.0001
Soil* Depth	Sand		12	Sandy Loam		48	-117.78	29.118	12.6	-4.04	0.0015
Soil* Depth	Sand		30	Sand		48	57.4018	16.309	23.5	3.52	0.0018
Soil* Depth	Sand		30	Sandy Loam		12	-194.46	17.475	33.5	-11.13	<.0001
Soil* Depth	Sand		30	Sandy Loam		30	-62.3872	16.972	14.2	-3.68	0.0024
Soil* Depth	Sand		48	Sandy Loam		12	-251.86	17.459	16.5	-14.43	<.0001
Soil* Depth	Sand		48	Sandy Loam		30	-119.79	14.496	15	-8.26	<.0001
Soil* Depth	Sand		48	Sandy Loam		48	-81.2656	21.276	4.66	-3.82	0.0141
Soil* Depth	Sandy Loam		12	Sandy Loam		30	132.07	16.108	26	8.2	<.0001
Soil* Depth	Sandy Loam		12	Sandy Loam		48	170.59	16.964	15.8	10.06	<.0001
Soil* Depth	Sandy Loam		30	Sandy Loam		48	38.5234	14.049	15.2	2.74	0.015
Sub		No			Submerged		54.6667	12.749	26.5	4.29	0.0002
Depth* Sub		No	12		Submerged	12	150.91	26.513	11.2	5.69	0.0001
Depth* Sub		No	12		No	30	117.79	18.701	18.7	6.3	<.0001
Depth* Sub		No	12		Submerged	30	71.2734	20.818	27.1	3.42	0.002
Depth* Sub		No	12		No	48	112.69	19.701	14.6	5.72	<.0001
Depth* Sub		No	12		Submerged	48	172.3	20.105	15.5	8.57	<.0001
Depth* Sub		Submerged	12		Submerged	30	-79.6395	24.484	24.1	-3.25	0.0034
Depth* Sub		No	30		Submerged	30	-46.5183	18.050	16.8	-2.58	0.0197
Depth* Sub		No	30		Submerged	48	54.5063	14.134	13.8	3.86	0.0018
Depth* Sub		Submerged	30		No	48	41.419	16.445	26.4	2.52	0.0182
Depth* Sub		Submerged	30		Submerged	48	101.02	17.154	27.9	5.89	<.0001
Depth* Sub		No	48		Submerged	48	59.6055	21.254	4.69	2.8	0.0406

Average final COD concentrations for all sand columns were below 25 mg/L. Sandy loam columns at a depth of 48 inches had an average effluent value of 63 mg/L, 30 inch non-submerged and 48 inch submerged sandy loam columns had concentrations of 81 and 87 mg/L, 30 inch sandy loam submerged were 116 mg/L, and the remaining 12 inch sandy loam columns had concentrations over 180 mg/L. Examining the removal percentages and the final concentrations of the effluent, sandy soils can remove COD at these influent concentrations at a depth of only 12 inches. However, sandy loam soils have a reduction in their efficiency as they have decreased oxygen diffusion rates and an increase in saturation and length of saturation. These soils require a minimum depth of 30 inches, although a 48 inch depth is recommended if groundwater is a factor.

Comparing the BOD_5/COD ratios, average influent ratios were 0.53 and decreased for all columns after leaching, confirming degradation. However, 12 inch columns of all treatments showed relatively high final BOD_5/COD ratios (0.38-0.42) indicating incomplete removal of biodegradable material, particularly because columns of greater length achieved increased removal of the same influent wastewater make-up (BOD/COD effluent ratios from 0.1-0.2).

Soil COD concentrations indicate that the soil COD levels for the sandy loam soil remained relatively equal with depth up to 30 inches, although slightly higher than the control column. The 48 inch sandy loam soil which was submerged followed these trends, however the 48 inch non-submerged soil indicated a decrease in the soil COD at a depth greater than 30 inches, Figure 21. This is consistent with the statistical analysis for COD effluent which indicates a significant difference from 30 to 48 inches and between the 48 inch submerged and non-submerged columns.

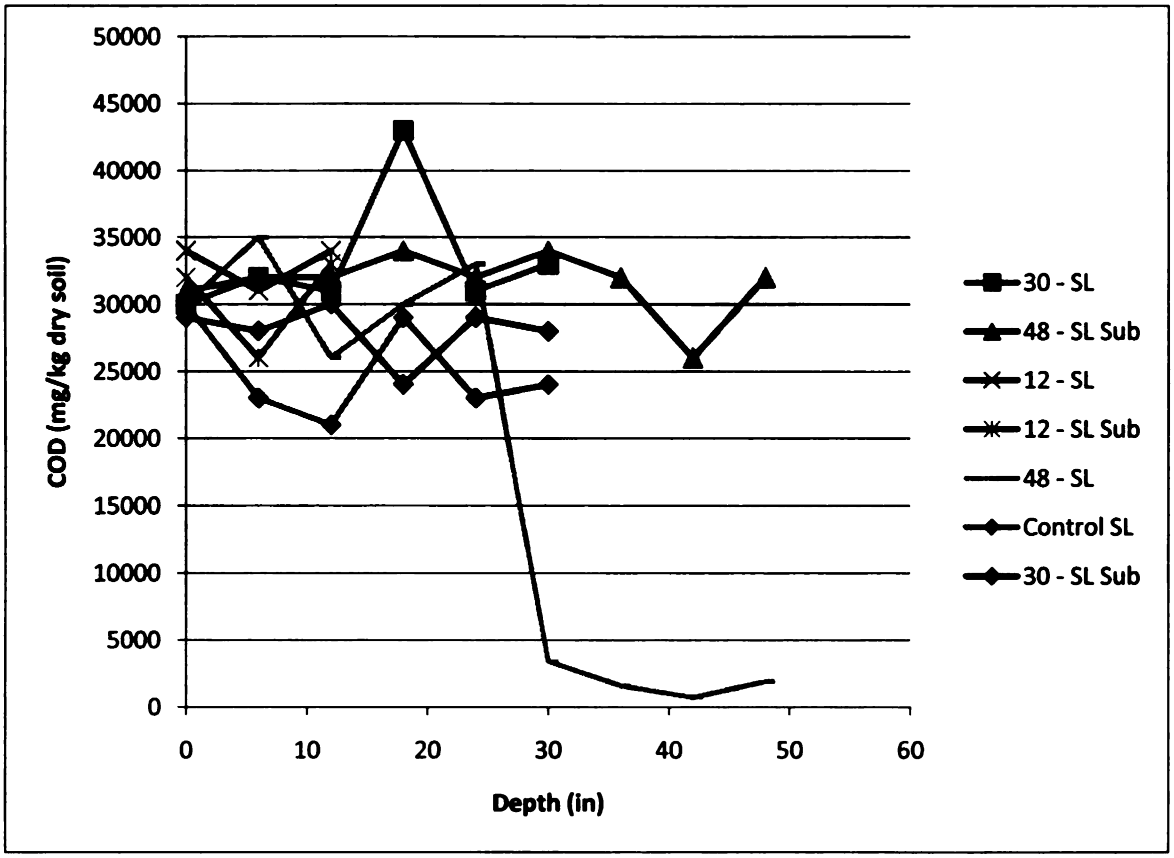

Figure 21: COD Effluent Concentrations as a Function of Soil Column Depth

Sand soil have a defined decrease in soil COD with an increase in depth, Figure 22. The soil COD concentrations decrease to zero after 20 inches, resulting in higher effluent COD concentrations at 12 inches than the greater depths.

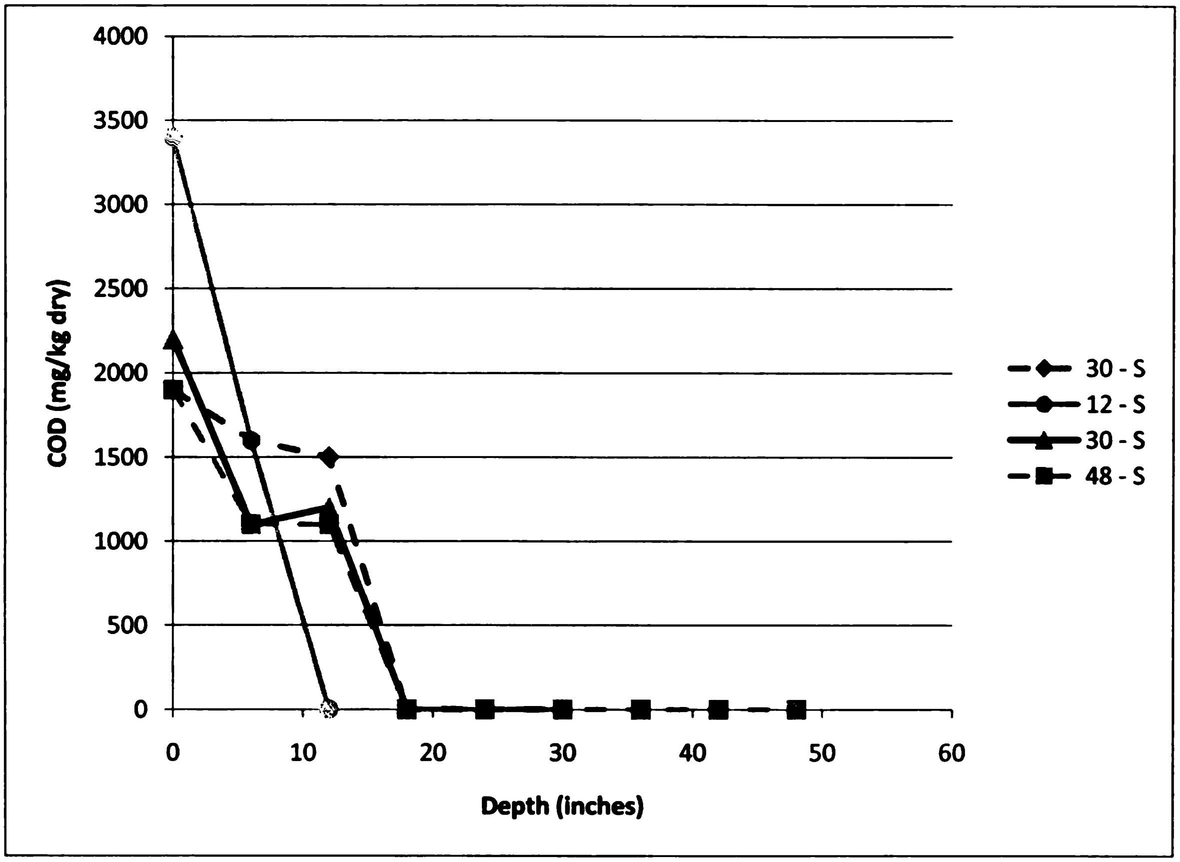

Figure 22: COD Effluent Concentrations as a Function of Soil Column Depth

4.3.3 Nitrogen

Nitrogen cycling is a dynamic process which required measurement of TKN, ammonia, nitrite, and nitrate to determine the underlying processes. Wastewater influent TKN concentrations averaged 29 mg/L-N of which an average of 26 mg/L-N, or 90%, was ammonia. Average TKN removal percentages for sand were 85%, 98%, and 97% for the 12, 30, and 48 inch depths, respectively. Final average effluent concentrations were 4mg/L-N for the 12 inch columns and under 1 mg/L-N for the longer columns, Figure 23. Removal of TKN relies on the process of ammonification by microorganisms, which was significant as the remaining concentrations of TKN are greatly reduced with an average of 90% being in the form of ammonia. The control column behaved very similarly to the 30 and 48 inch columns with an average effluent concentration below 1 mg/L.

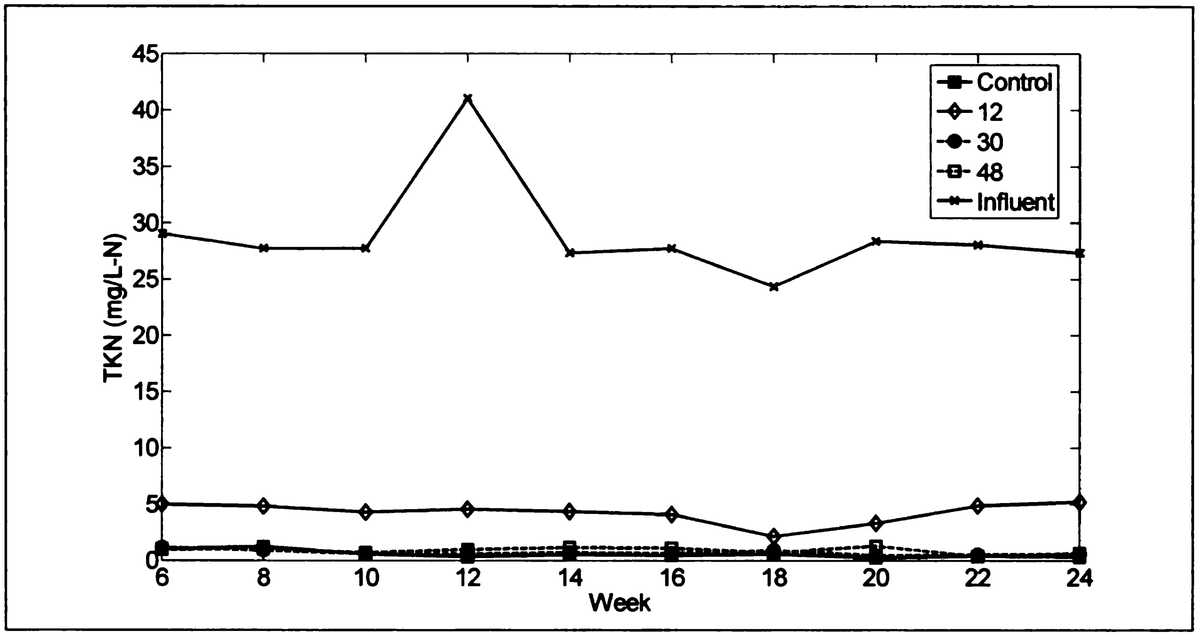

Figure 23: Sand Soil Column TKN Concentrations.

The sandy loam 12 inch columns had higher effluent values for TKN, compared to the columns of greater depth, Figure 24. Average removal was similar for the submerged and non-submerged columns as both of the 12 inch sandy loam (submerged and non-submerged) had an average TKN removal of 55%. This increased into the 80% range for the columns of greater depth. Final effluent average TKN concentrations were 12-13 mg/L-N for both 12 inch sandy loam columns, 6 mg/L-N for the 30 inch sandy loam submerged column, and 3-4 mg/L-N for the 48 inch submerged and the 30 and 48 inch sandy loam columns. The control columns produced similar results to the 48 inch submerged and the 30 and 48 inch non-submerged sandy loam columns with an average effluent of 3 mg/L-N.

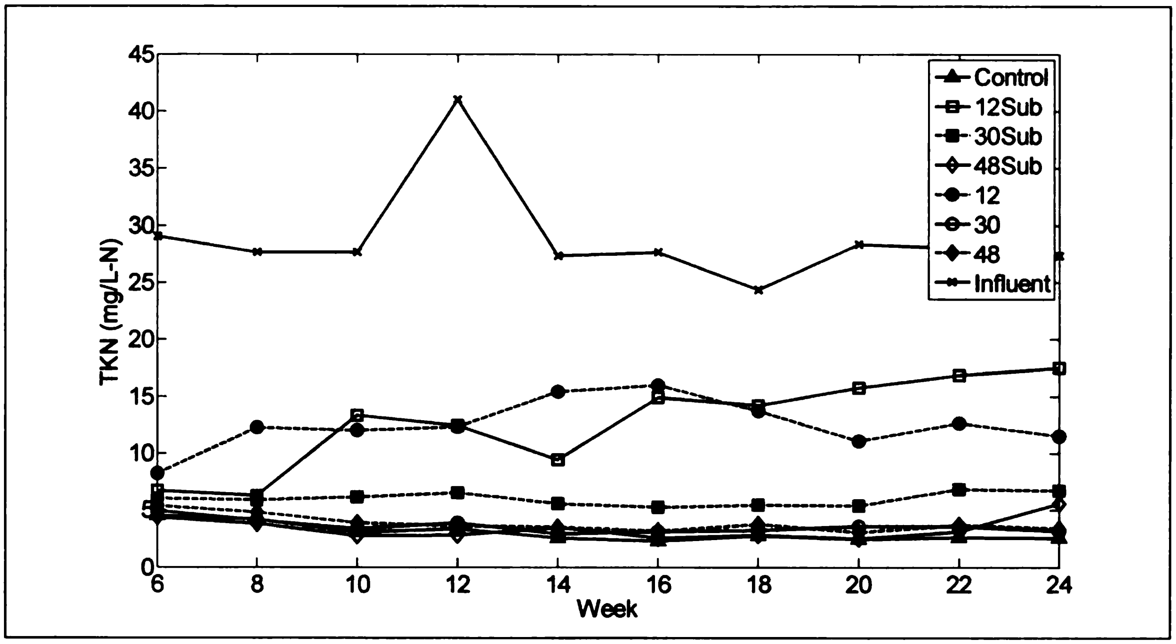

Figure 24: Sandy Loam Soil Column TKN Concentrations.

The statistical model for TKN included the main effects for soil, depth, and submergence as well as a number of their interactions, Table 53.

Table 53: Soil Column TKN Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	30.7	83.94	<.0001
Depth	2	37.5	55.69	<.0001
Soil*Depth	2	17.5	6.77	0.0067
Depth*Sub	2	19.4	13.98	0.0002
Time	9	73.9	3.95	0.0004
Soil*Time	9	74.1	4.4	0.0001
Sub*Time	9	71.6	3.53	0.0012

A statistically significant reduction in TKN concentrations exist for sand columns as compared to sandy loam columns. Depth was significant for TKN effluent concentrations as well, as each increasing depth has a greater statistically significant removal, Table 54. In comparing the interactions, the sandy loam soils had a significant difference between all depths for the submerged and non-submerged columns whereas there was no significant difference for sand soils at 30 and 48 inch depths. The increased contact time of the longer columns is thought to play a distinct part in the microbial removal of organically bound nitrogen. Mineralization rates, or ammonification rates, are the greatest when aerobic microorganisms are dominant (Vymazal 2007). Sand columns are also theorized to increase oxygen diffusion which would have a significant effect on increasing the microbial activity.

Table 54: Soil Column TKN Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Sub	Depth	Soil	Sub	Depth	Estimate	Std Error	DF	t Value	Pr > |t|
Soil	Sand			Sandy Loam			-4.8741	0.532	30.7	-9.16	<.0001
Depth			12			30	5.1498	0.7739	46.8	6.65	<.0001
Depth			12			48	6.8569	0.7272	50.1	9.43	<.0001
Depth			30			48	1.7071	0.3164	26.9	5.39	<.0001
Soil* Depth	Sand		12	Sand		48	4.1987	1.2895	35.3	3.26	0.0025
Soil* Depth	Sand		12	Sandy Loam		12	-8.5225	1.4478	36.2	-5.89	<.0001
Soil* Depth	Sand		30	Sand		48	1.8633	0.5379	20.2	3.46	0.0024
Soil* Depth	Sand		30	Sandy Loam		12	-10.858	0.8384	58.7	-12.95	<.0001
Soil* Depth	Sand		30	Sandy Loam		30	-2.8938	0.5132	34.2	-5.64	<.0001
Soil* Depth	Sand		30	Sandy Loam		48	-1.3428	0.5103	17.2	-2.63	0.0174
Soil* Depth	Sand		48	Sandy Loam		12	-12.7213	0.7524	50.5	-16.91	<.0001
Soil* Depth	Sand		48	Sandy Loam		30	-4.7571	0.4067	14.4	-11.7	<.0001
Soil* Depth	Sand		48	Sandy Loam		48	-3.2061	0.432	4.12	-7.42	0.0016
Soil* Depth	Sandy Loam		12	Sandy Loam		30	7.9642	0.7602	58.7	10.48	<.0001
Soil* Depth	Sandy Loam		12	Sandy Loam		48	9.5151	0.7322	47.6	13	<.0001
Soil* Depth	Sandy Loam		30	Sandy Loam		48	1.5509	0.368	10.5	4.21	0.0016
Depth* Sub		No	12		No	30	6.3796	0.768	37.5	8.31	<.0001
Depth* Sub		No	12		Subm erged	30	3.5603	0.8974	32.3	3.97	0.0004
Depth* Sub		No	12		No	48	6.127	0.7555	29.4	8.11	<.0001
Depth* Sub		No	12		Subm erged	48	7.2271	0.7758	31.3	9.32	<.0001
Depth* Sub		Subm erged	12		No	30	6.7394	1.2443	63	5.42	<.0001
Depth* Sub		Subm erged	12		Subm erged	30	3.9201	1.3281	58.6	2.95	0.0045
Depth* Sub		Subm erged	12		No	48	6.4867	1.2366	57.4	5.25	<.0001
Depth* Sub		Subm erged	12		Subm erged	48	7.5869	1.2486	58.6	6.08	<.0001
Depth* Sub		No	30		Subm erged	30	-2.8193	0.595	21.7	-4.74	0.0001
Depth* Sub		No	30		Subm erged	48	0.8475	0.3786	16.1	2.24	0.0397
Depth* Sub		Subm erged	30		No	48	2.5667	0.5725	14.3	4.48	0.0005
Depth* Sub		Subm erged	30		Subm erged	48	3.6668	0.5991	16.6	6.12	<.0001
Depth* Sub		No	48		Subm erged	48	1.1002	0.4334	4.17	2.54	0.0615

Ammonia concentrations followed similar trends to that of TKN, Figure 25. Influent concentrations for ammonia were 25.8 mg/L-N for the synthetic wastewater and 2.6 mg/L-N for the water applied to the control columns. Sandy soils produced removal percentages of 85% for the 12 inch columns and 97% for the 30 and 48 inch columns, with final average ammonia concentrations of 3.8 mg/L-N for the 12 inch columns and under 1 mg/L-N for the sand columns of greater depth. Control sand columns produced similar effluent concentrations to the 30 and 48 inch sand columns which received wastewater, below 1 mg/L-N.

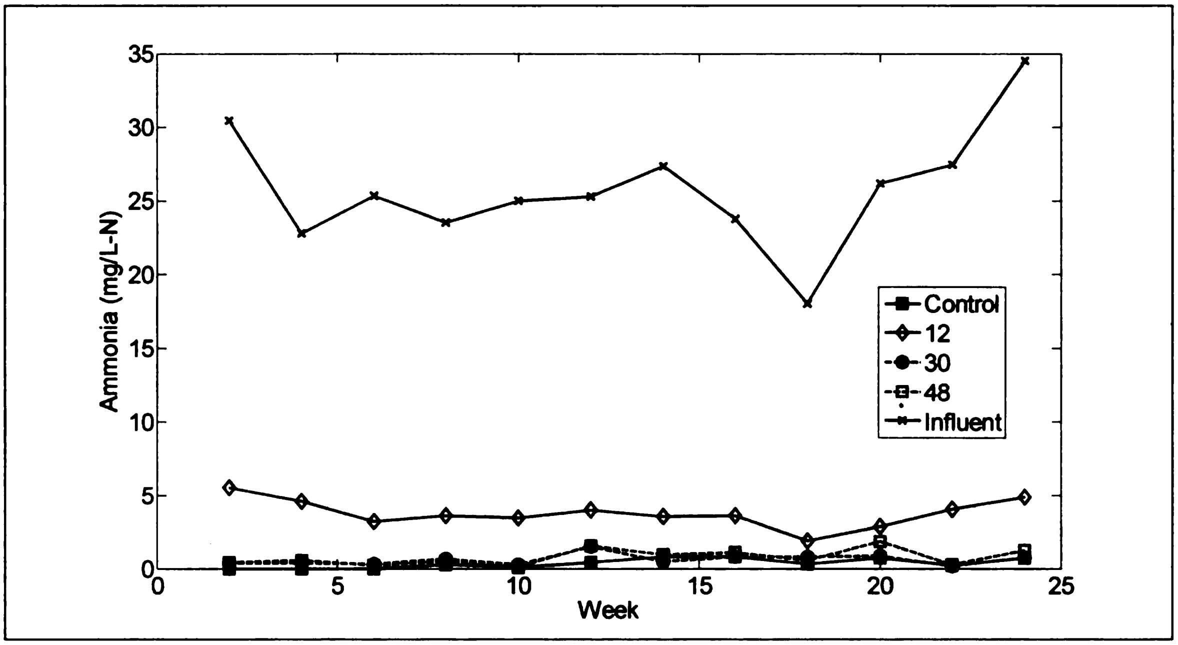

Figure 25: Sandy Soil Column Ammonia Concentrations.

The sand columns had lower concentrations than sandy loam columns of equal depths as removal of ammonia is an aerobic process and the sand columns are thought to have increased oxygen diffusion rates and remain saturated for a reduced amount of time increasing nitrification rates, Figure 26.

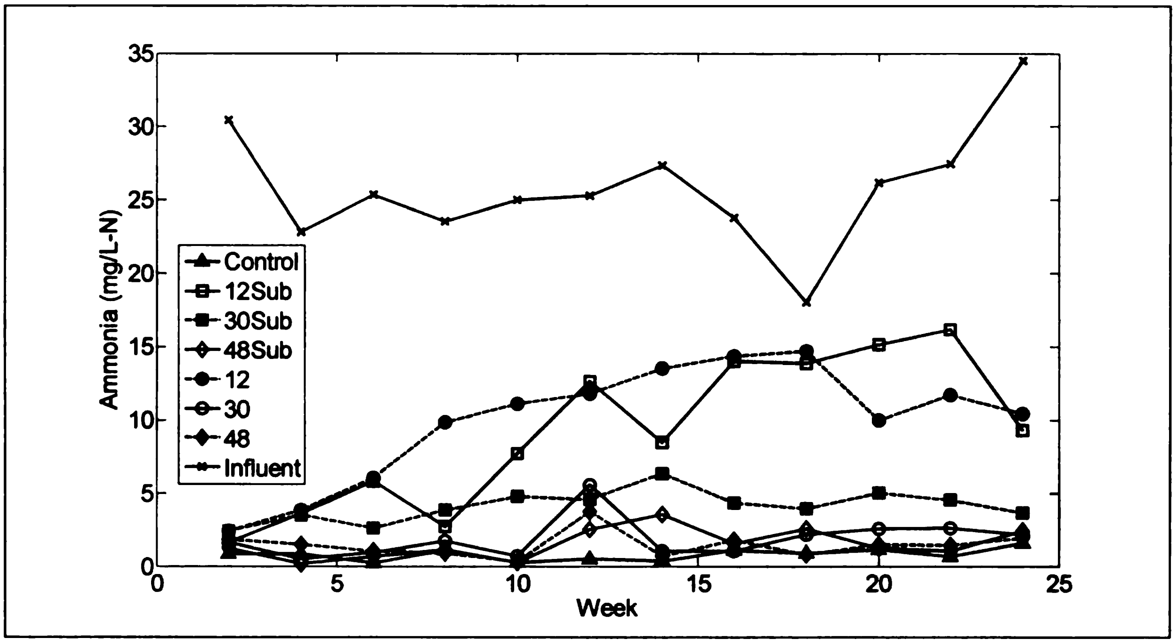

Figure 26: Sandy Loam Soil Column Ammonia Concentrations.

The increase in ammonia concentrations within the 12 inch sandy loam columns could be caused by the increase in TKN therefore increasing ammonia, a decrease in the conversion of ammonia to nitrite, a reduction in binding sites within the 12 inch sandy loam soil columns or a combination of these mechanisms. It is most likely a combination of these mechanisms as a decrease in the available oxygen (due to decreased oxygen diffusion and increased soil pore water) would retard ammonification and nitrification rates leading to an increase in TKN and ammonia, and a decrease in the conversion of ammonia to nitrite both resulting in ammonia build-up over time. In the case that sorption plays a role, it is reasonable to assume that the 12 inch columns were not sufficient in providing the sorption sites required to prevent ammonia within the effluent.

The sandy loam columns had average ammonia removal percentages of 60-63% for the 12 inch submerged and non-submerged columns, 84% for the submerged 30 inch columns, and 93-94% for the remaining columns. The final effluent concentrations were 9-10 mg/L for all sandy loam 12 inch columns, 4 mg/L for the 30 inch submerged and 1-2mg/L for the remaining columns. The sandy loam control column produced an effluent concentration below 1 mg/L. The larger effluent concentrations for the 12 inch sandy loam columns suggest a lack of oxygen during the time when water was leaching though the column producing the greater concentrations within the effluent. The low concentrations within the columns of greater length indicate there was not a lack of oxygen required for nitrification of ammonia. This is further supported by previous research which has shown soil nitrification rates are dependent upon temperature, pH, available carbon, and aeration which is effected by soil moisture and structure (Barker et al. 2000, Paul and Clark 1996, Yue-Mei et al. 2008; Sahrawat 2008). In this case, temperatures fall between 14.2°C – 24.2°C near the optimal nitrification temperature of 25°C and within the range of 5°C - 40°C outside of which nitrification if inhibited (Sahrawat 2008; Paul and Clark 1996; Yue-Mei et al. 2008). Microbial nitrification requires a carbon source, typically CO_2^- or carbonate (Paul and Clark 1996), which alkalinity concentrations indicate are in available in excess, section 4.3.5. The optimum pH range is 6.6 to 8.0 (Paul and Clark 1996; Tchobanoglous et al. 2003; Sahrawat 2008; Yue-Mei et al. 2008), of which all columns remain within the range for the duration of the experiment, section 4.3.5. This leaves aeration, which is affected by soil moisture, as the only remaining impact factor on nitrification rates, which further verifies the observations based on the experimental data. Nitrification rates are known to be inhibited at dissolved oxygen concentrations below 2 mg/L (Yue-Mei et al. 2008).

Statistical analysis support these observations with statistically significant differences for the main effects of soil and depth, with sand outperforming sandy loam in ammonia removal and increasing ammonia removal with increasing treatment depth. Interaction effects show that there is no difference between the effluent concentrations for the 12 inch sand columns and the 30 and 48 inch sandy loam columns. There is no difference in treatment for sand soils with increasing depth, but there is a significant difference at all depths for sandy loam soils indicating that aeration required for nitrification was adequate in all depths for sandy soil, but not for the 12 inch sandy loam columns. Other significant interaction effects can be found in Table 55.

Table 55: Soil Column Ammonia Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Sub	Depth (in)	Soil	Sub	Depth (in)	Est	Std Error	DF	t Value	Pr > |t|
Soil	Sand			Sandy Loam			-2.7811	0.6451	10.9	-4.31	0.0013
Depth			12			30	3.8341	0.8632	19	4.44	0.0003
Depth			12			48	5.7116	0.8442	18.9	6.77	<.0001
Depth			30			48	1.8775	0.4582	49.4	4.1	0.0002
Soil* Depth	Sand		12	Sandy Loam		12	-6.6274	1.6626	11.9	-3.99	0.0018
Soil* Depth	Sand		30	Sandy Loam		12	-7.757	0.9111	31.2	-8.51	<.0001
Soil* Depth	Sand		30	Sandy Loam		30	-1.2185	0.5739	74.1	-2.12	0.0371
Soil* Depth	Sand		48	Sandy Loam		12	-9.2739	0.9457	18	-9.81	<.0001
Soil* Depth	Sand		48	Sandy Loam		30	-2.7354	0.6866	15	-3.98	0.0012
Soil* Depth	Sandy Loam		12	Sandy Loam		30	6.5385	0.8259	25.8	7.92	<.0001
Soil* Depth	Sandy Loam		12	Sandy Loam		48	8.7766	0.845	14.6	10.39	<.0001
Soil* Depth	Sandy Loam		30	Sandy Loam		48	2.2381	0.5395	8.81	4.15	0.0026
Depth* Sub		No	12		No	30	5.8851	0.8794	13.6	6.69	<.0001
Depth* Sub		No	12		Submer ged	30	2.8741	1.0537	16.7	2.73	0.0145
Depth* Sub		No	12		No	48	6.0163	0.9241	9.66	6.51	<.0001
Depth* Sub		No	12		Submer ged	48	6.4979	1.0142	11.8	6.41	<.0001
Depth* Sub		Subm erged	12		No	30	4.7941	1.3207	25.7	3.63	0.0012
Depth* Sub		Subm erged	12		No	48	4.9252	1.3509	20.2	3.65	0.0016
Depth* Sub		Subm erged	12		Submer ged	48	5.4069	1.4141	21.8	3.82	0.0009
Depth* Sub		No	30		Submer ged	30	-3.0111	0.7199	40.8	-4.18	0.0001
Depth* Sub		Subm erged	30		No	48	3.1422	0.763	15.4	4.12	0.0009
Depth* Sub		Subm erged	30		Submer ged	48	3.6239	0.8699	19.8	4.17	0.0005

Typical ammonia surface discharge limits are 8 mg/L, which were achieved by all columns except the 12 in sandy loam submerged and non-submerged. Filter strip design requires a depth of 12 inches for sand soils and 30 inches for sandy loam soils for removal of ammonia below surface discharge levels.

Influent nitrite concentrations in the wastewater and water for the control column are negligible. Nitrite concentrations in the sand columns remain close to or below the detection limit in accordance with the control column for all columns except the 12 inch, Figure 27.

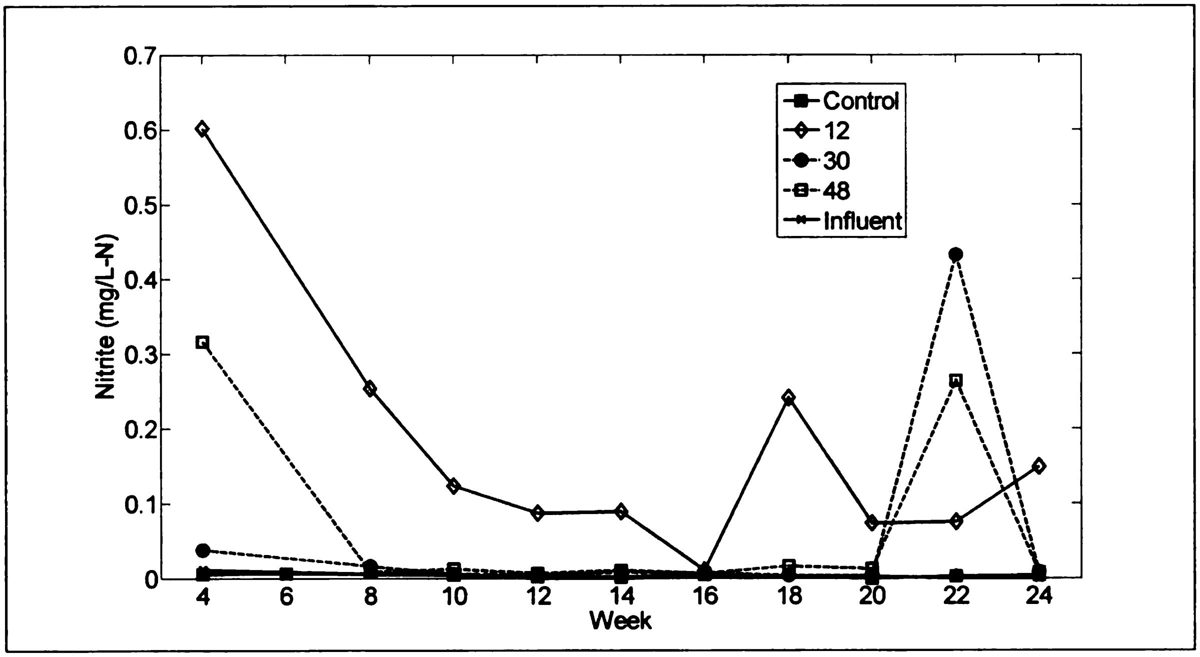

Figure 27: Sand Soil Column Nitrite Concentrations

The sandy loam columns follow a similar trend except the spikes at weeks 10 and 14 which are a magnification of the spikes in the TKN and ammonia, Figure 28. In this case however, the effluent concentrations are increased by one order of magnitude in comparison to the sand columns for the 12 inch columns.

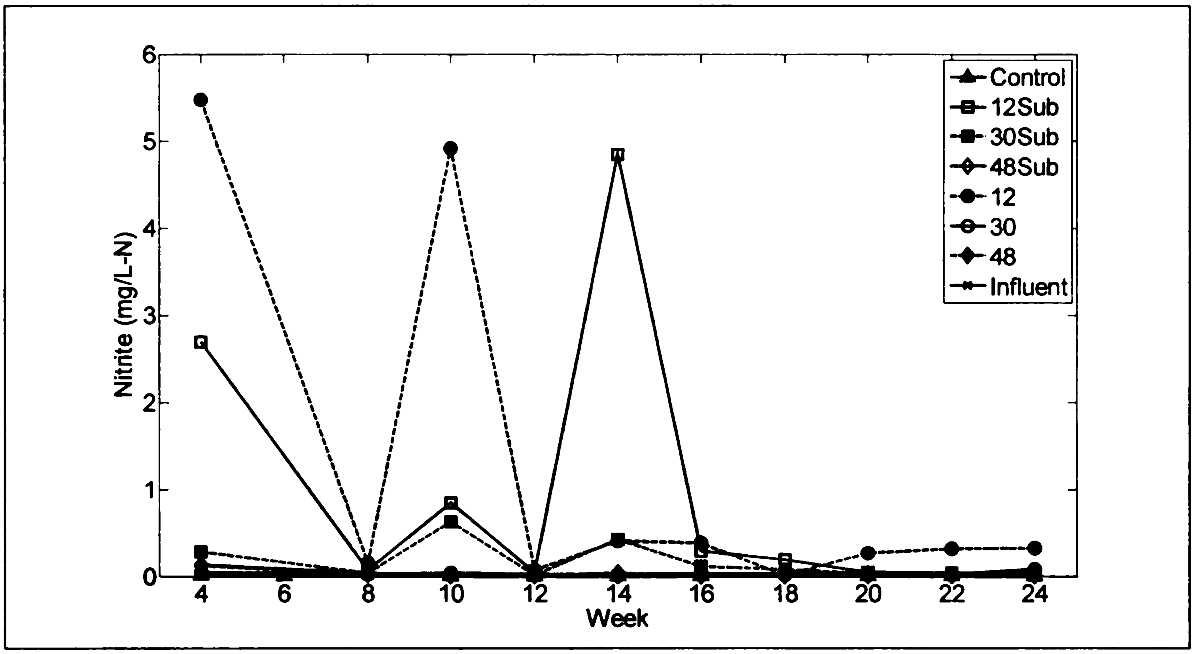

Figure 28: Sandy Loam Soil Column Nitrite Concentrations.

A statistical model was fit to the nitrite data which was significant for the main effects of soil, depth, and the repeated measure of time with a number of interaction effects, Table 56.

Table 56: Soil Column Nitrite Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	78.9	6.36	0.0137
Depth	2	64	4.01	0.0228
Soil*Depth	2	62.3	5.74	0.0052
Time	9	75.3	4.32	0.0001
Soil*Time	9	87.5	4.12	0.0002
Depth*Time	18	162	2.34	0.0026

There is a statistically significant difference in the nitrite effluent concentrations of the sand and sandy loam columns, Table 57. In terms of depth there is significant difference between the 12 inch columns and the longer columns, but not between the 30 and 48 inch columns.

Table 57: Soil Column Nitrite Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Depth (in)	Soil	Depth (in)	Estimate	Standard Error	DF	t Value	Pr > |t|
Soil	Sand		Sandy Loam		-0.3147	0.1248	78.9	-2.52	0.0137
Depth		12		30	0.5443	0.1914	77.8	2.84	0.0057
Depth		12		48	0.5292	0.19	76.3	2.79	0.0067
Soil*Depth	Sand	12	Sandy Loam	12	-0.9835	0.3683	75.5	-2.67	0.0093
Soil*Depth	Sand	30	Sandy Loam	12	-1.0591	0.2189	79.3	-4.84	<.0001
Soil*Depth	Sand	48	Sandy Loam	12	-0.9782	0.216	76.5	-4.53	<.0001
Soil*Depth	Sand	48	Sandy Loam	48	0.08563	0.03574	34.9	2.4	0.0221
Soil*Depth	Sandy Loam	12	Sandy Loam	30	1.0129	0.2176	78	4.65	<.0001
Soil*Depth	Sandy Loam	12	Sandy Loam	48	1.0638	0.2157	75.9	4.93	<.0001

Nitrate is the final step in the nitrification process and can indicate if the rates of nitrification/denitrification are not in sync. Influent nitrate concentrations are ~1 mg/L and unlike many other parameters in the sand columns increase with depth, Figure 29. Average effluent concentrations increase from 12 mg/L for the 12 inch sand column to 27 and 24 mg/L for the 30 and 48 inch columns respectively. Denitrification is inhibited by the presence of oxygen and low pH values (optimal pH from 6-8) (Paul and Clark 1996). Rates for denitrification, so long as organic carbon is available, require a water-filled pore space of 60–90% (Paul and Clark 1996). Values for pH are within the optimal range for denitrification, section 4.3.5, and there is available organic carbon as seen by the

BOD_5, COD and TOC concentrations, indicating denitrification is again controlled by oxygen and water content.

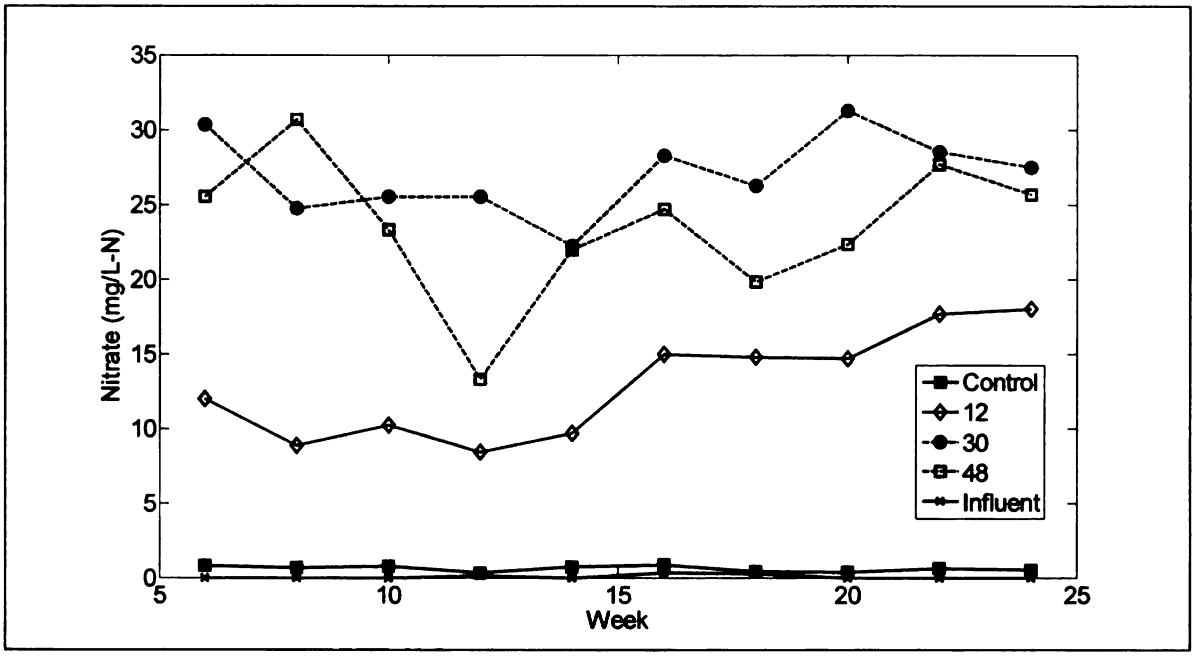

Figure 29: Sand Soil Column Nitrate Concentrations.

The sandy loam columns follow the same trends as the sand columns, with an increase in nitrate with depth, Figure 30. In this case the average nitrate concentrations for the submerged sandy loam columns increase with increasing depth from 10 mg/L, to 19 mg/L, to 29 mg/L for the 48 inch column. Non-submerged column nitrate effluent increases from 16 mg/L, to 19 mg/l, to 40 mg/L with depth. Nitrate conversion to nitrogen gas requires anaerobic conditions which have been evidenced to occur within the short columns only. This is thought to have caused the nitrate build-up within the longer column effluent. Indicating that the soil moisture requirements for denitrification are either not met or are not sustained.

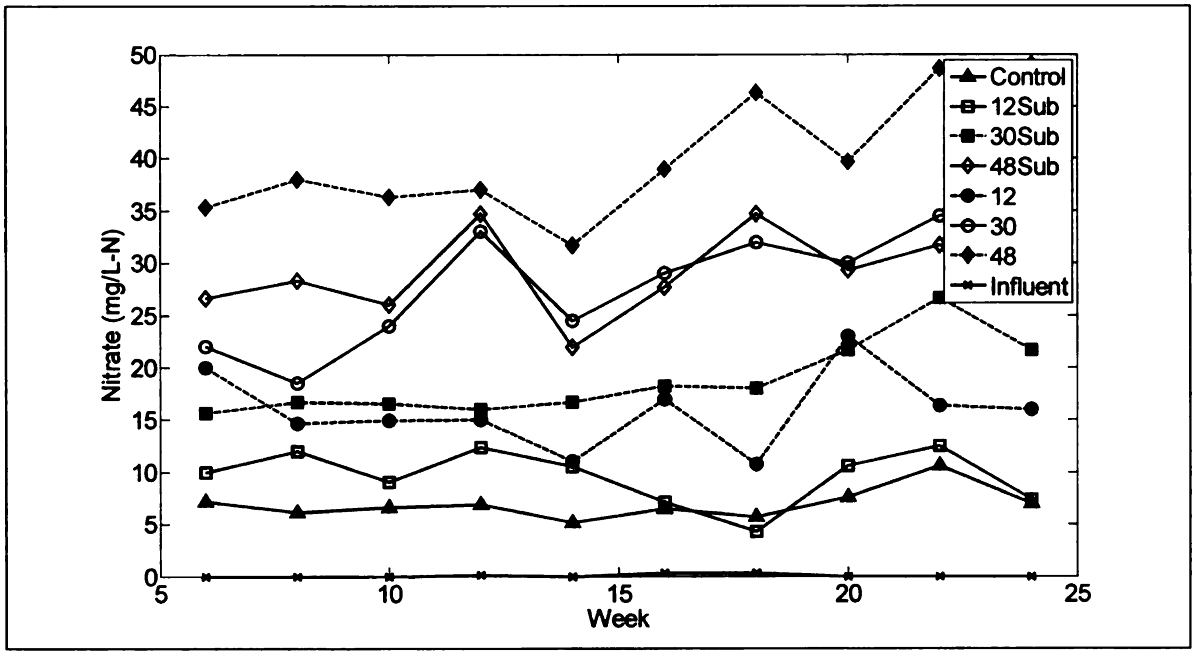

Figure 30: Sandy Loam Soil Column Nitrate Concentrations.

The details for the statistical model for nitrate are in Table 58. Significant main effects for soil, column, and submergence were found.

Table 58: Soil Column Nitrate Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	184	14.44	0.0002
Depth	2	224	33.06	<.0001
Soil*Depth	2	223	13.7	<.0001
Sub	1	189	25.79	<.0001
Time	9	265	4	<.0001

There is statistical significance between the soil types for nitrate effluent concentrations. Long columns of 30 and 48 inches are statistically different from the 12 inch columns. Sandy loam columns show significance for an increase in nitrate with increasing depth, Table 59.

Table 59: Soil Column Nitrate Differences of Least Squares Means – Comparison of Significance

Effect	Soil	Sub	Depth (in)	Soil	Sub	Depth (in)	Estimate	Std Error	DF	t Value	Pr > |t|
Soil	Sand			Sandy Loam			-9.1463	2.4071	184	-3.8	0.0002
Depth			12			30	-17.8048	2.4032	224	-7.41	<.0001
Depth			12			48	-19.9701	2.6394	220	-7.57	<.0001
Soil* Depth	Sand		12	Sand		30	-24.1811	4.1512	221	-5.83	<.0001
Soil* Depth	Sand		12	Sand		48	-16.2928	4.4818	224	-3.64	0.0003
Soil* Depth	Sand		12	Sandy Loam		12	-10.9456	4.5699	202	-2.4	0.0175
Soil* Depth	Sand		12	Sandy Loam		30	-22.374	4.1479	216	-5.39	<.0001
Soil* Depth	Sand		12	Sandy Loam		48	-34.5931	4.8526	173	-7.13	<.0001
Soil* Depth	Sand		30	Sand		48	7.8883	2.7565	271	2.86	0.0045
Soil* Depth	Sand		30	Sandy Loam		12	13.2355	2.906	228	4.55	<.0001
Soil* Depth	Sand		30	Sandy Loam		48	-10.412	3.339	161	-3.12	0.0022
Soil* Depth	Sand		48	Sandy Loam		30	-6.0813	2.7494	267	-2.21	0.0278
Soil* Depth	Sand		48	Sandy Loam		48	-18.3003	3.7416	178	-4.89	<.0001
Soil* Depth	Sandy Loam		12	Sandy Loam		30	-11.4284	2.4181	234	-4.73	<.0001
Soil* Depth	Sandy Loam		12	Sandy Loam		48	-23.6475	2.8013	205	-8.44	<.0001
Soil* Depth	Sandy Loam		30	Sandy Loam		48	-12.2191	2.8364	164	-4.31	<.0001
Sub		No			Sub mer ged		11.3549	2.2361	189	5.08	<.0001

4.3.4 Phosphorus

Complications with phosphorus testing methods result in validity in general trends only, not in concentrations. The general trends indicate removal percentages from 25-75%, but removal has a high rate of deviation and no conclusions can be drawn with confidence. However, phosphorus removal is based on adsorption within the soil profile so will theoretically have increased removal for greater depths as the residence time for assimilation is greater, and there is more surface area within the larger columns for adsorption.

4.3.5 pH/Alkalinity

The pH values for sand soils can be found in Figure 31 and Figure 32 for sandy loam soils. Average pH values for all treatment columns are between 7.3 and 8.0. All columns produce a neutral pH and pose no issues for treatment.

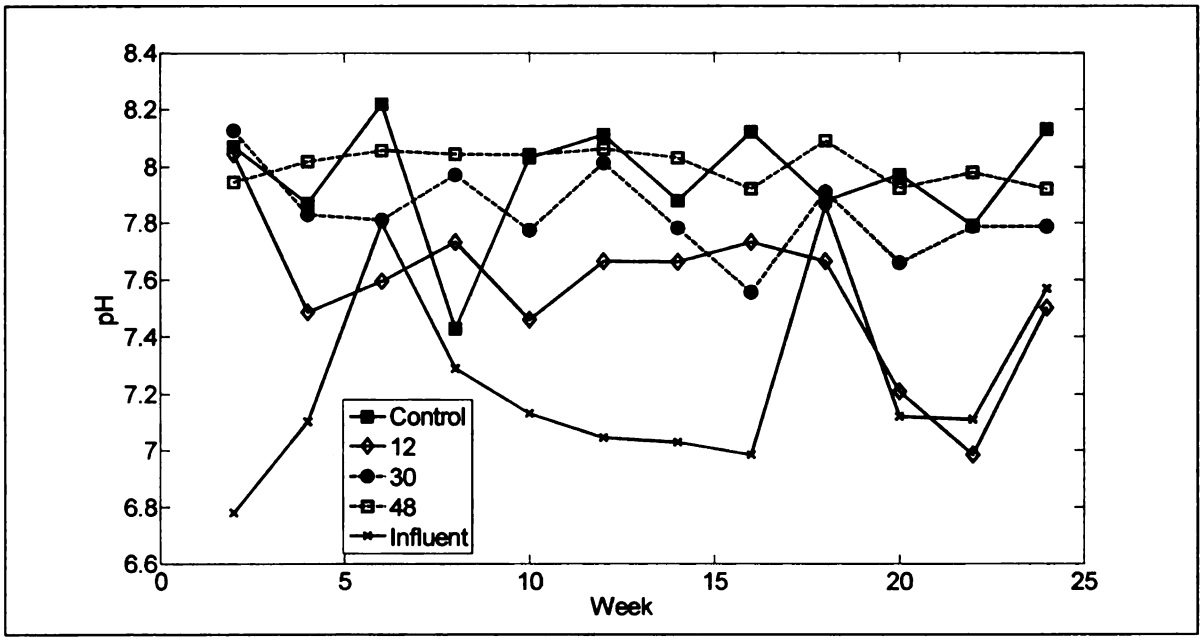

Figure 31: Sand Soil Column pH Concentrations.

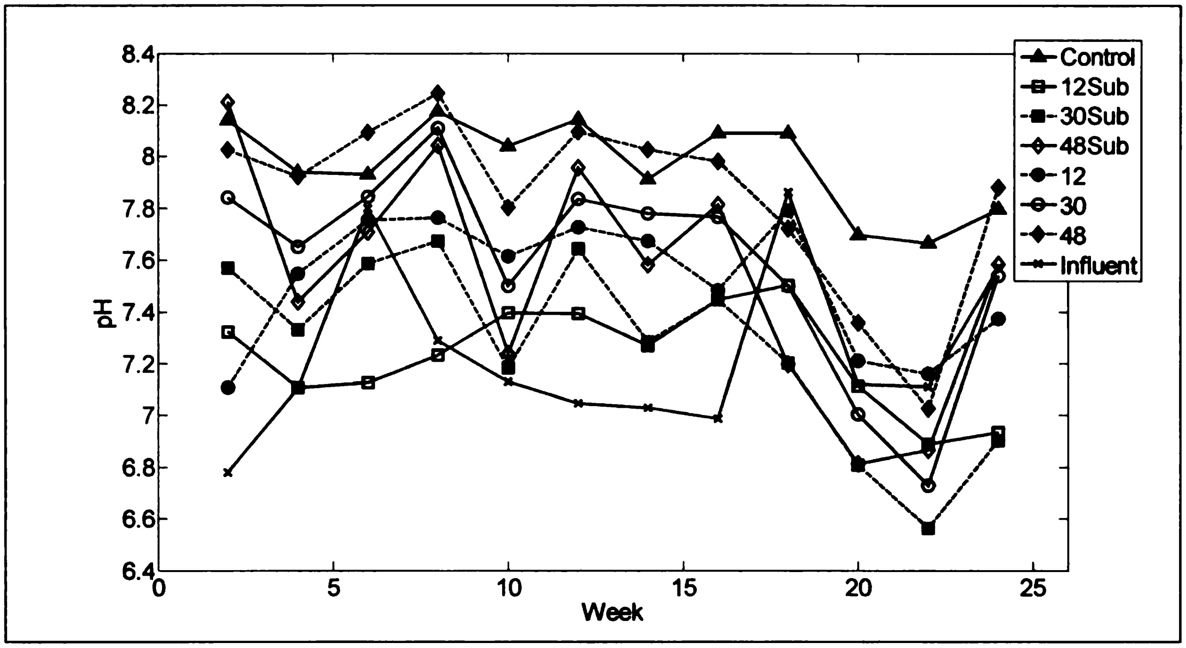

Figure 32: Sandy Loam Soil Column pH Concentrations.

The statistical model for pH produced no statistical significance for all effects, indicating that the treatment means are not significantly different for pH.

Alkalinity influent concentrations were measured at 336 mg/L as $CaCO_3$ for the synthetic wastewater and 340 mg/L as $CaCO_3$ for water applied to the control columns. Effluent from the sand columns produced average alkalinity values of 293 mg/L as $CaCO_3$, 239 mg/L as $CaCO_3$, and 251 mg/L as $CaCO_3$ for increasing depths, which was similar to the control column with an average of 253 mg/L as $CaCO_3$, Figure 33.

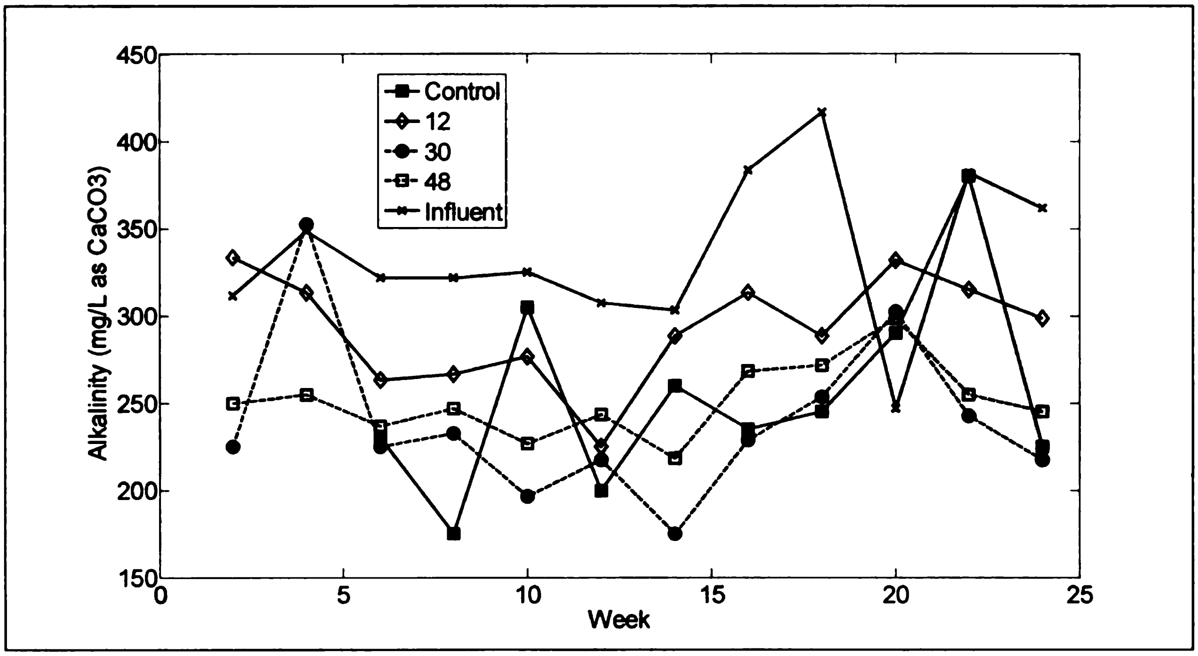

Figure 33: Sand Soil Column Alkalinity Concentrations.

Alkalinity concentrations for submerged sandy loam columns averaged between 303-335 mg/L as $CaCO_3$, the non-submerged from 262-294 mg/L as $CaCO_3$, Figure 34.

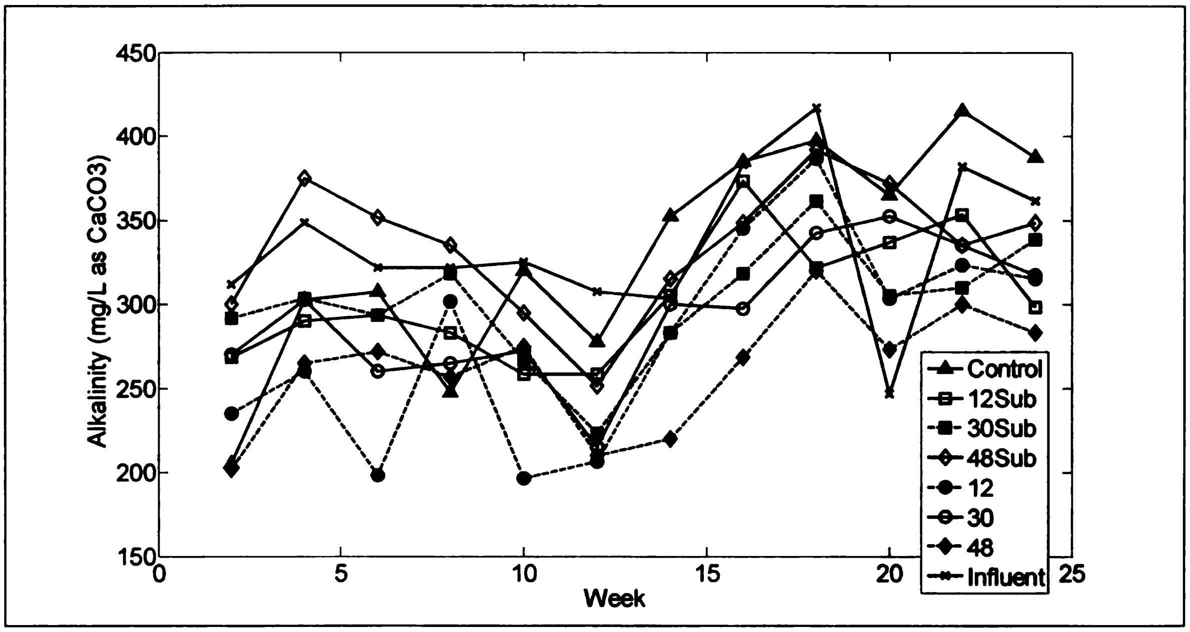

Figure 34: Sandy Loam Soil Column Alkalinity Concentrations.

The statistical model for alkalinity is shown in Table 60.

Table 60: Soil Column Alkalinity Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	20.5	1.61	0.2186
Depth	2	63.6	6.83	0.0021
Soil*Depth	2	20.3	5.09	0.0162
Sub	1	20.6	15.6	0.0008
Depth*Sub	2	21.8	4.38	0.0252
Time	11	113	16.38	<.0001
Soil*Time	11	113	4.53	<.0001

There is a statistically significant difference between all depths and those columns that are submerged and not submerged, Table 61.

Table 61: Soil Column Alkalinity Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Sub	Depth (in)	Soil	Sub	Depth (in)	Estimate	Std Error	DF	t Value	Pr > |t|
Depth			12			30	32.9088	13.477	51.7	2.44	0.0181
Depth			30			48	-41.1901	11.7213	101	-3.51	0.0007
Soil* Depth	Sand		12	Sand		30	70.0459	24.6058	40.6	2.85	0.0069
Soil* Depth	Sand		30	Sand		48	-72.1576	20.3979	40.8	-3.54	0.001
Soil* Depth	Sand		30	Sandy Loam		12	-56.5002	15.6832	74.9	-3.6	0.0006
Soil* Depth	Sand		30	Sandy Loam		30	-60.7283	13.0973	207	-4.64	<.0001
Soil* Depth	Sand		30	Sandy Loam		48	-70.9509	16.2391	29.2	-4.37	0.0001
Sub		No			Subm erged		-44.678	11.3128	20.6	-3.95	0.0008
Depth* Sub		No	12		Subm erged	48	-70.0605	20.7665	24.8	-3.37	0.0024
Depth* Sub		Subm erged	12		No	30	54.4062	17.9676	48	3.03	0.004
Depth* Sub		Subm erged	12		Subm erged	30	43.9307	22.2405	58.3	1.98	0.053
Depth* Sub		Subm erged	12		No	48	53.4981	20.0274	22.9	2.67	0.0137
Depth* Sub		No	30		Subm erged	48	-91.9475	17.647	29.9	-5.21	<.0001
Depth* Sub		Subm erged	30		Subm erged	48	-81.472	21.9823	41.2	-3.71	0.0006
Depth* Sub		No	48		Subm erged	48	-91.0394	21.8758	9.87	-4.16	0.002

Larger decreases in alkalinity concentrations for columns which had increased nitrification is in agreement with nitrogen columns data as nitrification uses carbonate as a carbon source for aerobic metabolism.

4.3.6 Metals

Influent Mn concentrations for the wastewater were 550 ug/L, and 436 ug/L for the water applied to the control columns. Sand columns had reduction percentages of 63% for the 12 inch columns and 98% for the 30 and 48 inch columns. Final Mn effluent concentrations for the 12, 30, and 48 inch column depths were 109 ug/L, 5 ug/L, and 7 ug/L, Figure 35.

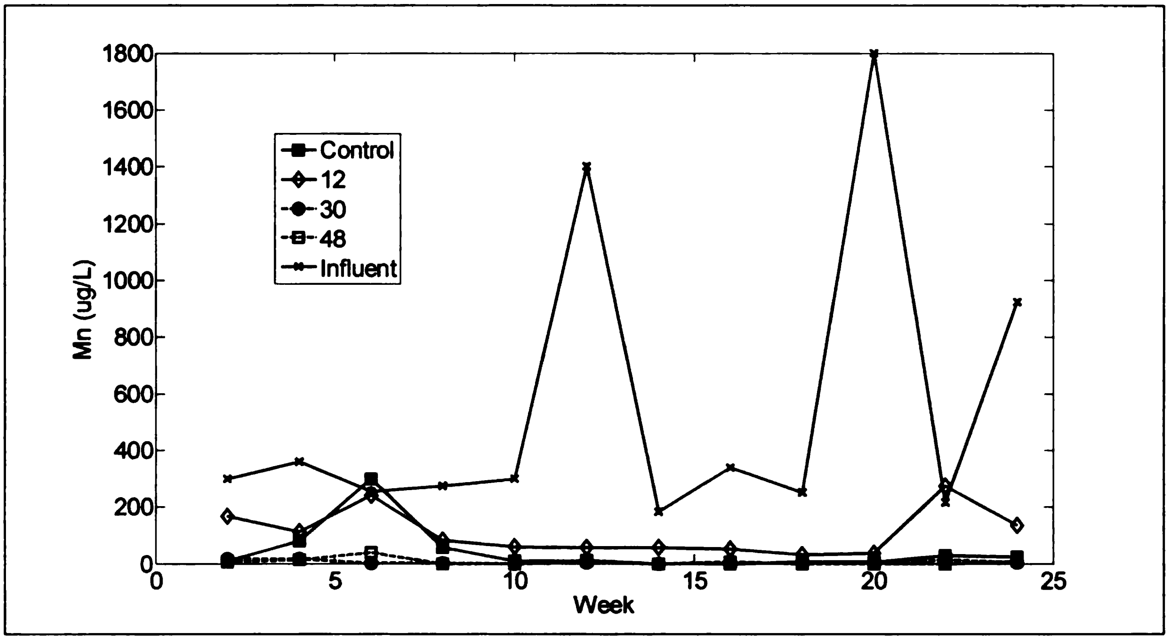

Figure 35: Sand Soil Column Mn Concentrations.

Sandy loam submerged columns produced average removal percentages of -66%, 0%, and 17% with increasing depth, with final effluent concentrations of 608 mg/L, 304 ug/L, and 256 ug/L, Figure 36. Leaching of Mn within the 12 inch columns confirms that the conditions within the 12 in columns were anaerobic, and greater depths maintained more aerobic conditions. In addition, the 12 inch column has greater effluent concentrations as the column is short enough to allow leaching metals from the rest of the column to reach the bottom without re-oxidizing to an insoluble form. The non-submerged columns had improved removal percentages with depth from 25% for 12 inch columns to 46% and 85% for the 30 and 48 inch columns, respectively. Final average effluent concentrations had values of 290 ug/L, 150 ug/L, and 37 ug/L reducing concentrations with increasing depth.

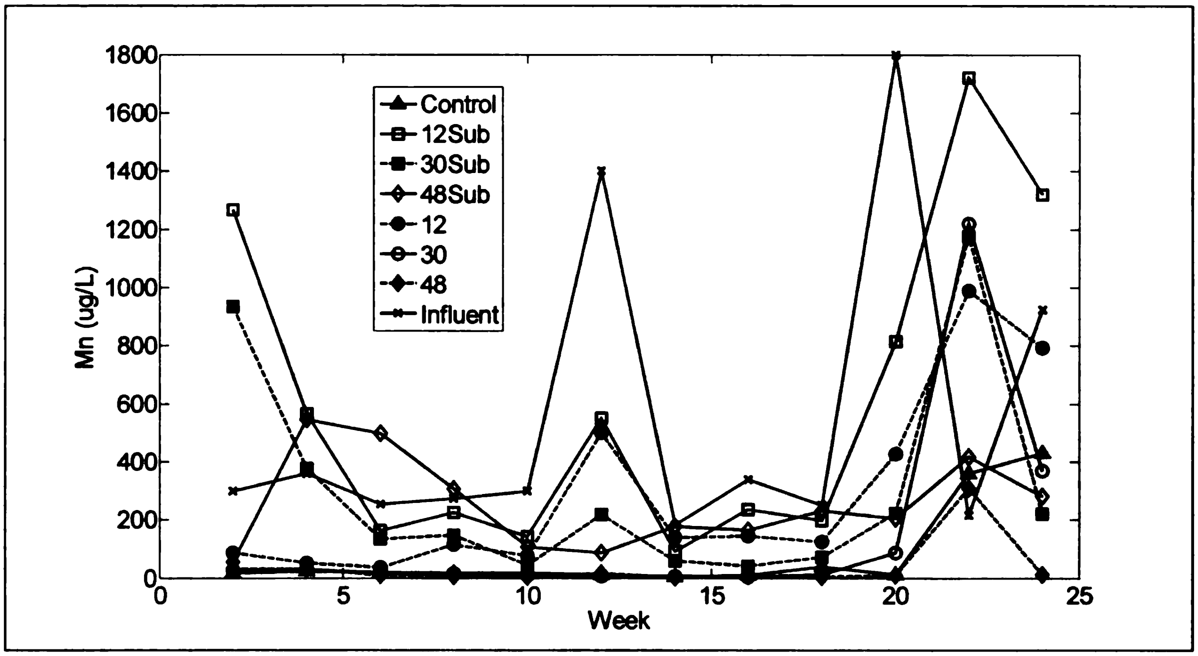

Figure 36: Sandy Loam Soil Column Mn Concentrations.

Initial soil Mn concentrations are 3x greater within the sandy loam soil than the sand soil, Table 47. This difference in initial soil concentration may contribute to the differences in subsurface effluent concentrations.

Table 62 includes the details for the statistical model for Mn. The model produced significance for the all levels of depth, between soils, and between submerged and non-submerged columns validating the differences noted above, Table 63.

Table 62: Soil Column Mn Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	71.1	9.7	0.0027
Depth	2	64	13.69	<.0001
Sub	1	50.2	63.83	<.0001
Time	11	153	5.96	<.0001
Soil*Time	11	71.1	4.36	<.0001
Depth*Time	22	89.1	3.1	<.0001
Sub*Time	11	50.2	2.49	0.0141

Table 63: Soil Column Mn Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Sub	Depth (in)	Soil	Sub	Depth (in)	Est	Standard Error	DF	t Value	Pr > |t|
Depth			12			30	192.39	45.3612	101	4.24	<.0001
Depth			12			48	235.79	45.7017	109	5.16	<.0001
Depth			30			48	43.4007	19.4957	33.9	2.23	0.0327
Soil	Sand			Sandy Loam			-75.916	24.3724	71.1	-3.11	0.0027
Sub		No			Subm erged		-208.5	26.0969	50.2	-7.99	<.0001

Soil Mn concentrations as a function of depth show a decrease in Mn to a depth of 12 inches where the concentrations levels off, Figure 37. This decrease again supports the conclusions from the statistical analysis and effluent trends which show a significant difference between the 12 inch columns and those deeper, again supporting the conclusion that the 12 inch soil column is not significant for treatment of Mn. This data also suggests that sorption may be significant within the sand columns but does not appear to be within the sandy loam columns.

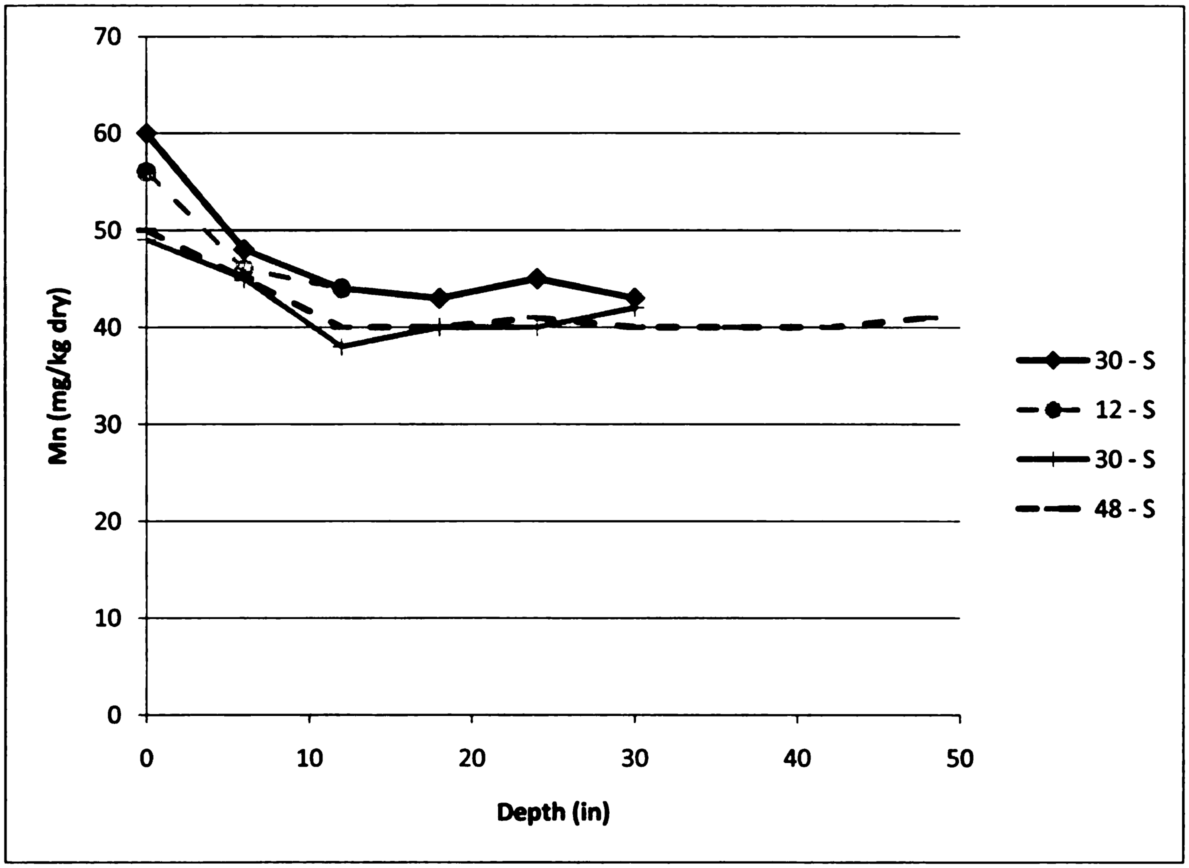

Figure 37: MnSoil Concentration as a Function of Depth

According to the electron tower, Fe is leached following Mn leading to the mobilization of iron when it is reduced from Fe(III) to Fe(II). Leaching of Fe before the complete leaching of Mn is due to the high levels of soluble Fe added as compared to the insoluble form of Mn. Influent average Fe concentrations for the synthetic wastewater and the tap water applied to control columns were 2050 ug/l and 1600 ug/L. Sand columns reached removal percentages and final effluent concentrations of 37% and 677 ug/L for the 12 inch columns, 94% and 63 ug/L for the 30 inch columns, and 91% and 81 ug/l for the 48 inch columns. Sand column at a depth of 12 inches performed much more poorly than those at greater depths, Figure 38. Control columns performed similarly to the 30 and 48 inch columns.

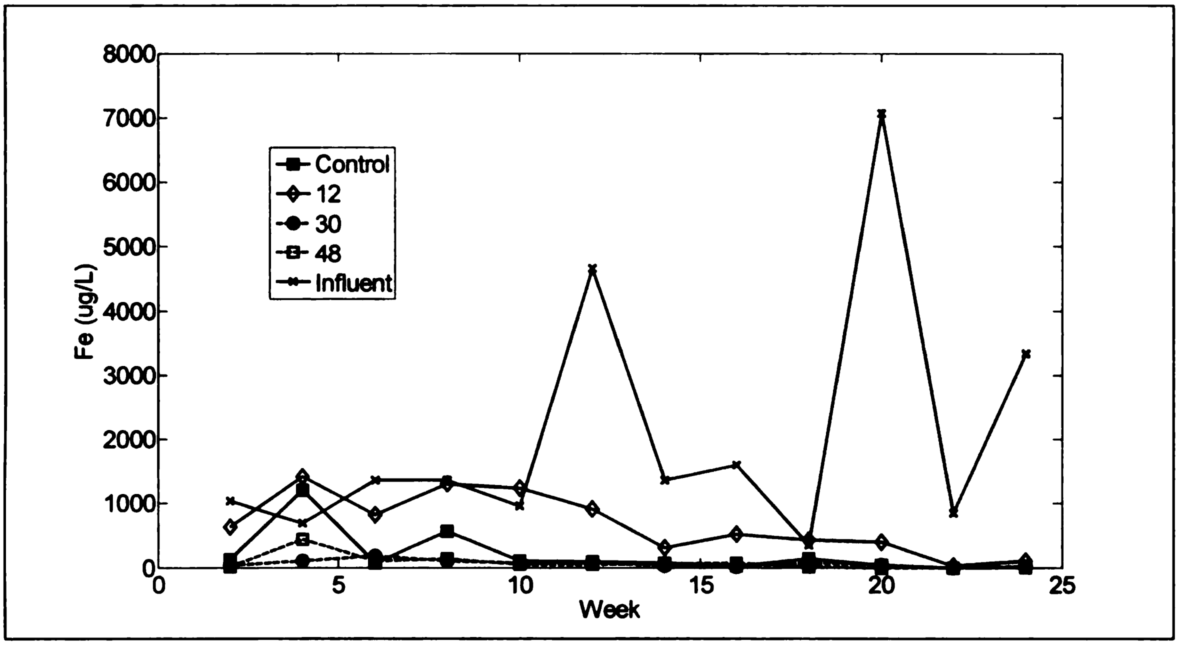

Figure 38: Sand Soil Column Fe Concentrations.

Average Fe removal percentages for the submerged sandy loam columns for depths of 12, 30, and 48 inches are -50%, 38%, and 31%. Final effluent concentrations for the submerged columns with increasing depth are 1631 ug/L, 760 ug/l, and 794 ug/L. The non-submerged columns had increased removal rates of 23%, 44%, and 34% for the 12, 30, and 48 inch sandy loam columns with final concentrations of 1026 ug/L, 658 ug/l, and 778 ug/L, Figure 39. The average final effluent concentration for the control columns was 722 ug/L, very similar to the 30 inch and 48 inch submerged and non-submerged columns.

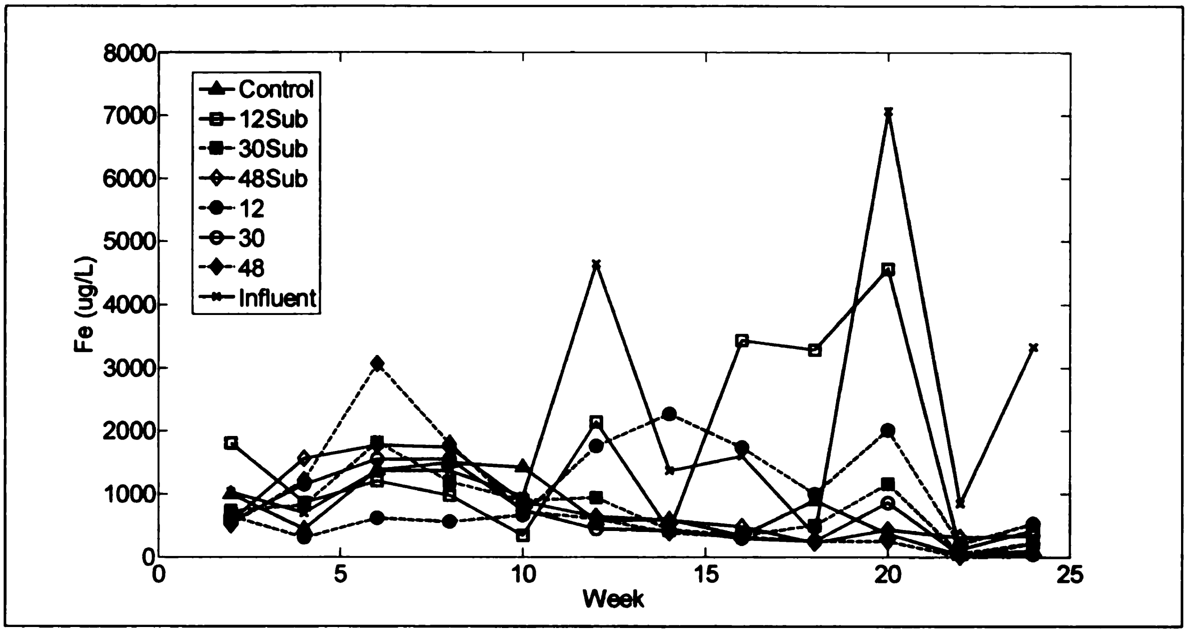

Figure 39: Sandy Loam Soil Column Fe Concentrations.

The statistical model for Fe is in Table 64. Statistical analysis indicated there was a significant difference for the 12 inch columns with the 30 and 48 inch columns, but there was no difference between the 30 and 48 inch Fe effluent concentrations, Table 65. Statistical analysis also indicated a significant difference for soil type. Looking at the removal percentages and the final effluent concentrations it is apparent that the sand was more effective in reducing Fe leaching.

Table 64: Soil Column Fe Statistical Model

Type 3 Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Soil	1	16.8	107.57	<.0001
Depth	2	21.4	8.67	0.0017
Time	11	117	2.55	0.0063
Soil*Time	11	312	13.61	<.0001
Depth*Time	22	202	2.91	<.0001

Table 65: Soil Column Fe Differences of Least Squares Means - Comparisons of Significance

Effect	Soil	Depth (in)	Soil	Depth (in)	Estimate	Standard Error	DF	t Value	Pr > |t|
Depth		12		30	645.41	153.76	82.2	4.2	<.0001
Depth		12		48	593.29	157.13	70.7	3.78	0.0003
Soil	Sand		Sandy Loam		-645.49	62.236	16.8	-10.37	<.0001

4.3.7 Plant Tissue

Plant tissue was analyzed after column deconstruction for nutrients and metal content. Plant tissue from the sand columns did not provide enough tissue mass for analysis. The control column had decreased nutrient content and increased metal concentrations as compared to columns with wastewater application, Figure 66. Differences in nutrient percentages within the columns applied with wastewater were not significant; however there were significant increases within some metal concentrations. No differences in nutrient percentages or metal concentrations were found with depth.

Table 66: Plant Tissue Concentrations by Soil Column

	12-SL	30-SL	48-SL	12-SL Sub	30-SL Sub	48-SL Sub	Control-SL
Nitrogen %	2.89	2.93	2.68	2.99	3.01	2.29	1.63
Phosphorus %	0.33	0.51	0.44	0.36	0.35	0.32	0.19
Potassium %	4.65	5.53	4.14	3.09	4.14	3.06	1.35
Calcium %	0.79	0.64	0.8	1.31	0.72	0.77	0.8
Magnesium %	0.49	0.45	0.44	0.56	0.32	0.48	0.27
Sodium %	0.14	0.08	0.07	0.17	0.04	0.09	0.03
Sulfur %	0.27	0.28	0.26	0.44	0.24	0.24	0.18
Iron (ppm)	456	515	1417	2243	914	1483	6037
Zinc (ppm)	33	38	62	67	38	48	74
Manganese (ppm)	31	50	49	89	49	55	172
Copper (ppm)	12	14	22	37	21	14	51
Boron (ppm)	9	12	9	9	10	11	6
Aluminum (ppm)	134	186	655	904	341	632	2620

The increase in nutrient within the columns with wastewater application is expected due to the increase in available nutrients for uptake. The increase in the concentration of metals within the tissue of the 12 inch submerged columns with applied wastewater compared to the non-submerged scale indicates the availability of soluble metals with increased water content and corresponding anaerobic conditions. High metal concentrations within the control columns indicate possible competition with microbial biomass, Figure 40 & 41.

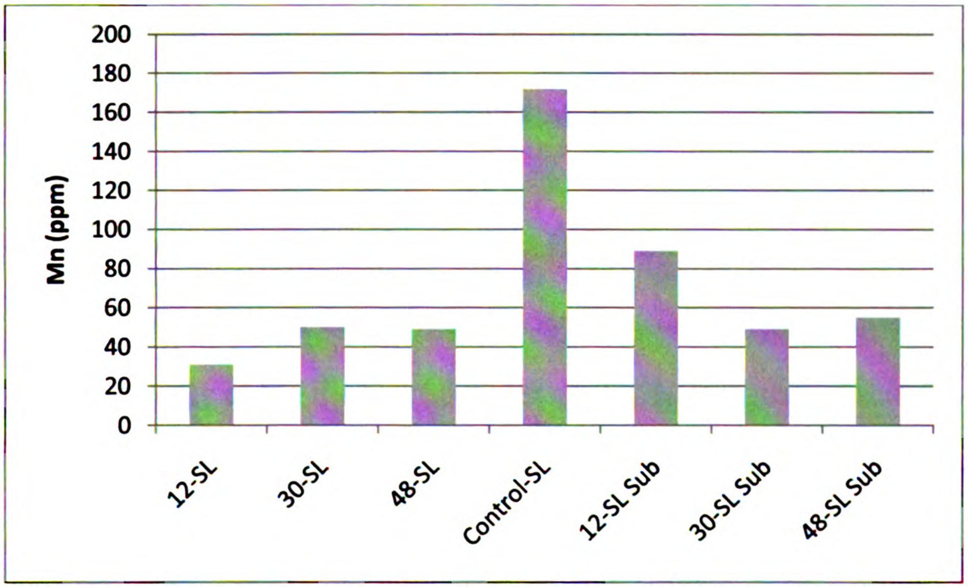

Figure 40: Mn Plant Tissue Concentrations by Soil Column

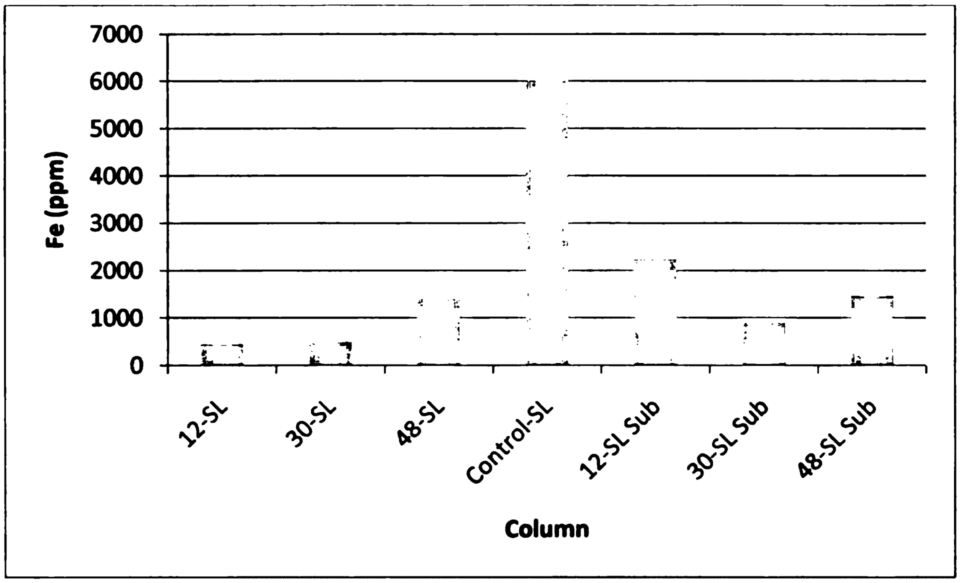

Figure 41: Fe plant Tissue by Soil Column

4.4 Comparison of Treatment from Field and Laboratory Studies

Soil columns of equal depth had greater removal percentages than field studies for the majority of the measured water quality parameters, Table 66. The table includes field and soil column removal percentages (shaded) in order to compare like soils and depths. Columns follow general trends found within the field system, although there are significant differences particularly at the MSU dairy site which can be explained by the experimental conditions explained below. However, it should be noted that there are significant differences in performance, but the general trends and the conclusions drawn hold true for the data presented.

Table 67: Removal Percentages (%) for Filter Strips in Comparison to Soil Columns

Location	BOD	COD	TKN	Ammonia	Nitrite	Nitrate	Alkalinity	pH	Mn	Fe
Filter Strip 1	28	18	32	30	-694	26	-32	1	-343	-2
Filter Strip 2	-6	4	20	22	-1203	-164	-227	-9	-375	27
12 SL	58	43	56	60	-1113	-10580	16	-4	25	23
12 SL sub	56	45	55	63	-5566	-6581	8	0	-66	-50
Small MI Dairy 1.5 ft	89	85	87	84	-20	4	54	10	-27	43
12 S	95	93	85	85	-300	-8311	11	-5	63	37
Small MI Dairy 2.5 ft	79	86	80	73	-16	36	60	10	-8	58
30 S	95	94	98	98	-495	-17348	27	-8	98	94

MSU dairy filter strips do not perform as well as soil columns of similar characteristics, the 12 inch sandy loam submerged and non-submerged. This can be attributed in part to the influent/effluent relationship of the sandy loam soils found at the MSU site. The greater increases in the influent concentrations have a direct effect on the higher effluent concentrations leading to reduced removal percentages. Additionally, it has been shown that removal is thought to be based primarily on oxygen availability within the soil. The soil at the MSU dairy site has an average clay content of 11.3%, whereas the soil column content has a reduced average clay content of 4.3%, reducing the soil porosity and increasing the soil moisture holding capacity at the MSU site, Table 67. An increase in clay reduces soil oxygen diffusion rates, increases the soils ability to retain moisture, and reduces overall oxygen within the soil prior to wastewater application, limiting available oxygen. The BOD_5, COD, TKN and ammonia reduction percentages all support this theorized difference in oxygen availability. Increases in nitrification that result due to increased oxygen also result in the lower concentrations of alkalinity within the soil columns. Increases in nitrite concentrations are theorized as a result of the stress on the nitrifying bacteria due to low dissolved oxygen required for conversion to nitrate, and the increase in the hetertrophic bacteria which can dominate and reduce nitrifying bacteria (Tchobanoglous et al. 2003). Both of these theories are supported by data in which there is less oxygen availability within the shorter columns and an increase in BOD removal requiring heterotrophic bacteria (as it is shown that 12 inches is not sufficient to remove BOD). Smith et al. (2003) also indicate that an accumulation of nitrite will only occur due to increases in pH, decreases in dissolved oxygen, or inhibition by free ammonia (which plays a more significant role than pH), or high organic matter (Master et al. 2004). A build up of nitrate within the system, as indicated by the negative removal, would be expected to occur as data indicated that there was oxygen inhibiting denitrification. The 12 inch submerged columns and the field system all leached more metals than what was present in the influent in contradiction to the 12 inch sandy loam columns that provided some metal removal. The submerged columns were anaerobic at the bottom of the column which can result in the leaching of metals. Non-submerged columns were exposed to the air at the bottom, and it is theorized that this allowed increased oxygen diffusion and more aerobic conditions. It can also be theorized that the increase in the initial concentration of Mn and Fe in the filter strips had an impact on the final concentrations within the leachate.

Table 68: Filter Strip Soil Characteristics

Parameter	Filter Strip 1	Filter Strip 2	Sandy Loam Columns	Sand Columns
pH	7.4	7.5	6.9	8.8
P (ppm)	29	31	98	3
K (ppm)	106	112	133	8
Ca (ppm)	1239	1412	966	632
Mg (ppm)	223	247	198	198
Zn (ppm)	5.2	3.4	4.9	3.1
Mn (ppm)	32.7	40.5	13.9	4.4
Cu (ppm)	2.8	3.0	13.9	0.8
Fe (ppm)	60.7	71.5	44.7	8.1
Organic Matter (%)	2.9	3.2	2.3	0.3
Carbon (%)	1.7	1.9		
Chloride (ppm)	56	55	59	61
Total N (ppm)	0.14	0.17	0.10	n.d.
Nitrate-N (ppm)	1.83	2.45	11.0	0.6
Ammonium-N (ppm)	1.45	1.99	1.4	0.5
Sand (%)	61.1	61.2	69.8	93.5
Silt (%)	26.2	28.9	25.9	2.8
Clay (%)	12.7	9.9	4.3	3.7
Soil Type	Sandy Loam	Sandy Loam	Sandy Loam	Sand

The small MI dairy site resulted in much closer removal rates for the full-scale implementation as compared to the soil columns for BOD_5, COD, TKN, and ammonia. Again, significant build-up in nitrite may indicate toxicity. Increases of nitrate within the soil columns are theorized to be due to aerobic conditions from the exposure of the bottom of the column to the atmosphere.

Final effluent concentrations from the columns for sandy loam soils are much lower than the effluent from the filter strips, which is highly dependent of the influent concentrations. The sand columns reflect the conclusion that the soil was aerobic as there was significant reduction in nitrogen from nitrification and denitrification. However, the metal leaching and a reduction in nitrate at the small MI dairy site indicate that reducing conditions may have occurred.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

Examining feedlot sources for runoff quality was able to identify the most problematic sources in terms of pollutant loadings to provide recommendations for on-farm management. Nine storm events were used to characterize farmstead runoff pollutant sources. The heat check lot produced the greatest concentrations of COD, BOD_5, ammonia, TKN, SO_4^-, solids, TOC, and Cl^-, however, was only 9% of the drainage area. Runoff from the roadway had substantially lower concentrations resulting in a more significant effect on the composite concentrations. Feed sources did not produce as great of average concentrations as the heat check lot, but due to its large surface area, 29% of the drainage area, posed greater concern to waterways and treatment. Overall composite concentrations from feed areas and associated roadways were too great for complete removal in agricultural filter strips. In particular, the low pH values between 4 and 5 from feed sources impede biological treatment and burned the vegetation. The upright silos also produce a significant amount of arsenic within runoff, posing environmental and human health concerns as the concentrations were over the US EPA 10 ug/L drinking water standard. Water quantity and dilution proved to be the determining factor to pollutant loading concerns and allocation of management practices.

The fall months were responsible for a large portion of the high concentrations of pollutants in the feed sources due to leaching from the upright silos. Consequently, management of feed sources requires allocation of resources to ensure proper upright silage filling practices. Additionally, bunker silos need to be covered and swept prior to precipitation and feed faces need to be maintained to limit the transport of pollutants from this large surface area. If manure is a larger component than in this study, covering the manure prior to precipitation events or providing barriers such as berms or curbs can limit the transport of pollutants.

A field study was conducted to determine pollutant removal of three agricultural filter strips within the soil profile to determine the potential impact to groundwater. The three filter strips were designed and operated according to the NRCS standard to treat farmstead runoff. Two filter strips were installed at the MSU 160 cow dairy at and the third at a 40 cow small MI dairy. Comparison of the three filter strips revealed that the small MI dairy site had greater overall pollutant removal suspected to be due to a bioretention area that provided a large percentage of removal before application of runoff to filter strips, and the sandy soils which provided characteristics to improve oxygen availability and reduce the moisture within the soil subsurface. However, these soils still produced nitrates and metal leaching which posed environmental ground water concerns. The MSU filter strip 1 (heat check lot and roadway sources) performed better in terms of removal than the second MSU filter strip (feed runoff sources). Poor performance of the second filter strip is suspected to be due to the greater influent concentrations, as subsurface samples had higher conductivity, indicative of salt and soluble nutrient build-up over time, which is consistent with overloading. In addition, performance initially was impacted by the preferential flow to one side of the filter strip 2 resulting from improper grading, determined to be a critical design component.

Based on the results from all three sites (10 sampling events at the MSU dairy, and 6 at the small MI dairy site) subsurface samples indicated that a soil depth of 1 to 1.5 feet is not capable of eliminating pollutants to a degree suitable for groundwater protection. Concentrations of BOD_5 were well above the 30 mg/L discharge limit. The greatest removal still resulted in average subsurface effluent concentration averages of 150 mg/L, however this is similar to the level from a septic tank. Increases in nitrite concentrations occurred in all systems, which is unusual and may indicate toxicity. Nitrate values were consistently over the 10 mg/L standard. A build up of nitrate within the system, as indicated by the negative removal, may be a result of available oxygen inhibiting denitrification. Arsenic concentrations were also over the 10 ug/L drinking water standard, particularly at the small MI dairy filter strip. The greater sources of arsenic at this site were theorized to be due to influences from high groundwater arsenic levels being transported through excess plate cooler water entering the storage basin. Metal leaching was also a concern for groundwater sources as those measured in subsurface effluent were in greater concentrations than were in the waste stream prior to infiltration. Metal leaching and a reduction in nitrate at the small MI dairy site indicate that reducing conditions may have occurred. Overall, the small MI dairy had higher removal rates in 1.5 and 2.5 feet deep subsurface samples compared to the first foot in the MSU subsurface samples, but final effluent concentrations indicated potential problems for nitrate and metal leaching.

Improved performance of the MSU site can be achieved through a variety of operations and design alterations. As was indicated by the influent to effluent concentration correlations, an increase in management to reduce influent concentrations would have a significant effect on effluent concentrations. Additionally, mechanical aeration in the basin of within the field (using a lawn aerator or venting within the soil profile) can provide an increase in oxygen required to increase treatment. A further measure of backfilling sand has potential to increase aeration and decrease moisture within the soil profile to increase performance.

Cold weather performance was evaluated at the MSU site only and was found to have no effect on performance, although the poor performance of these systems year round may influence the difference realized with seasonal variation, as a correlation may result in a more efficient system. However several deductions could be made from analysis of daily operation. Due to the transport system design requiring increased temperatures for application of waste to the treatment systems it is likely that the filter strips will have unfrozen soil subsurfaces during a runoff application ina thaw event, reducing the runoff from impermeable soils.

However, there is a need to investigate the temperature effect on microbial degradation in a more efficient treatment system as microbial populations are known to be affected by temperature differentials.

Results from the field research indicated the need for a laboratory study which could investigate treatment depth with greater control of the environmental and experimental conditions. Further investigation over a 7 month period examined the relationship of depth of treatment, soil type, and groundwater interaction on pollutant removal. Depth was determined to be a significant factor for all measured pollutant subsurface effluent concentrations. There was a significant difference in removal at 12 inches compared to that at greater depths. Removal percentages for many parameters increased into the 90% range for depths of 30 inches or greater.

Soil was also determined to be a statistically significant as a main effect for all measured parameters except alkalinity. Sand soil had greater pollutant removal percentages than sandy loam soils for almost all parameters. Sandy loam soils also had significant increases in Mn leaching. Soil physical properties have a significant impact on soil moisture and are largely responsible for the oxygen availability within the soil profile. Soils with a high porosity, such as sand, have a high oxygen diffusion rate and the larger macro pore size decreases the soil moisture holding capacity enabling the maintenance of high oxygen levels within the system. Oxygen availability was theorized to be rate limiting in the nitrification process. Nitrification processes within the soil, and lack of denitrification, indicate that the soil in columns greater than 12 inches had oxygen present. It is unknown whether sorption mechanisms were a significant factor in pollutant removal within the soil columns. Sand metal data indicated that the sorption of metal may have been a factor in reducing the metal concentrations. However, the increased CEC in the sandy loam soils due to the increased clay content and organic matter did not increase removal, therefore indicating that sorption was not the dominant factor in pollutant removal.

Column submergence was only statistically significant as a main effect for Mn, Alkalinity, Nitrate, and COD. However, many other parameters were significant for the interaction of depth and submergence. In general column submergence resulted in decreased pollutant removal.

Similar trends for the filter strip columns and the field data were observed, indicating that results from the soil column data can be extended to field scale filter strip design. Differences within the results were significant in some areas, but for the conclusions drawn the general trends were consistent and a significant portion can be explained by the experimental conditions. The MSU filter strips had sandy loam soils and subsurface samples were collected at an average of 1 foot from the soil surface, a similar design to the 12 inch sandy loam column. Higher treatment removal percentages are a result of the greater pollutant loadings as can be seen from the increase in concentrations to the decreased removal from filter strip 2, to filter strip 1, to the column results for 12 inch sandy loam soils. The MSU filter strips also received a larger hydraulic loading and had a significant increase in soil clay content, reducing oxygen concentrations as is indicated by the reduced BOD_5, COD, TKN, and ammonia results. Increased leaching of metals also indicated anaerobic conditions within the field, as indicated by the results from the submerged columns. Increases in metal leaching from the field were due to the more than 2x higher concentrations in the initial concentrations in field soils and the decrease in available oxygen. As with sandy loam soils, the 12 inch sand columns had on average greater removal percentages than the filter strips at the small MI dairy site, however the results were very similar. Again the differences in the concentrations can all be explained by the slight differences in the soil characteristics and the reduced oxygen concentrations. Comparison of the laboratory data to field results allowed for further interpretation of field performance and implications to the larger scale design, and include the following:

- Sand soils have greater performance than sandy loam soils, as increase in porosity increases oxygen diffusion and decreases the moisture holding capacity, and reductions in the ratio of wastewater volume to pore volume increases available oxygen. Ratios of wastewater volume to soil porosity should be below 0.27 to provide adequate oxygen. Backfilling filter strips with higher porosity soils will increase treatment, or potentially mechanically increasing soil porosity, or selecting vegetation that can increase oxygen diffusion and porosity.

- BOD_5 removal requires a depth of 30 inches for sand and sandy loam soils to reach a concentration below 30 mg/L. If groundwater is present within the system, it is recommended that the depth be increased to 48 inches. Appropriate site selection for filter strips at increased depth to groundwater will reduce groundwater impact.
- COD removal requires only 12 inches of depth in sand columns, as increases in depth do not produce a significantly greater removal, and a depth of 30 inches or greater for sandy loam soils. This is verified by BOD_5/COD ratios that indicate incomplete removal of biodegradable material in 12 inch columns.
- Removal of TKN and ammonia rely on nitrification and require 30 inches of treatment depth in sand. Sandy loam soils reach adequate treatment in 30 inch soils but realize an even greater removal in 48 inch soils. Nitrification rates are controlled by alkalinity availability, pH, temperature, and oxygen availability. All soils produce an excess of alkalinity and remain between the optimum pH, so if temperatures can be maintained between 5°C and 40°C, nitrification is based solely on oxygen. Design implications are to increase oxygen within the treatment system by increasing porosity to reduce the soil moisture holding capacity or increase oxygen diffusion as discussed above.

- Reductions in wastewater volumes at field locations can increase the available oxygen thereby increasing removal percentages. This can be accomplished by diverting clean water not required for dilution, or increasing the width or slope of the filter strip to increase the volume of soil used to treat runoff.

Siting plays a critical role in treatment efficiency and potential for contamination of ground and surface water. Siting should include the following:

- Location to surface water: minimum of 150 feet to surface water is required to reduce the potential for direct runoff discharge.
- Depth to groundwater: minimum of 30 inches is required, but may require an increase in depth depending upon soil type and loading.
- Soil type: hydraulic conductivities to eliminate ponding and increased porosities and a decrease in soil water holding capacity to increase oxygen diffusion; use of naturally occurring soils will reduce capital costs.
- Farm maintenance practices: minimizing open feed sources and general farm upkeep to determine potential loadings to the filter strip.
- Slope: adequate natural slope over 1%, or locations suited for excavation (increases in costs required for excavation).
- Available land area: availability of land adjacent to the farmstead area to reduce transport costs associated with pumping.

This research has identified that depth of soil and soil type play the most significant role in pollutant removal for land application systems, and identified application to filter strip design. Further research should focus on:

- Further investigating nitrite build-up, including possible toxicities.
- Reducing nitrate concentrations within the system prior to reaching groundwater, but without reducing the soil oxygen within the desired treatment depth outlined above. Possibilities include a defined clay layer to provide an anaerobic zone after aerobic treatment.
- Measurement of dissolved oxygen under various treatment conditions.
- Life cycle of soil in terms of metal adsorption.
- Optimization for vegetation in pollutant removal within the soil subsurface, particularly in regards to an increase in soil oxygen and the impact to soil conductivity.

Results provide insight to source characterization, provide runoff infiltrate data for field-scale implementation, and make design recommendations based on soil columns experimentation. Further research can expand in this area to increase the performance of land application systems.

APPENDIX A

MSU Dairy Teaching and Research Facility Management Plan

Properly store all materials to avoid transfer due to environmental processes (wind, rain, etc.). Source Area

1. Sweep silage areas, feed areas, and traffic areas and scrape livestock areas on a minimum weekly basis, increase frequency when rain is forecast or an excessive amount of solids have accumulated
2. Maintain cover over silage and feed, ensure the rainwater is diverted from the area
3. Maintain a smooth and vertical feed out face
4. Avoid vehicle travel through manure piles, feed supplies etc. to avoid tracking into paved areas
5. Avoid spillage around silage, clean up when necessary
6. When collecting manure, ensure the trailer is positioned correctly to avoid any spillage. Properly clean off the conveyor before moving the trailer.
7. Maintain areas around storm drains, avoid solids build-up
8. Keep manure, feed supplies, bedding etc. in their designated places only, avoid temporary storage especially while moving
9. Do not overfill trailers or trucks

Collection System

10. Dry weather leachate should be transferred to a storage facility on a minimum weekly basis, increase frequency if a large volume has accumulated or rain is forecast

11. Scrape accumulated solids from the storage basin after removing dry weather leachate and transport to a storage area

Transfer System

12. Manually operate basin pumps within 72 hours after a storm event to low alarm level
13. Do not reset flow meters, they will be monitored and maintained by researchers
14. Log pump operation, maintenance issues, mowing or any other event concerning the filter strip in the log book located in the pump operation housing

Vegetative Area

15. Do not graze animals in the filter strip area until research has ended
16. Reseed vegetation to maintain the desired cover when necessary
17. Mowing:
    a. Harvest vegetation to promote growth and maintain health and upright growth
    b. Maintain a vegetation height of 6 inches
    c. Harvest vegetation and remove from the filter strip area
    d. Mow direction should be across filter strip, not down the grade (mow north to south)
    e. Do NOT mow filter strips when the soil is saturated
    f. Do NOT use herbicides on or around the filter strip vegetation area
18. Rake rock checks bimonthly to maintain an even surface

19. Inspect the rock checks biannually for solids accumulation, clean/replace if necessary

Contacts

20. Contact Becky Larson prior to pump operation or with any questions/concerns, phone is preferred for pump operation notification, but an email is acceptable if sufficient time is given

APPENDIX B

Table 69: QA/QC

QA/QC	Description	Purpose	Frequency	Analysis Requiring Procedure	Acceptance Criteria	Corrective Action
Duplicate	Take one sample volume and separate into two separate samples which are then prepared and analyzed using identical procedures	Determination of precision in equipment and procedures	Minimum of at least once per use of equipment and once every 10 samples within each use	All Parameters (excluding pH and BOD)	Relative percent difference less than 20%	Improve handling and precision, repeat procedure to ensure acceptance criteria is met
Blanks	Reagent water analyzed as a sample	Detection of error in a zero reading, procedure contamination, or background concentrations	Minimum of at least once per use of equipment and once every 10 samples within each use	All Parameters (excluding pH)	Should produce a near zero or neutral reading, or used as an offset	Ensure proper set-up and procedure, clean equipment, check all reagents and chemicals, find the error in procedure or equipment, reanalyze until criteria met
Standards	Known quantities of sample are analyzed to determine accuracy of equipment	Determination of accuracy of equipment and procedures	Minimum of at least once per use of equipment and once every 10 samples within each use	All Parameters (excluding nitrite)	Relative percent difference less than 20%	Ensure proper set-up and procedure, clean equipment, check all reagents and chemicals, find the error in procedure or equipment, reanalyze until criteria met
Calibrate pH meter	2-point pH calibration with buffers	Ensure accurate and precise readings	Every use	pH	±0.05 pH units for every buffer	Clean probe, retest, replace if acceptance criteria cannot be met

APPENDIX C

USDA-NRCS-MICH Standard

Wastewater Treatment Strip

(Acre) 635

DEFINITION

A treatment component of an agricultural waste management system consisting of a strip or area of herbaceous vegetation.

PURPOSES

The purpose of this practice is to improve water quality by reducing loading of nutrients, organics, pathogens, and other contaminants associated with animal manure and other wastes, and wastewater by treating agricultural wastewater and runoff from livestock holding areas.

CONDITIONS WHERE PRACTICE APPLIES

This practice applies where all the following conditions apply:

1. Wastewater is generated by runoff from areas where livestock are concentrated, runoff and leachate from silage storage areas, runoff and leachate from waste storage facilities for solid manures, runoff from composting areas, or runoff from feed handling areas;
2. Polluted runoff (stormwater and snow melt) may be treated in a wastewater treatment strip;
3. Manure and/or silage solids from the contributing drainage area can be effectively trapped prior to discharge to the wastewater treatment strip; and
4. The area contributing runoff and/or leachate to the wastewater treatment strip is less than 1 acre and confines less than 200 animal units (1 animal unit = 1,000 pounds live weight).

This practice does not apply to filed borders (practice standard 386) or Filter Strips (practice standard 393 A).

This practice standard does NOT apply to the control or treatment of milking center wastewater or any other process washwater.

CRITERIA

General Criteria Applicable to All Purposes

Wastewater treatment strips shall be planned, designed, and installed to meet all federal, state, local and tribal laws and regulations.

The term "silage" as used in this standard include haylage, wheatlage, and any other ensiled livestock feed stored on the farm.

Location and Use. To minimize the potential for contamination of streams, wastewater treatment strips, including the outlet storage area, should be located outside of floodplains. However, if site restrictions require location within a floodplain, the wastewater treatment strip, including the outlet storage area, shall be protected from inundation or damage from a 25-year flood event, or larger if required by laws and regulations.

Wastewater treatment strips shall not be constructed within an area that typically has a seasonal high water within 1 foot (0.3 m) of the surface. Subsurface drainage may be used to lower the seasonal high water table to an acceptable level provided the subsurface drain lines are at least 10 feet (3 m) away from the wastewater treatment strip. All other field tile (subsurface drains) within 10 feet (3 m) of a wastewater treatment strip shall be removed and capped.

Wastewater treatment strips must limit access and control grazing, where appropriate.

Do not use wastewater treatment strips as a travelway for livestock of farm equipment.

<u>Dilution</u> of the runoff to be treated in the wastewater treatment strip is not needed if the contributing drainage area is managed to minimize pollution of the runoff by manures and/or silage. Where suitable management is not provided, the runoff shall be diluted by combining clean runoff with the polluted runoff. The clean runoff contributing area shall be at least equal in area to the polluted runoff contributing drainage area. The combined area for both shall not exceed 1 acre.

<u>Suitable management to minimize pollution</u> of runoff includes the following actions by source area:

- Livestock areas – scraped at least weekly.
- Silage storage areas – have impermeable covers over stored silage, scrape and/or sweep the storage floor and apron at least weekly to collect feed that is spilled, and the silage is kept nearly vertical where it is being removed for feeding.
- Waste storage facilities for solid manure or composting facility – manure or compost is stacked as high as possible (based on design height) and in as small an area as possible; the area where manure or compost has not yet been stacked is scraped and/or swept at least weekly to collect manure that has been spilled.

Collection system. A collection system shall be provided to settle and collect solids, collect dry weather leachate (where applicable), and control the discharge of runoff to the wastewater treatment strip. The collection system shall be designed to facilitate clean-out. Where dry weather silage leachate is anticipated, the collection system shall be designed to minimize deterioration from exposure to the leachate. Collection systems that may erode during an overflow event shall have a freeboard of 6 inches (150 mm). Collection systems that will not erode during an overflow event are not required to have a freeboard.

Refer to Waste Storage Facility (313) for collection system structural design criteria. Structures shall be designed to withstand the anticipated static and dynamic loading. Settling facilities shall be installed above the water table. When curbs are needed in conjunction with collection systems, they shall be constructed of either concrete or pressure-treated wood and shall be adequately anchored. Curbs shall be of sufficient height to ensure flow control up to the design discharge. Refer to the Manure Transfer (634) practice standard for safety criteria and for design criteria for pipes associated with the collection system. Livestock shall be excluded from the collection system, as appropriate, to prevent damage and to avoid harm to the animals.

The minimum collection system design volume shall be the volume of runoff from the 25-year, 24-hour rainfall event on the contributing drainage area less the outflow volume at the design discharge over a 24 hour period. The design outflow discharge from the collection system to the wastewater treatment strip shall not exceed the peak discharge from a 2-year, 24-hour rainfall event on the contributing drainage area.

The collection system design volume shall include a solids and dry weather leachate storage area with a minimum capacity equivalent to the volume of 0.15 inches 94 mm) of runoff from the contributing drainage area. The liquid in the solids and dry weather leachate storage area may not be directed to the wastewater treatment strip during a runoff event. To facilitate dewatering of accumulated solids, the liquid in the solids and dry weather leachate storage area may be directed to the wastewater treatment strip after the runoff event has ended. Stored dry weather leachate may not be directed to the wastewater treatment strip. The solids and dry weather leachate storage area shall be emptied within 72 hours of a runoff event or weekly when dry weather leachate has accumulated. The collected solids and dry weather leachate shall be transferred to a storage facility or utilized in accordance with the Nutrient Management practice standard (590).

<u>Design Discharge and Dimensions.</u> Minimum wastewater treatment strip dimensions shall be based on the peak inflow rate resulting from a 2-year, 24-hour rainfall on the contributing drainage area. A level spreader, grated pipe, sprinklers, or other facilities shall be provided across the upstream end of the wastewater treatment strip to establish sheet flow.

The wastewater treatment strip shall not allow discharge to surface waters for up to the 25-year, 24-hour storm event. The wastewater treatment strip shall prevent lateral discharge to surface waters as the water passes along the length of the wastewater treatment strip up to the 25-year, 24-hour storm event. This may be accomplished by natural or artificial boundaries.

Use the equation below to compute the design peak discharge from the contributing drainage area. (Tabular hydrograph method maximum unit peak discharge of 1,000 csm (cfs/mi^2/in runoff) for Type II storms in Urban Hydrology for Small watersheds, Technical Release No. 55, NRCS, June 1986.)

Peak Discharge Qp (cfs):

$$Qp = R \times A \times 0.000036$$

R = Runoff depth (in.)

Compute using a curve number of 90 for unpaved areas and 98 for paved areas or roof areas

A = Contributing drainage area (sq. ft.)

The wastewater treatment strip shall be a relatively uniform grass area of grassed channel. Wastewater treatment strips shall be designed for natural or constructed slopes of 0.3 to 6 percent. The first 100 feet at the upstream end should not be flatter than 1 percent. Where constructed slopes are required, salvage existing topsoil and spread at final grade.

Grass are (overland) wastewater treatment strips shall be generally on the contour and sufficiently wide to pass the peak flow at a depth of 0.5 inches (13 mm) or less. Maximum flow width (perpendicular to the direction of flow) shall be 100 feet (30 m). Flow length parallel to the direction of flow) shall be sufficient to provide at least 15 minutes of flow-through time. Flow-through time equals the wastewater treatment strip length divided by the average velocity. Average flow velocity shall be determined using Manning's equation with an "n" value of 0.3.

To minimize the development of flow concentrations which will short-circuit the sheet flow need to maintain the effectiveness of the grass wastewater treatment channel, rock checks will be installed at 100 foot intervals along the length of the channel.

A rock check is a shallow trench filled with MDOT 22A or 23A coarse aggregate. The trench should be 1 to 1.5 feet (0.3 to 0.5 m) deep, extend 2 to 4 feet (0.6 to 1.2 m) in the direction of flow, and extend the full width of the channel up to the design depth. The top of the stone in the trench should be flush with the bottom of the channel.

Preventing discharge to surface water. The outlet of the wastewater treatment strip shall be designed to prevent discharge to surface water. To accomplish this, the wastewater treatment strip must outlet into an outlet storage area or must maintain a minimum outlet setback distance to surface water.

Outlet storage areas shall have the capacity to contain the entire runoff volume from the 25-year, 24-hour storm from the contributing drainage area plus the wastewater treatment strip area. The outlet storage area capacity may be reduced by the volume of runoff captured by the solids and leachate collection system. The outlet storage area may be a natural depression area, or a constructed depression area. The outlet storage area shall not be a wetland. The outlet storage area shall be able to infiltrate the collected water within 72 hours based on the permeability of the most restrictive layer in the root zone regardless of its thickness. Earth berms used for constructed depressional areas shall be less than 3 feet (1 m) in height, have a freeboard of at least 0.3 (0.1 m) feet above the design high water elevation, have a top width of at least 4 feet (1.2 m), and side slopes of at least 3:1 or flatter.

Minimum outlet setback distance is 150 feet (45 m) measured along the flow path from the outlet of the wastewater treatment strip to the surface water. Surface water may be a stream, surface drain, surface inlet, road ditch or other conveyance. The slop on any portion of the outlet setback distance may not exceed 12 percent. The flow path must be either established permanent vegetation (such as hayland, pasturelands, grassland, or vegetated buffer) or cropland.

Establishment of vegetation. Runoff shall be diverted away from the wastewater treatment strip channel until the vegetation is well established. A minimum height of 4 inches (0.1 m) and 90 percent ground cover is desirable. Select one of the seed mixtures in table 1, depending on soil type and drainage conditions. Limed and fertilized in accordance with the Critical Area Planting (342) practice standard.

TABLE 1 – Vegetative Mixtures for Wastewater Treatment Strips

Soils – Well and moderately well drained coarse to fine textured soils.

Species or Seeding Mixtures	Seeding Rate (lbs/acre)	Established Stem Density (stems/sq ft)
Reed Canarygrass	10	50
Smooth Brome	20	50
Tall Fescue	20*	60
Smooth Brome Tall Fescue*	12 10	60
Orchardgrass Timothy Red Clover Alfalfa	5 5 6 6	70

Soils – Somewhat poorly drained or poorly drained soils without artificial drainage

Species or Seeding Mixtures	Seeding Rate (lbs/acre)	Established Stem Density (stems/sq ft)
Garrison Creeping Foxtail	10	70
Reed Canarygrass	10	50
Tall Fescue*	20	60
Orchardgrass Redtop Alsike Clover White Dutch Clover	5 2 3 2	70

* Do not use Endophyte fungus susceptible Tall fescue varieties if area is planned for grazing or forage.

Use vegetation adapted to the site that will accomplish the desired purpose. Preference shall be given to native species in order to reduce the introduction of invasive plant species; provide management of existing invasive species; and minimize the economic, ecological, and human health impacts that invasive species may cause. If native plant materials are not adaptive or proven ineffective for the planned use, then non0native species may be used. Refer to the Field Office Technical Guide, Section II, Invasive Plant Species for plant materials identified as invasive species.

CONSIDERATIONS

Consider the potential effects of installation and operation of wastewater treatment strips on the cultural, archeological, historic and economic resources.

Consider the ability of the landowner/operator to manage and operate the wastewater treatment strip in accordance with the operation and maintenance plan.

APPENDIX D

Key: MSU Dairy: F=filter strip (1 or 2), MH=man hole (numbered 1 through 3 down the slope), RC=Rock Check (numbers following indicate the rock check numbered from top to bottom and the letters from left to right of the filter strip looking up the slope), SB=storage basin (1 or 2). Small MI Dairy: Basin=settling basin, BIO=bioretention basin, RC=rock check, T1=subsurface samples at 1.5ft (A=3 ft down slope, B=13 ft down slope), T2=subsurface samples at 2.5 ft.

Table 70: MSU Dairy Alkalinity Data

Alkalinity (mg/L)	09/17/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	304	330	250	300	192	155	140	100	350
F1MH2	320	330	260	320		155	145	170	390
F1MH3			280	315				150	325
F1RC1A	140	320	235	315					
F1RC1B	160	370	240	315			125		
F1RC1C	116	330	250	260	196			135	315
F1RC2A	196		250	290				90	
F1RC2B	152		235	295					380
F1RC2C	188		245	305					
F1RC3A			250	310					290
F1RC3B			245	310				125	
F1RC3C			245	315					
F2MH1	380	340	195	0	798		535	415	345
F2MH2	396	290	675	0			490	635	335
F2MH3		360	245	0				360	305
F2RC1A		330	190	0			490		250
F2RC1B		380	190	0				155	
F2RC1C		350	200	0	940				
F2RC2A		320	220	0				150	
F2RC2B			180	0					135
F2RC2C			195	0					
F2RC3A		330	245	0					
F2RC3B		340	260	0					
F2RC3C		320	250	0				175	230
SB1	124	320	285	315	193	130	140	80	240
SB2	260	330	35	0	650	200	350	55	160

Table 71: MSU Dairy Ammonia Data

Ammonia (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	26.75	3.75	7.75	0	10.5		5	0.5	1.5	
F1MH2	29.25	6.25	12.75	1.5	14.5		4	54.5	1	24.5
F1MH3	24.5			5	13.5				1	16.75
F1RC1A		10	16.75	6	21.5					
F1RC1B		8.75	16.25	12.5	20			0.5		
F1RC1C		8.75	21	6	17.5	10.5			2.5	30.25
F1RC2A		10.5		4.5	21.5				3	
F1RC2B		10.5		6.5	21					29
F1RC2C		11.5		6.5	20					
F1RC3A				7	19.5					19.25
F1RC3B				8.5	19.5				1.5	
F1RC3C				13.5	21					
F2MH1	14.25	28	64	29.5	67.5	105		75.5	19.5	7.5
F2MH2	13	31.5	59.25	15.5	46.5			68	3	4.75
F2MH3	10.5		51	27.5	61				16.5	7.75
F2RC1A				47	78.5			72.5		13.5
F2RC1B				40.5	68				40.5	
F2RC1C				41.5	83	160				
F2RC2A				39	81.5				34.5	
F2RC2B				36	81.5					12.25
F2RC2C				40	82					
F2RC3A				36	72					
F2RC3B				33	76.5					
F2RC3C				37	77				32.5	10.25
SB1	23.75	10.75		10	19.5	9	5.5	0	3.5	16.75
SB2	18.25	58.5		31	50	179	72.5	4	15	8

Table 72: MSU Dairy COD Data

COD (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	555	190	510	544	600	246	220	222	136	272
F1MH2	760	220	350	432	616		190	308	111	222
F1MH3	675			572	628				155	310
F1RC1A		350	160	668	780					
F1RC1B		250	190	704	780			271		
F1RC1C		250	280	844	672	334			200	400
F1RC2A		175		660	792				161	322
F1RC2B		300		564	832					
F1RC2C		350		588	732					
F1RC3A				532	808					
F1RC3B				544	732				124	248
F1RC3C				616	744					
F2MH1	1015	1100	1910	3716	10384	7550		6830	1440	2880
F2MH2	940	1230	1750	2556	7472			8110		
F2MH3	865		1210	3452	9704				1520	3040
F2RC1A			1660	3056	10368			6930		
F2RC1B			2250	2992	9432				2540	5080
F2RC1C			2060	3948	11576	13440				
F2RC2A			1180	3836	11104				2410	4820
F2RC2B				3736	11632					
F2RC2C				3776	11336					
F2RC3A			1440	3756	10864					
F2RC3B			1700	3548	11296					
F2RC3C			1590	3588	11616				2190	4380
SB1	705	400	700	556	788	354	248	281	136	272
SB2	1130	2700	2225	3040	7232	13260	4920	6440	1140	2280

Table 73: MSU Dairy Soluble COD Data

Soluble COD (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	650	210	190	300	348	98	67	105	71	306
F1MH2	775	200	325	272	484		68	106	60	329
F1MH3	815			312	292				76	286
F1RC1A		425	265	380	488					
F1RC1B		350	170	352	460			139		
F1RC1C		350	300	352	460	106			82	343
F1RC2A		275		284	580				91	
F1RC2B		450		332	476					342
F1RC2C		325		308	464					
F1RC3A				252	488					326
F1RC3B				292	484				68	
F1RC3C				208	508					
F2MH1	1025	970	2210	3676	10950	7280		6400	1810	417
F2MH2	900	1130	1915	2504	7840			7790	320	367
F2MH3	935		2060	3420	10030				1930	378
F2RC1A			2005	3392	10440			6620		567
F2RC1B			3565	3240	9330				3110	
F2RC1C			2340	3716	11950	12630				
F2RC2A			1610	3716	11840				2990	
F2RC2B				3284	11680					549
F2RC2C				3672	11540					
F2RC3A			1910	3476	11500					
F2RC3B			2120	3420	11870					
F2RC3C			2560	3524	11730				2670	518
SB1	805	350	475	380	464	120	69	113	88	198
SB2	1215	2800	2235	2740	7450	12400	4480	6150	1660	330

Table 74: MSU Dairy TS Data

TS (mg/L)	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009
F1MH1	704	1024	830	744	984	552
F1MH2	852	8136	5380	680	1140	780
F1MH3	804	1048				700
F1RC1A	1064	1376				
F1RC1B	1436	1268			860	
F1RC1C	2008	1408	1225			552
F1RC2A	1740	1084				492
F1RC2B	1012	1200				
F1RC2C	1092	1116				
F1RC3A	824	1212				
F1RC3B	1096	1236				488
F1RC3C	1356	1216				
F2MH1	2608	7636			4956	1704
F2MH2	1932	5848			5668	952
F2MH3	2564	6916				1652
F2RC1A	3204	7440			5020	
F2RC1B	2536	6780				2272
F2RC1C	2572	8128	7125			
F2RC2A	2900	8272				2128
F2RC2B	2760	8920				
F2RC2C	2336	8568				
F2RC3A	3416	8196				
F2RC3B	2820	1044				
F2RC3C	2832	8600				2096
SB1	816	1268	1005	588	876	316
SB2	1960	4984	7070	3072	4128	1212

Table 75: MSU Dairy TSS Data

TSS (mg/L)	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	4/1/2010
F1MH1		120	180	100	140	110
F1MH2	280	100		180	120	130
F1MH3		80				100
F1RC1A	500	200				
F1RC1B	1000	200			60	
F1RC1C	1100	260	1380			120
F1RC2A	1520	120				
F1RC2B		200				230
F1RC2C	600	460				
F1RC3A	340	140				130
F1RC3B	280	360				
F1RC3C	480	240				
F2MH1	260	200	400		220	110
F2MH2		80			250	50
F2MH3		120				90
F2RC1A	420	280			400	170
F2RC1B	300	300				
F2RC1C	40	280	200			
F2RC2A	140	220				
F2RC2B	580	660				100
F2RC2C	80	620				
F2RC3A	1400	220				
F2RC3B	520	280				
F2RC3C		440				190
SB1	360	120	160	100	100	60
SB2	300	160	780	100	270	140

Table 76: MSU Dairy VS Data

VS (mg/L)	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009
F1MH1	256	384	215	180	368	396
F1MH2	404	5548	3140	152	488	268
F1MH3	356					344
F1RC1A	424	660				
F1RC1B	432	532			336	
F1RC1C	556	564	245			280
F1RC2A	524	364				200
F1RC2B	356	588				
F1RC2C	344	456				
F1RC3A	396	540				
F1RC3B	400	564				232
F1RC3C	432	516				
F2MH1	1716	5172			3168	912
F2MH2	1064	3828			3496	424
F2MH3	1620	4896				944
F2RC1A	1804	4788			2976	
F2RC1B	1484	4440				1568
F2RC1C	1572	5272	4305			
F2RC2A	1760	5712				1380
F2RC2B	2348	5620				
F2RC2C	1264	5616				
F2RC3A	1680	5744				
F2RC3B	1668	488				
F2RC3C	1740	5740				1304
SB1	364	544	270	96	448	200
SB2	1364	3544	4195	1900	2624	896

Table 77: MSU VSS Alkalinity Data

VSS (mg/L)	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	4/1/2010
F1MH1		120	120	60	120	50
F1MH2	220	40		80	60	60
F1MH3		60				70
F1RC1A	280	140				
F1RC1B	220	140			30	
F1RC1C	200	160	360			70
F1RC2A	360	100				
F1RC2B		80				110
F1RC2C	140	140				
F1RC3A	240	80				90
F1RC3B	220	100				
F1RC3C	140	120				
F2MH1	120	160	360		200	80
F2MH2		0			210	50
F2MH3		100				0
F2RC1A	140	180			230	160
F2RC1B	160	240				
F2RC1C	0	260	140			
F2RC2A	120	160				
F2RC2B	360	300				70
F2RC2C	0	160				
F2RC3A	440	180				
F2RC3B	200	220				
F2RC3C		320				130
SB1	120	100	120	0	80	10
SB2	80	140	600	60	230	60

Table 78: MSU Dairy Nitrite Data

Nitrite (mg/L-N)	09/17/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	0.36	0.022	0	0.372	0.044	0.032	0.272	0.026
F1MH2	0.229	0.044	0.02		0.052	0.07	0.034	0.018
F1MH3		0.024	0.026				0.07	0.04
F1RC1A	0.013	0.04	0.008					
F1RC1B	0.008	0.022	0.045			0.028		
F1RC1C	0.006	0.14	0.008	0.104			0.018	0.026
F1RC2A	0.002	0.042	0.034				0.02	
F1RC2B	0.524	0.036	0.008					0.034
F1RC2C	1.86	0.03	0.008					
F1RC3A		0.03	0.02					0.042
F1RC3B		0.026	0.008				0.022	
F1RC3C		0.022	0.018					
F2MH1	0.005		0.328	3.6		0.662	2.224	0.012
F2MH2	0	0.052	0.094			0.444	0.008	6.7
F2MH3		0.046	0.08				1.73	0.012
F2RC1A		0.034	0.576			0.724		0.012
F2RC1B		0.016	1.89				0.018	
F2RC1C		0.04	0.016	0				
F2RC2A		0.032	0				0.05	
F2RC2B		0.03	0					0.012
F2RC2C		0.002	0					
F2RC3A		0.032	0					
F2RC3B		0.03	0					
F2RC3C		0.042	0.038				0.014	0.02
SB1	0.007	0.026	0.01	0	0.012	0.018	0.28	0.068
SB2	0	0.03	0.006	0	0	0	0.138	0

Table 79: MSU Dairy Nitrate Data

Nitrate (mg/L-N)	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	7.6	16	3.8	23.0	13	1.4	11.8
F1MH2	10.4	8		2.7	30	2.2	27
F1MH3	8.6	6.6				6	3.8
F1RC1A	9.4	15.8					
F1RC1B	7.6	17.2			12.6		
F1RC1C	6.4	16.4	11.1			2.6	6.6
F1RC2A	15.2	14.8				9.8	
F1RC2B	10	20.6					5.6
F1RC2C	9.4	16.8					
F1RC3A	6.2	15.8					11.8
F1RC3B	10.2	21.8				3	
F1RC3C	13.6	21.4					
F2MH1	6	13.4	49.2		176	17.4	17.4
F2MH2	10.4	12.4			148	22	41.2
F2MH3	8.2	10.2				22.2	0
F2RC1A	4.8	23			54		2.6
F2RC1B	15.4	24				2	
F2RC1C	21.4	24.2	116.0				
F2RC2A	9.6	18.2				20.8	
F2RC2B	6.8	20.2					4
F2RC2C	10.8	15.2					
F2RC3A	1.6	22.4					
F2RC3B	10.4	14.4					
F2RC3C	14	44				26.4	6.4
SB1	7.6	9.6	0.0	18.0	0	8.4	36.8
SB2	6.6	32.8	9.6	0.0	39	7.2	9.2

Table 80: MSU Dairy pH Data

pH	09/09/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	10/23/2009	4/1/2010
F1MH1	6.87	6.7	6.63	6.62	6.54	6.61	6.62	6.45	6.78
F1MH2	6.87	6.94	6.79	6.7		6.58	6.55	6.55	6.74
F1MH3	6.73	6.95	6.56	6.45				6.41	6.83
F1RC1A		6.96	6.88	6.74					
F1RC1B		6.99	6.2	6.74			6.67		
F1RC1C			6.82	6.75	6.88			6.7	6.92
F1RC2A			6.98	6.94				6.59	
F1RC2B			6.91	6.75					7.19
F1RC2C			6.82	6.9					
F1RC3A			6.95	6.8					7.1
F1RC3B			6.98	6.87				6.49	
F1RC3C			7	6.81					
F2MH1	6.53	6.03	5.2	4.3	5.82		5.63	6.33	6.67
F2MH2	6.53	6.12	5.86	4.58			5.61	6.75	6.71
F2MH3	6.75	6.45	5.29	4.45				6.41	6.95
F2RC1A		6.41	5.08	4.48			5.61		6.77
F2RC1B		6.28	5.05	4.65				5.21	
F2RC1C		6.4	5.13	4.36	5.39				
F2RC2A		6.66	5.12	4.28				5.41	
F2RC2B			5.13	4.33					6.95
F2RC2C			5.2	4.36					
F2RC3A		6.64	5.23	4.36					
F2RC3B		6.54	5.38	4.36					
F2RC3C		6.5	5.29	4.36				6.49	7.11
SB1	6.8	6.78	6.65	6.74	6.73	6.92	6.66	6.3	7.07
SB2	6.2	6.11	4.88	4.27	5.48	5.45	5.46	4.75	6.79

Table 81: MSU Dairy Phosphorus Data

Phosphorus (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	06/09/09	8/19/2009	8/31/2009	10/8/2009	4/1/2010
F1MH1	12.5	9	7.5	2.2	1.2	9.3	11.2	20	20
F1MH2	12	34	9	1.3	1.1		5.5	22.5	22.75
F1MH3	10.5			1.9	1.1				13.5
F1RC1A		7.5	5	6.0	9.0				
F1RC1B		0	6.5	5.5	8.5			20.5	
F1RC1C		10.5	9	11.6	9.0	12.4			20.75
F1RC2A		10		4.6	8.0				
F1RC2B		10.5		7.5	8.5				15.5
F1RC2C		12.5		5.0	8.0				
F1RC3A				3.3	7.2				24.25
F1RC3B				3.0	7.7				
F1RC3C				5.0	7.8				
F2MH1	12.5	29	18	32.5	23.3			46	21.5
F2MH2	10	32	15.5	13.4	14.2			57	18.25
F2MH3	9.5		12	27.9	19.5				20.5
F2RC1A				37.4				57	31.25
F2RC1B				35.6					
F2RC1C				34.6					
F2RC2A				30.9					
F2RC2B				30.1					26
F2RC2C				33.3					
F2RC3A				31.0	25.9				
F2RC3B				27.7	28.2				
F2RC3C				17.4	27.9				27.25
SB1	10.5	9.5		2.8	1.6	12.9	10.7	20	17.5
SB2	20.5	22.5		23.1	16.6		40.7	20	24.3

Table 82: MSU Dairy Mn Data

Mn (ug/L)	09/09/08	09/17/08	10/02/08	5/14/2009	06/09/09	08/19/09	08/31/09	4/1/2010
F1MH1	1900	1200	280	130	310	52	62	280
F1MH2	980	1300	450	140	400		31	190
F1MH3	5800			180	370			130
F1RC1A		330	1100	580	250			
F1RC1B		300	1600	1500	190			
F1RC1C		330	430	1900	140	59		320
F1RC2A		540		1100	280			
F1RC2B		320		270	1100			240
F1RC2C		420		190	290			
F1RC3A				620	310			79
F1RC3B				1100	360			
F1RC3C				810	230			
F2MH1	1300	3600	860	1300	4100			270
F2MH2	1700	4700	840	2400	6900	4500		130
F2MH3	1400		570	1500	4200			69
F2RC1A			1500	1600	1700			130
F2RC1B			1300	1500	1500			
F2RC1C			970	1500	1700	2100		
F2RC2A			590	1800	2200			
F2RC2B				2000	2700			31
F2RC2C				2000	4100			
F2RC3A			870	3200	3100			
F2RC3B			1200	2600	3500			
F2RC3C			970	2400	4300			25
SB1	120	220	330	190	230	270	140	120
SB2	200	470	540	400	720	1400	580	140

Table 83: MSU Dairy Fe Data

Fe (ug/L)	09/09/08	09/17/08	10/02/08	5/14/2009	06/09/09	08/19/09	08/31/09	4/1/2010
F1MH1	2100	1300	2100	1400	1300	910	880	1200
F1MH2	1600	1300	2200	1000	1200		640	1600
F1MH3	4200			1400	1200			860
F1RC1A		2600	7300	17000	4800			
F1RC1B		1800	9400	75000	2100			
F1RC1C		2500	4800	100000	3200	1200		900
F1RC2A		4600		48000	2000			
F1RC2B		2800		6900	38000			820
F1RC2C		3200		3600	1500			
F1RC3A				16000	1400			810
F1RC3B				34000	7300			
F1RC3C				23000	2900			
F2MH1	1200	2900	2900	4200	14000	13000		560
F2MH2	1100	3900	2100	1800	7000			1700
F2MH3	720		1200	3400	10000			320
F2RC1A			14000	13000	12000			590
F2RC1B			17000	16000	6900			
F2RC1C			4700	5700	16000	13000		
F2RC2A			1600	14000	19000			
F2RC2B				12000	18000			470
F2RC2C				10000	91000			
F2RC3A			3600	66000	22000			
F2RC3B			4800	29000	17000			
F2RC3C			2300	10000	30000			370
SB1	1100	1500	2200	2200	1900	2600	1100	740
SB2	1300	6600	5700	3200	12000	18000	6000	940

Table 84: MSU Dairy TOC Data

TOC (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	6/9/2009	08/19/09	08/31/09	4/1/2010
SB1	240	100	190	190	210	56	44	110
SB2	420	980	780	940	2800	3600	1400	130
F1MH1	200	55	97	160	160	46	37	140
F1MH2	270	64	130	140	190		41	140
F1MH3	230			170	170			120
F1RC1A		85	120	200	260			
F1RC1B		73	89	190	220			
F1RC1C		84	140	180	210	55		140
F1RC2A		65		160	240			
F1RC2B		76		160	240			140
F1RC2C		66		160	230			
F1RC3A				160	240			140
F1RC3B				150	230			
F1RC3C				150	230			
F2MH1	400	370	710	1200	3900			160
F2MH2	330	410	660	850	2800			130
F2MH3	320		620	1200	3500			130
F2RC1A			630	1200	3600			190
F2RC1B			770	1300	3300			
F2RC1C			730	1300	4100	3900		
F2RC2A			540	1300	4100			
F2RC2B				1300	4100			180
F2RC2C				1200	4100			
F2RC3A			620	1200	4200			
F2RC3B			660		4200			
F2RC3C			690	1200	4100			170

Table 85: MSU Dairy Conductance Data

Conductance (umhos/cm)	09/09/08	09/17/08	10/02/08	05/14/09	06/09/09	08/19/09	08/31/09	4/1/2010
SB1	1032	560	1492	973	1357	1242	836	1239
SB2	1030	1634	1944	1290	2700	4950	2211	536
F1MH1	1573	898	1290	998	1337	1199	942	2553
F1MH2	1509		1410	996	1316		950	2884
F1MH3	1626			1054	1129			2446
F1RC1A		561	1373	1036	1601			
F1RC1B		578	1382	1022	1535			
F1RC1C		563	1324	1059	1555	1224		2538
F1RC2A		582		1021	1477			
F1RC2B		606		1006	1484			2434
F1RC2C		610		1005	1450			
F1RC3A				978	1431			2340
F1RC3B				992	1448			
F1RC3C				1007	1456			
F2MH1	1227	1379	1815	1995	3860	3790		901
F2MH2	1206	1391	1744	1639	3230			908
F2MH3	1127		1576	1942	3720			780
F2RC1A			1925	1972	4050			673
F2RC1B			1953	1964	3790			
F2RC1C			1852	1990	4060	5070		
F2RC2A			1857	1981	4000			
F2RC2B					4060			671
F2RC2C				1977	4140			
F2RC3A			1836	2005	4060			
F2RC3B			1870	1927	4280			
F2RC3C			1908	1940	4280			701

Table 86: MSU Dairy Cl Data

Cl (mg/L)	09/09/08	09/17/08	10/02/08	05/14/09	6/9/2009	08/19/09	08/31/09	4/1/2010
F1MH1	118	53	143	123	225	236	172	548
F1MH2	120		162	121	187		176	654
F1MH3	135			134	125			548
F1RC1A		48	162	134	251			
F1RC1B		47	156	132	242			
F1RC1C		45	163	136	265	253		550
F1RC2A		31		135	216			
F1RC2B		41		134	216			549
F1RC2C		38		136	218			
F1RC3A				141	210			518
F1RC3B				139	213			
F1RC3C				139	215			
F2MH1	20	18	24	39	112	83		26
F2MH2	17	19	27	30	102			23
F2MH3	18		22	36	108			23
F2RC1A			23	42	140			23
F2RC1B			23	41	117			
F2RC1C			22	42	135	113		
F2RC2A			22	42	124			
F2RC2B					124			20
F2RC2C				42	133			
F2RC3A			22	41	133			
F2RC3B			22	41	130			
F2RC3C			25	44	132			19
SB1	13	38	191	114	187	253	161	195
SB2	18	20	27	26	76	112	44	17

Table 87: MSU Dairy Arsenic Data

Arsenic (µg/L)	09/09/08	09/17/08	10/02/08	05/14/09	6/9/2009	08/19/09	08/31/09	4/1/2010
F1MH1	5	3.1	3.4	2.8	2.5	2	1.8	1.5
F1MH2	3.4	3.4	4.2	2.8	2.4		2	2
F1MH3	5.7			2.9	2.2			1.6
F1RC1A		6.7	12	7.4	2.5			
F1RC1B		6.2	14	20	1.2			
F1RC1C		5.5	5.5	26	2.5	2		1.7
F1RC2A		11		15	1.1			
F1RC2B		9.8		5.5	10			1.3
F1RC2C		11		3.9	1.2			
F1RC3A				7.5	1			1.6
F1RC3B				14	3.8			
F1RC3C				7.4	2.1			
F2MH1	3.1	6.4	4.5	6.8	14	20		2.6
F2MH2	3.6	7.4	4.3	7.3	13			2.9
F2MH3	3.9		5.1	7.5	14			2.6
F2RC1A			9.8	8.4	9.5			1.8
F2RC1B			6.8	8.6	8.2			
F2RC1C			5.1	4.8	9.8	13		
F2RC2A			11	9	10			
F2RC2B				8.7	13			2.7
F2RC2C				10	40			
F2RC3A			9.8	24	13			
F2RC3B			13	14	14			
F2RC3C			7.2	11	22			2.8
SB1	1.9	1	2.4	2.5	0	1.8	1.4	1.2
SB2	1.9	2.8	3.9	3.1	4.8	8.3	4	1.2

Table 88: Small MI Dairy TOC Data

TOC (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	1400	960	1300	1200	820	490
BIO	840	680	490	330	760	320
RC1	400	430	520	380	320	320
T1A	190	170	130	140	150	130
T1B		110	210	170	110	110
T2A	86	77	110	110	210	240
T2B		74	330	270	150	130

Table 89: Small MI Dairy Mn Data

Mn (ug/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	2100	1500	1500	1400	1100	600
BIO	8100	4400	2600	2900	3300	1800
RC1	8200	2200	3500	3100	2400	1000
T1A	8500	4000	2100	1400	2800	990
T1B		1100	2800	4500	7700	7900
T2A	1500	1500	4000	6800	2600	1800
T2B		1600	6600	3500	2800	2900

Table 90: Small MI Dairy Fe Data

Fe (ug/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	16000	9800	6400	6600	3700	2400
BIO	130000	50000	23000	32000	42000	20000
RC1	210000	14000	24000	18000	17000	6300
T1A	88000	48000	22000	5100	6100	5300
T1B		26000	12000	7100	16000	26000
T2A	18000	1000	4500	9800	9400	8800
T2B		63000	37000	10000	4800	5400

Table 91: Small MI Dairy Conductance Data

Conductance (umhos/cm)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	5540	4580	5920	5330	4872	2990
BIO	4780	4420	3150	2795		2618
RC1	2698	2397	3210	3000		2479
T1A	1836	1921	1867	2076	2202	2072
T1B		957	1835	1802	2062	2083
T2A	1847	1768	1847	2021	2142	1848
T2B		1362	2144	2296	1638	1208

Table 92: Small MI Dairy Cl Data

Cl (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	394	319	387	282	312.2	179
BIO	312	328	208	180		162
RC1	141	134	193	169		144
T1A	106	122	142	155	115	122
T1B		62	144	140	134	137
T2A	119	117	125	131	108	102
T2B		92	129	154	90	67

Table 93: Small MI Dairy As Data

As (ug/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	30	28	22	24	24	33
BIO	70	39	34	30	32	25
RC1	79	30	32	31	23	17
T1A	41	28	14	12	18	11
T1B		16	12	15	13	15
T2A	8.3	3.1	0	11	17	20
T2B		25	27	22	14	15

Table 94: Small MI Dairy BOD_5 Data

BOD_5 (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/24/2010
BASIN	1161	1542	1434	1434	981.5714286
BIO		672	314	314	
RC1	277	404	455	455	
T1A		146			
T1B			165.428571	165	
T2A	243				129
T2B			344	344	

Table 95: Small MI Dairy Alkalinity Data

Alkalinity (mg/L as $CaCO_3$)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	1855	2080	2500	3280	2440	1140
BIO	2105	2160	3500	1720	2440	1320
RC1	1190	1260	3440	1600	1540	1240
T1A	795	1060	920	1020	1320	1220
T1B		720	920	1000	1280	1140
T2A	620	940	980	1040	1240	980
T2B		780	1040	1100	980	640

Table 96: Small MI Dairy COD Data

COD (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	6480	4790	5340	5360	3030	3010
BIO	7120	13890	1670	1010	2120	2320
RC1	7160	1780	1790	1470	810	1270
T1A	2130	722	522	537	410	980
T1B		486	1006	385	450	270
T2A	381	288	398	754	930	370
T2B		642	1595	1070	430	580

Table 97: Small MI Dairy TKN Data

TKN (mg/L- N)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	350	270	290	270	200	130
BIO	370	290	130	110	240	91
RC1	210	110	140	120	100	86
T1A	54	37	25	23	28	17
T1B		30	43	39	27	23
T2A	23	19	24	24	63	61
T2B		28	87	75	28	24

Table 98: Small MI Dairy Ammonia Data

Ammonia (mg/L-N)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	146	74	100	169	81.5	51
BIO	154	37	61	74	81.5	52
RC1	59	49	66	79	56.5	33.5
T1A	25	14	23	5	12.5	10
T1B		12.5	4.5	15.5	15.5	15
T2A	11	33.5	7	15.5	39	30
T2B		30.5	38.5	44	10.5	9.5

Table 99: Small MI Dairy Nitrate Data

Nitrate (mg/L-N)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	25	30			90	40
BIO		105	30	10	40	7.5
RC1		75		45	20	37.5
T1A	40	33	0	1.1	35	17.5
T1B		20	50	45	20	17.5
T2A		100	45	25	5	12.5
T2B		0	3.3	1.3	35	17.5

Table 100: Small MI Dairy Nitrite Data

Nitrite (mg/L-N)	5/18/2010	5/24/2010	6/4/2010
BASIN	0.3	0.425	0.06
BIO	0.1	0.003	0
RC1	0.1	0	0
T1A		0.1	0.12
T1B		0.8	0.04
T2A		0.275	0.18
T2B		0.001	0.06

Table 101: Small MI Dairy pH Data

pH	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	8.07	8.15	7.91	8.05	7.66	8.96
BIO	7.82	7.85	7.06	6.88	7.69	7.19
RC1	7.45	7.37	7.27	7.35	7.33	7.64
T1A		7.2	7.51	7.44	7.37	7.94
T1B		6.84	7.63	6.79	7.44	7.4
T2A		7.64	7.5	6.94	7.4	7.17
T2B		7.46	7.55	7.38	7.1	6.87

Table 102: Small MI Dairy Phosphorus Data

Phosphorus (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	96	121	74	74	128	73
BIO	154.5	317	9	16	143	44
RC1	45	95	9	19	59	65
T1A	20	43.5	0	0	20	21
T1B		37	0	0	7.5	28.5
T2A	69	17.5	0	0	29.5	38.5
T2B		45.5	0	0	20.5	27.5

Table 103: Small MI Dairy Soluble COD

Soluble COD (mg/L)	6/10/2010	6/16/2010	6/24/2010
BASIN		1686	1420
BIO	716	1550	930
RC1	729	923	1053
T1A	360	488	355
T1B	590	320	322
T2A	397	745	768
T2B	303	440	356

Table 104: Small MI Dairy TS Data

TS (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	5788	6204		9940	4584	3028
BIO	10514	14568		7892	8936	2212
RC1	12714	2476		4712	2460	2416
T1A	3756	3120		4224	2124	1532
T1B		1604	2192	4376	1584	1504
T2A	6330	1136	2756	3312	1900	1600
T2B		2444	1336	2812	1288	912

Table 105: Small MI Dairy VS Data

VS (mg/L)	5/18/2010	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	2360	3372		3552	1856	1312
BIO	4812	6252		5640	4032	816
RC1	2852	1180		1732	1080	840
T1A	362	588		1744	824	524
T1B		420	1068	2656	796	340
T2A	916	272	892	724	888	516
T2B		740	320	928	604	228

Table 106: Small MI Dairy TSS Data

TSS (mg/L)	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	2520		1880	540	720
BIO	16967		440	7440	220
RC1	910		800	300	260
T1A	2580		440	127	
T1B	880	5020	487	187	40
T2A	84	3700	187	133	107
T2B	2180	148	407	113	13

Table 107: Small MI Dairy VSS Data

VSS (mg/L)	5/24/2010	6/4/2010	6/10/2010	6/16/2010	6/24/2010
BASIN	1920		1240	520	400
BIO	7633		180	3680	0
RC1	570		640	220	180
T1A	293		187	60	
T1B	20	1060	347	73	
T2A	24	780	93	107	67
T2B	253	48	360	73	13

APPENDIX E

Statistical analysis was conducted by first defining the statistical models, then validating assumptions of normality of the residuals and the homogeneity of the variances, looking at time, soil, and depth. Groupings were examined when necessary, no transformations of data were necessary, and the models were then evaluated for significance using ANOVA and the covariance structures examined to determine the best fit models. The final soil column SAS statistical analysis models for each parameter follow.

```
proc mixed data=FSBOD;
class Soil Depth Sub Time;
model BOD=Soil Depth Soil*Depth/ddfm=kr outp=BODoutput;
random Column(Soil Depth Sub);
repeated time/group=Depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Depth Soil Soil*Depth/pdiff;
run;
```

```
proc mixed data=FSCOD;
class Soil Depth Sub Time;
model COD=Soil Depth Soil*Depth Sub Depth*Sub Time Sub*Time Soil*Time
Sub*Time/ddfm=kr outp=CODoutput;
random Column(Soil Depth Sub);
repeated time/group=Depth subject=Column(Soil Depth Sub) type=arh(1);
lsmeans Soil Depth Soil*Depth Sub Depth*Sub/pdiff;
run;
```

```
proc mixed data=FSTKN;
class Soil Depth Sub Time;
model TKN=Soil Depth Soil*Depth Sub Depth*Sub Time Soil*Time
Sub*Time/ddfm=kr outp=output;
random Column(Soil Depth Sub);
repeated time/group =Depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Soil Depth Soil*Depth Sub Depth*Sub/pdiff;
run;
```

```
proc mixed data=FSAmmonia;
class Soil Depth Sub Time;
model Ammonia=Soil Depth Soil*Depth Sub Depth*Sub Time/ddfm=kr
outp=output;
random Column(Soil Depth Sub);
repeated time/group =Depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Soil Depth Soil*Depth Sub Depth*Sub/pdiff;
run;

proc mixed data=FSNitrite;
class Soil Depth Sub Time;
model Nitrite=Soil Depth Soil*Depth Time Soil*Time Depth*Time/ddfm=kr
outp=output;
random Column(Soil Depth Sub);
repeated time/group =Depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Soil Depth Soil*Depth/pdiff;
run;

proc mixed data=FSNitrate;
class Soil Depth Sub Time;
model Nitrate=Soil Depth Soil*Depth Sub Time/ddfm=kr outp=output;
random Column(Soil Depth Sub);
repeated time/subject=Column(Soil Depth Sub) type=cs;
lsmeans Soil Depth Soil*Depth Sub/pdiff;
run;

proc mixed data=FSpH;
class Soil Depth Sub Time;
model pH=Soil Depth Sub Time Soil*Time/ddfm=kr outp=output;
random Column(Soil Depth Sub);
repeated time/subject=Column(Soil Depth Sub) type=csh;
lsmeans Soil Depth Sub Time/pdiff;
run;

proc mixed data=Alk;
class Soil Depth Sub Time;
model Alkalinity=Soil Depth Soil*Depth Sub Depth*Sub Time Soil*Time/ddfm=kr
outp=output;
random Column(Soil Depth Sub);
repeated Time/subject=Column(Soil Depth Sub) type=arh(1);
lsmeans Soil Depth Soil*Depth Sub Depth*Sub/pdiff;
run;
```

```
proc mixed data=Mn;
class Soil Depth Sub Time;
model Mn=Soil Depth Sub Time Soil*Time Depth*Time Sub*Time /ddfm=kr
outp=Mnoutput;
random Column(Soil Depth Sub);
repeated time/group=depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Depth Soil Sub/pdiff;
run;

proc mixed data=FSFe;
class Soil Depth Sub Time;
model Fe=Soil Depth Time Soil*Time Depth*Time/ddfm=kr outp=output;
random Column(Soil Depth Sub);
repeated time/group =Depth subject=Column(Soil Depth Sub) type=cs;
lsmeans Depth Soil/pdiff;
run;
```

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