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Nitrogen Migration from Deep-Row Biosolids Incorporation on a Hybrid Poplar Tree Farm

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Abstract. *Deep row incorporation of biosolids at rates of 383 to 659 Mg/ha (171 to 294 dry tons per acre) using hybrid poplar trees involves the placement of biosolids into trenches that are immediately covered with overburden, eliminating odor problems, and maintaining the biosolids in a fairly stable, anaerobic environment. The site is then planted with hybrid poplar trees, the roots of which provide a natural recycling system that utilizes the nutrients over a six-year period in most cases.*

For three years, nitrate beneath the trenches was essentially zero. Subsequently, nitrate has increased to between 1 and 10 mg/L. Ammonium in the suction lysimeters remained between 100-600 mg/L from March 2003 until August 2004, when there was a peak in the highest application rate to 800 mg/L from November 2004 to September 2005. Ammonium concentration is clearly decreasing with distance from the biosolids for all treatments, with significant differences between treatments becoming apparent from November 2003 to November 2005. Ammonium levels in October 2006 at 15 and 60 cm(6 and 24 in.) may have stabilized or decreased slightly to 1900 and 400 mg/L, respectively. While ammonium levels at 30 and 60 cm (12 and 24 in.)may have leveled off, additional suction lysimeters were installed during the summer of 2007 at 120 cm(48 in.) below the biosolids trench in all plots to better understand ammonium movement in the profile.

Keywords. *Biosolids, nitrogen, nitrate, ammonium, water quality*

Introduction

Biosolids pose a beneficial use challenge that, in many cases, proves costly and carries the potential to generate pollution if not well managed. It is therefore of societal interest to develop safe, effective, and economical means of biosolids disposal, or better yet, recycling.

Current United States regulations for disposal are delineated in The Standards for Use of Disposal of Sewage Sludge (Title 40 of the Code of Regulations {CFR} Part 503). In addition to incineration, landfilling, and composting, these Environmental Protection Agency (EPA) regulations allow for land application of biosolids, and strongly encourage implementation of this technique for beneficial uses. Most beneficial uses consist of land application to agricultural fields and other nutrient-deficient lands to enhance vegetative growth. In such cases, application must follow the protocols in 40 CFR part 503 to ensure that excess nutrients are not transported to surface water or leached to groundwater.

Biosolids utilization in forest lands, particularly in silviculture operations, has gained increased popularity in the United States. Surface spraying, spreading, and subsurface mixing in the soil are the primary distribution techniques, typically with applications required each year or multiple times a year to successfully meet the nutrient needs of the trees and production goals of the operation. Because it is not a food crop, concerns related to the potential uptake and ingestion of biosolids contaminants do not exist. Not only do the biosolids provide a nutrient source for the trees, they also build up the topsoil, reduce erosion and increase above and under ground ecosystem diversity.

An alternative land application regimen, referred to as deep row application, has been in use on private property owned and managed by the Environmental Reclamation Company, Inc. (ERCO, Inc.) since the early 1980s. This technique was established on an exhausted surface sand and gravel mine that, prior to reclamation as a tree farm, consisted of sand and gravel remnants underlain by a clay layer. As such, it was devoid of organic matter and subject to erosion. In concert with regulatory requirements to reclaim abandoned mine sites, ERCO devised a reclamation plan to grow hybrid poplar trees over deep rows that had been filled with biosolids. The biosolids would serve as a long-term nutrient source for the fast-growing, nutrient-demanding hybrid poplars. The poplars, in turn, would provide erosion control, wildlife habitat, and potentially become a marketable product.

Background

Biosolids production in 2002 for the Washington, D.C. & Baltimore, MD Metro area, including the counties of Baltimore, Howard, Montgomery, Prince George’s, Charles, and Anne Arundel, was 827,514 wet tons (MDE, 2002; DC-WASA, 2002). These biosolids were utilized as follows on a percentage basis: applied on agricultural land outside of Maryland (56%); hauled out of Maryland but utilization unknown (14%); applied on agricultural land in Maryland (9%); composted (7%); storage (9%); incinerated (3%); and landfilled (2%). It is clear from these statistics that Maryland relies heavily on agricultural land application in adjoining states (Virginia and Pennsylvania) to utilize the majority of biosolids produced in-state.

The passage of the Clean Water Act in 1972 resulted in elevated pressure on municipalities to find methods other than dumping to utilize biosolids from treatment plants. Presently, biosolids are surface-applied on farmland, marketed for compost, and incinerated; however, the most cost-effective methods of biosolids management are either by application to agricultural land or burial in landfills. Agricultural land application makes up a significant portion of the biosolids utilized in the Metro area, but the passage of the Water Quality Improvement Act (WQIA) of

1998 in Maryland may reduce farmland application due to phosphorous-based application requirements. Agricultural land application of municipal biosolids can boost soil productivity for field crops and improve soil characteristics. However, regular broadcast applications necessary to provide crop nutrient requirements can cause logistical, safety, and economic problems due to transportation cost, poor weather, frozen soils, restricted availability of labor, and other problems. Resentment by rural landowners and offensive odors in urbanizing areas has resulted in many local application restrictions. Difficulty in permitting and developing new landfills and possible future restrictions on out-of-state hauling may result in restriction and/or increased cost of landfill disposal of biosolids. The developing drawbacks of landfill and agricultural land application points to the need for alternative utilization technologies for biosolids that are both cost-effective and environmentally sound (Sikora and Calacicco 1980; Kays et al., 1997).

Previous Work

The land application of biosolids on native forests, reclamation sites, and plantations through regular broadcast applications has been used in other parts of the country, with significant growth responses documented (Cole et al., 1986; Heilman et al., 1995; Sopper, 1993; Aschmann, 1988; Purkable, 1988). Deep-row biosolid applications for forest products production has the potential to solve many of the problems associated with agricultural land application and other land disposal methods and enhance the multi-state Chesapeake Bay cleanup effort.

Deep Incorporation Research

Documented records regarding the utilization of sewage as fertilizer dates back to the 1500s in Germany, where sewage was used on croplands. Under the Federal Water Pollution Control Acts of 1972, land application of biosolids was recognized as a protocol for disposal, provided the disposal was managed in accordance with the applicable regulations. In conjunction with this recognition, experts from the EPA, United States Department of Agriculture (USDA), and National Land Grant Universities pooled their resources to form a Coordinating Committee on Environmental Quality that developed a subcommittee on Recycling Efforts of Sludges on Land. This subcommittee evaluated research that had been conducted on the pros and cons of biosolids application to provide guidance on the most appropriate protocols for use. This increased interest, along with the ongoing buildup of biosolids at wastewater treatment plants, sparked a series of research projects that evaluated the impacts of biosolids application to land (Lue-Hing, et al., 1992).

Burying biosolids in deep-rows covered by a soil overburden was researched in the 1970s (Sikora and Colacicco, 1980; Taylor et al., 1978). In the early 1970s, the Washington Suburban Sanitary Commission purchased hundreds of acres of land in the counties surrounding Washington, D.C. for the purpose of burying biosolids in trenches at rates approximately equal to and greater than 383Mg/ha (171 dry tons per acre). Long-term monitoring of the sites has found elevated nitrates in groundwater at some areas (Sikora et al., 1982). While the production of annual corn crops on treated areas was researched (Sikora et al., 1980), no forest crops or other deep-rooted perennial crops were intentionally established to utilize the large reservoir of nutrients.

Two studies are of particular interest. Sikora, with USDA-ARS at Beltsville, MD, reported on trench studies in both sandy soils and heavier soils. Sikora's tests placed lime-stabilized biosolids in trenches 610 mm (24 in.) wide by 500-1300 mm (20-50 in.) deep on 1270 mm (50 in.) centers, covered with 0.15-0.30 m (0.49-0.98 ft.) of subsoil. Water samples were collected from drainage tile lines, a catchment pond, and monitoring wells within and around the trenched plot. Sikora's various experiments grew corn and grass in field studies and Taylor et al. (1978) grew corn in 160 day simulated deep-row experiments in a greenhouse setting.

In the Beltsville studies (Sikora et al., 1982), five years after the application on Manor and Glenelg silt loam soils, no increases in N and Cl were detected in groundwater, although elevated levels were found in the soil water just beneath the trenches. However, in similar studies on a sandy soil (Sikora et al., 1979a), groundwater pollution was recorded. Specifically, these studies showed a peak in chloride levels 18 months after entrenchment and a peak in nitrate concentration a year after the chloride peak (i.e., 30 months after entrenchment). Nitrate concentrations were below the EPA MCL of 10 mg/L nitrate-N (NO_3-N) in wells above and below the trench plot. Though a high nitrate concentration of 60 mg/L occurred during November 1974 in one well within the trench plot, most concentrations (>85%) were below 10 mg/L. Tile drains exhibited a high NO_3-N concentration of 32 mg/L. Other observations of note were that metals did not migrate and pathogens were significantly reduced.

Metal movement through soil is generally considered minimal except in instances when the pH is below 5.5 (Chaney et al., 1977), which is not a problem on most sites due to liming requirements. The general conclusion concerning groundwater pollution by biosolids at other deep-row sites is groundwater immediately beneath the sites has the potential to experience increases in N and Cl and these levels decrease with time.

The researchers (Sikora et al., 1979b, 1980) noted some interesting characteristics of the biosolids in the trenches. Observations included an analysis of the original sludge sample and then the progression of the sludge starting at 22 months after entrenchment. First, the biosolids dewatered from the top down, or, in other words, the tops of the trenches were dry, whereas the bottoms of the trenches remained wet. After 22 months of entrenchment, the top portion of the sludge (5-20 cm or 2-8 in. from the top of the trench) had dried out and was densely penetrated with roots. The middle and bottom portions of the trench did not dewater until 49 months after entrenchment. After this four-year period, the entire trench contents appeared to have stabilized. Similar to Walker's observations (1974), dewatering occurred from the top down. This observation led to the conclusion that mineralization and subsequent transformations began in the uppermost portion of the biosolids shortly after entrenchment but that denitrification was taking place concurrently as the leachate from the upper portion of the biosolids moved into the wetter, lower portions of the entrenched biosolids. Sikora et. al. (1980) reported on the trenching of digested biosolids. Certain physical observations of the biosolids-filled trenches are meaningful. The first sampling of these trenches occurred almost two years after biosolids placement. At that time the top portion of the trench was densely rooted and "peat-like" and the middle portion was only sparsely rooted, wet in appearance, and odorous. After four years, the top and middle portions were brown and odorless. The Sikora team concluded that trenched biosolids become "stabilized" with respect to further decomposition after about four years.

Nitrogen Fate

The Nitrogen Cycle.

In order to understand the implications of sewage sludge disposal techniques and associated scientific studies, the nitrogen cycle must be understood. Nitrogen is one of the most important nutrients for plant growth. Only certain water-soluble inorganic forms, however, including ammonium ($NH4^+$) and nitrate (NO_3^-), can be absorbed by higher plants. In biosolids, the ratio of organic to inorganic forms of nitrogen is determined by the treatment process. Liquid anaerobically digested sludge may contain a majority of nitrogen in the form of ammonium, with lesser amounts as organic nitrogen and negligible amounts of nitrate. In undigested lime-stabilized biosolids, however, the majority of nitrogen present is in the form of organic nitrogen (Shepherd, 1996; Gshwind and Pietz, 1992). Several biochemical processes must therefore occur before plants benefit from this nutrient source. Mineralization is an enzymatic process in which organic nitrogen is decomposed to inorganic forms. The first step is ammonification, in which microbes break down organic nitrogen and produce the ammonium cation ($NH4^+$). This process occurs in either anaerobic or aerobic conditions and is performed by a broad group of heterotrophic organisms.

Nitrification consists of two main sequential steps that include: 1) the oxidation of ammonium to nitrite (NO_2^-) by the autotrophic Nitrosonomas bacteria; and immediately thereafter 2) oxidation of nitrite by Nitrobacter bacteria to produce nitrate. The swift transition from nitrite to nitrate prevents accumulation of nitrite. Both of the nitrifying organisms responsible for this reaction sequence are aerobes, requiring the presence of oxygen to perform these conversions. In addition, they favor soils with no more than 60% of pore volume filled with water, need a carbon source (i.e., bicarbonates and carbon dioxide), and optimally perform at temperatures between 20-30°C (68-86°F)(Brady and Weil, 2002).

Nitrate is an anion that is not readily adsorbed to soil particles, is water soluble, and is therefore highly mobile. Of the forms of nitrogen described above, nitrate presents the highest risk of leaching through the soil profile to the groundwater table. Additionally, nitrate warrants the most concern from a human health and environmental pollution perspective. Most acutely in infants and ruminant animals, ingested nitrate is reduced to nitrite, which decreases the oxygen-carrying ability of red blood cells and produces a condition known as methemoglobinemia (Brady and Weil, 2002). Consequently nitrate is a regulated pollutant in drinking water with a Maximum Contaminant Level (MCL) of 10 mg/L for NO_3-N (EPA, 1994).

Initially, biosolids contain extremely low levels of nitrate, about 0.019% (Pepperman, 1995). Nitrate evolves slowly from biosolids when anaerobic conditions prevail and lime stabilized biosolids have a significantly lower nitrate production rate than do digested biosolids (Taylor et al., 1978).

Nitrate also can have a pronounced impact on aquatic systems. An influx of nitrate promotes algal blooms that, upon dying, are decomposed by oxygen-demanding bacteria. Exponential growth and decay results in exponential demand and depletion of oxygen. Hypoxic conditions result that are toxic to many forms of aquatic life. Proliferation of this cycle can expand these inhospitable zones on a yearly basis, rendering once productive waters lifeless.

The converse of mineralization is immobilization, in which ammonium or nitrate is complexed into an organic form via biotic or abiotic means. Both processes occur simultaneously, as microbe populations grow and die, and rates are dependent upon the composition of the soil. Some forms of nitrogen, particularly organic nitrogen and ammonium, can also be adsorbed on active sites of the soil, limiting movement through the soil profile.

Denitrification refers to those processes in which nitrate ions are converted to gaseous forms of nitrogen {e.g., nitric oxide gas (NO_2^+), nitrous oxide gas (N_2O^+), and nitrogen gas (N_2)}. The majority of bacteria performing this function are facultative anaerobes that can be either heterotrophs (i.e., obtain their energy and carbon from oxidation of organic compounds) or autotrophs (i.e., obtain their energy and carbon from carbon dioxide or carbonates). Required environmental conditions include: low soil air content (<10%), temperatures between 2-50°C (36-122°F)(with an optimum range of 25-35°C, or 77-95°F), and an appropriate energy source (Brady and Weil, 2002).

Patrick and Gotoh (1974) studied the impact of O_2 levels in nitrogen loss from saturated soils and indicated that anaerobic conditions greatly inhibited biological oxidation of NH_4^+ to NO_3^-. In their study, nitrate that was formed then migrated to an anaerobic layer where denitrification occurred. Again, under very saturated conditions, Lindau et al. (1988) demonstrated that nitrogen applied as N-urea and N – KNO_3 was denitrified. Between 44% and 77% of applied N was denitrified and between 28% and 40% of the applied and denitrified N became trapped in the soil.

Land Application of Biosolids.

Land application of biosolids to improve soil conditions, enhance crop production, and reclaim mined land has been extensively studied. Biosolids are either applied 1) on the surface, 2) by disking or plowing into the soil to a prescribed depth (usually no more than 15 cm or 6 in.) or 3) via injection underneath the surface. Nitrogen requirements of the crop and background soil concentration dictate application rates, with seasonal or yearly applications often being performed. Site and crop specific management are the key to optimizing growth while preventing nitrogen loss from the system (Ritter, 2001; USEPA, 1994b; Outwater, 1994; Granato and Pietz, 1992).

Numerous examples of nitrate leaching under biosolids-amended agricultural land have been reported in the literature (Ritter, 2001; Shepherd, 1996; Clapp, et al., 1994; Sopper, 1993). In these studies, the timing and rate of application, type of biosolids used, nutrient demands of the crop, and soil conditions influenced the loss of nutrients. Often, a majority of the leaching could have been prevented through more careful management.

Conversely, other studies have been performed that demonstrate the ability to minimize nitrate leaching. Studies as varied as those performed by Mitchell, et al. (2000) in a small stand of Scots pine in Scotland to larger scale reclamation operations (Van Ham, et al., 2000; Sopper 1993; Lue-Hing, 1992) and agricultural operations (Shepherd, 1996) show that with appropriate biosolids type, application rates, and conditions, nitrogen from the biosolids can be preserved and recycled in the upper layers of the soil profile. The reclamation project presented by Sylvis Environmental in British Columbia (Van Ham, et al., 2000) transformed nutrient depleted gravel mines into self-sustaining tracts of vegetation that increased the environmental quality of the site. The vegetation not only enhanced the aesthetic and ecological value of the site, but actually reduced nitrogen and phosphorus movement that previously migrated to a nearby aquifer. When properly used, biosolids are an environmentally safe and effective nutrient source that greatly improves soil condition, optimizes crop production, and enhances the soil and land ecosystem into which it is introduced.

Leaching potential.

Monitoring of nitrogen and chlorides in biosolids and soils below trenches was conducted in an effort to determine potential for leaching (Sikora et al., 1980). Chloride, a water-soluble anion commonly found in biosolids, does not interact chemically with most soils and provides an indication of water flow and maximum leaching potential through the biosolids and soil profile. The data from these analyses demonstrated two distinct trends. First, the levels of ammonium, nitrate and chlorides all generally diminished from the first sampling period (665 days) to the last (1,508 days).

The second observation made from these data is an apparent enrichment of both ammonium and chloride with depth on the same sampling date. For example, for almost every sampling event, the dry weight concentration of ammonium was greater in the lower portion of the trenches than in the middle portion. The concentration of ammonium in the middle portion was generally greater than in the top portion.

This distribution did not hold for nitrate. For each sampling date, nitrate nitrogen concentration in the lowermost portion of the trench was less than or about equivalent to the concentrations in the upper two sections of the trench. Given relatively high levels of ammonium, the precursor to nitrate formation in these samples, it would be expected that the nitrate concentration in the samples would show similar trends as ammonium and chloride unless some mechanism for nitrate removal was acting.

Leaching is the first mechanism that comes to mind to explain this anomaly. However, the enrichment of the lowermost portion of the trench with chlorides suggests that leaching was not occurring rapidly enough to account for low nitrate concentrations. Two other explanations are plausible. The first is that conditions in the lower section of the trench were not conducive to nitrate formation, so conversion of ammonium was quite slow (this would account for the accumulation of ammonium in the lowermost portion). The production of nitrate via mineralization of ammonium requires an aerobic environment, which only existed in the top of the trench at the beginning of the experiment. Subsequent dewatering of the trench fostered conditions for additional mineralization to occur deeper in the trench. A second mechanism may be denitrification. Conditions that are not favorable for nitrification are required for denitrification. It is probable that both mechanisms were at work (Pepperman, 1995).

Denitrification.

Also important to note is that once produced, nitrate will either 1) be taken up by plants or microorganisms or 2) leach further down the trench with the water flow and/or 3) undergo denitrification. The fact that nitrate concentrations do not correspond to the timing patterns exhibited by the equally water-soluble chloride indicates that 1) nitrate production via mineralization was delayed for months after biosolids entrenchment and 2) once produced, though some nitrate may have leached the bottom of the trench, the waterlogged, anaerobic conditions were optimal for denitrification. This theory is supported by the fact that concentrations in the bottom of the trench did not reach the levels in the upper portions. Additionally, concentrations in the soil below the trenches, though elevated for a time to a maximum of 54 mg/kg, decreased to low levels (2-6 mg/kg) by the end of the experiment.

Sikora et al. (1982) found that NO_3–N levels in biosolids did not change between 20 months and 45 months except in the top 20 cm (8 in.). The inorganic N content in water beneath the biosolids increased and then decreased with time. Denitrification in the soil profile was demonstrated. Walker (1974) indicated that entrenchment promoted slow nitrification and favored denitrification. Again, nitrogen was found beneath the biosolids but not in groundwater wells.

The comparison of chloride and nitrate concentrations in water samples from below the biosolids was utilized to assess the potential for leaching and, in this study, also to determine if denitrification occurred. The ratio of nitrates to chlorides decreased with depth below the trench, indicating that there existed some mechanism for reduction in nitrates (since both nitrates and chlorides are expected to move through the soil at generally the same rate). Since there were no plant roots at the depths evaluated and microbial immobilization was discounted, it appeared that denitrification was occurring (Sikora et al., 1979b).

Taylor and his fellow researchers (Taylor et al., 1978) indicated that the relatively low oxygen and high methane content of the soil atmosphere adjacent to the biosolids would be an ideal environment for denitrification. It was suggested that, from the levels of nitrate found within the biosolids, after 160 days some nitrification had occurred. They concluded, however, that the extremely low levels of nitrate within the soil surrounding the biosolids indicated that, if such a transformation were occurring, very little nitrate was moving from the biosolids. They further concluded that it was likely that any nitrate which did move from the biosolids would have been subjected to denitrification.

Literature suggests that mineralization is depressed by both temperature and anoxic conditions. These same conditions favor denitrification, so nitrate is generated only slowly and it is likely that any nitrate that is not quickly captured by the roots of the trees is denitrified.

Experiments provided evidence, however, that recharge would likely dilute the nutrients. Consequently, the specific characteristics of an individual site would need to be evaluated to determine if groundwater contamination posed too much risk for this technique. It is important to note, however, that these experiments did not attempt to utilize a deeply rooted crop or plant a specific crop density that could reach and utilize the nutrient reservoir supplied by the biosolids.

Nutrient Losses to the Vadose Zone

The following is a literature synthesis that identifies nitrate concentrations emanating from corn crops, from corn crops that use biosolids, and finally, from strip mine spoils treated with biosolids. This provides a backdrop against which the results from this project can be compared.

Nitrate in Corn - fertilizer losses

This section is a brief literature review of data evaluating corn losses of nitrate for the purpose of comparing the deep row forestry system to a well researched conventional agricultural practice. Baseline NO_3-N in subsurface drainage water ranges from 3-10 mg/L if no fertilizer is added to the crop. This level of fertilizer results in a crop that is not economical (Andraski et al., 2000). For corn with yields of 125 to 175 bushels per acre, nitrate in drain tile water ranged from 18 to 30 mg NO_3-N/L (Weed and Kanwar, 1996; Kanwar, 1997). Randall et al. (1997) found nitrate in drain tile water to average 23 mg NO_3-N /L for corn-soybean two-year rotation and 32 mg NO_3-N /L for continuous corn. Randall and Vetch (2005) found nitrate in drain tile water to range from 10.7 mg NO_3-N /L to 14.3 mg NO_3-N /L for the corn phase of a corn-soybean two-year rotation. Jaynes et al. (2001) found that flow-weighted NO_3-N concentrations for economically viable levels of corn production were13-25 mg NO_3-N /L. Most of these studies indicated that nitrate concentration could be reduced to 10-12 mg NO_3-N /L if N application rate was reduced, cover crops were used, applications were split, and continuous corn was replaced with a corn-soybean two year rotation.

During periods of depressed rainfall, no subsurface flow occurred beneath crop systems that had high above ground biomass, while flow occurred beneath continuous corn and corn-soybean rotation (Randall et al. (1997). Perennial crops reduced the amount of residual soil nitrogen and reduced the drainage water flow compared to row crops.

Andraski et al. (2000) found NO_3-N levels from 18 to more than 20 mg/L in suction lysimeters for various economically viable levels of corn production.

In summary, nitrate in soil water and drainage water ranged from 10 mg/L if numerous BMPs were applied to 32 mg/L for continuous corn.

Nitrate in Corn - biosolids losses

Agronomic soils.

Nitrate losses from spring-fertilized corn were greater than losses from spring-applied anaerobically digested biosolids Evanylo (2003) and coarser soils lost more nitrate than did finer soils.

Binder et al. (2002) applied biosolids to plots in the spring of 1996. Sampling indicated that a pulse of nitrate moved through the profile. It was not until the fall of 1999 that the nitrate distribution profile returned to levels that were similar to untreated soils (Figure 1). The authors point out that the nitrate in the profile had two exit pathways: leaching and denitrification.

In the same experiment, rates of biosolids were examined. For biosolids applications in the spring of 1999 and soil tests from the fall of 1999, where the application rate was approximately the recommended rate of 177 kg organic N/ha (160 lbs. N/ac.) there was less nitrate in the profile than in the control (Figure 1). When the application rate was increased to 353 kg organic N/ha (315 lbs. N/ac.) the profile had slightly more N than the control. Only when excessive rates were applied (706 kg organic N/ha or 630 lbs. N/ac.) did the nitrate in the profile greatly exceed the control.

In the same experiment, biosolid application rates were examined. For biosolids applications in the spring of 1999 and soil tests from the fall of 1999, where the application rate was approximately the recommended rate of 177 kg organic N/ha (160 lbs. N/ac.) there was less nitrate in the profile than in the control (Figure 2). When the application rate was increased to 353 kg organic N/ha (315 lbs. N/ac.) the profile had slightly more N than the control. Only when excessive rates were applied (706 kg organic N/ha or 630 lbs. N/ac.) did the nitrate in the profile greatly exceed the control.

Clearly, surface applied biosolids increases soil nitrate concentration and higher application rates result in higher levels of soil nitrate (Binder et al., 2002).

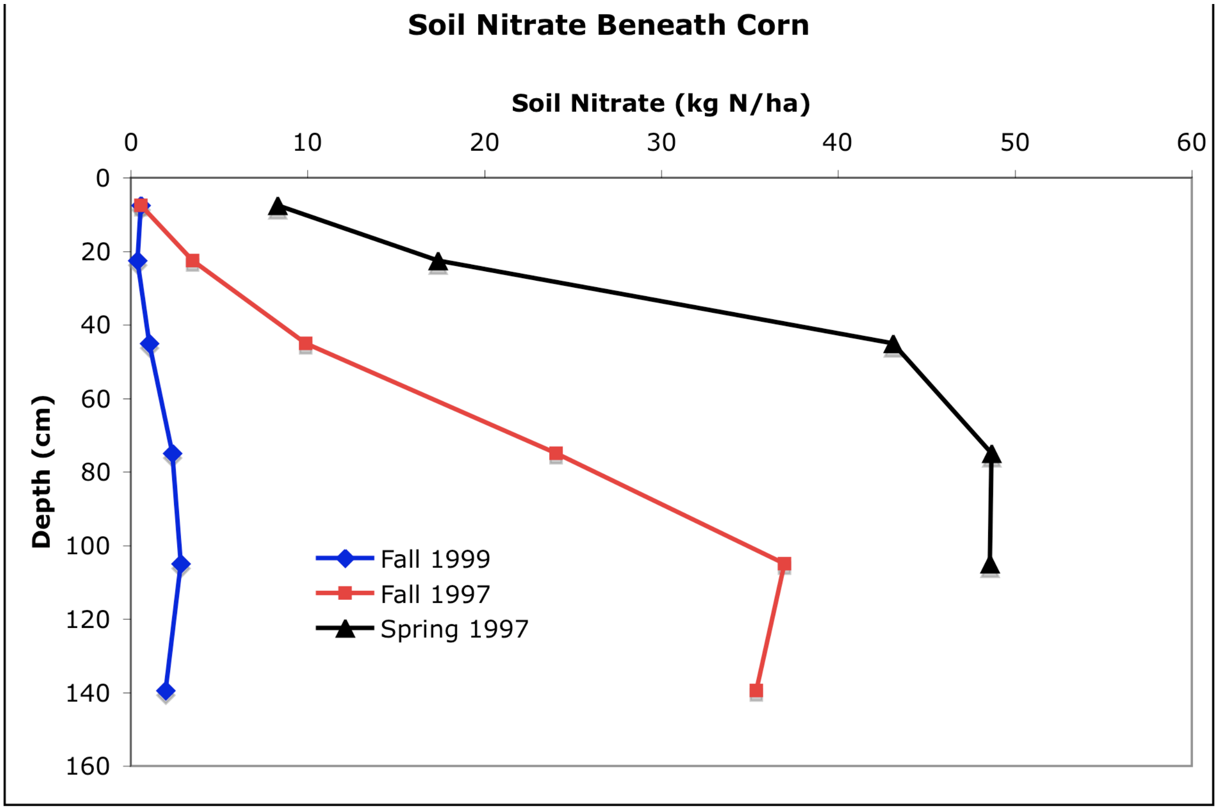

Figure 1. Soil nitrate profiles at different times following spring 1996 application of biosolids (Binder et al., 2002).

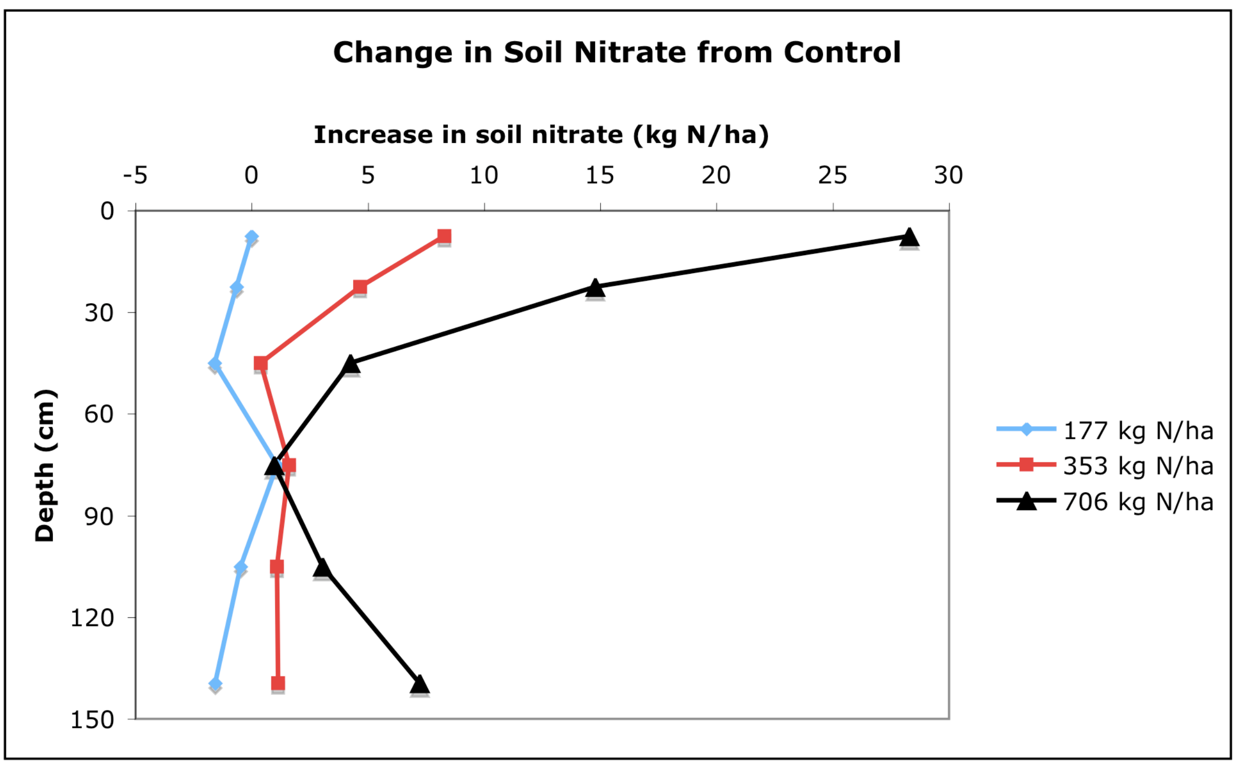

Figure 2. Change in soil nitrate from control levels for different rates of biosolids applications (Binder et al., 2002).

Strip mine spoils.

A mixture of 50% anaerobically digested biosolids and 50% composted anaerobically digested biosolids were applied at rates of up to 152 dry Mg/ha (68 dry tons/ac) or 5290 kg organic N/ha (4719 lbs. N/ac.). Using pan lysimeters installed one meter (3.3 ft)deep on a Pennsylvania strip mine, nitrate loss (Figure 3) occurred beginning approximately five months after application (Oct. 2001) and tapered off to low values by June of the following year (Stehouwer et al., 2006). The peak nitrate concentrations occurred in October 2001 and were greater than 300 mg NO_3-N/L. A smaller peak (150 mg NO_3-N/L) occurred in October of 2002. It was estimated that one-third of applied N was lost from the surface by leaching. Four groundwater monitoring wells were installed at between 5.2 and 9.4 m depth (17-31 ft.) and these wells showed very little impact from the nitrogen leaching. Brief increases in NO_3-N rose to concentrations of 6.5 mg NO_3-N/L or less.

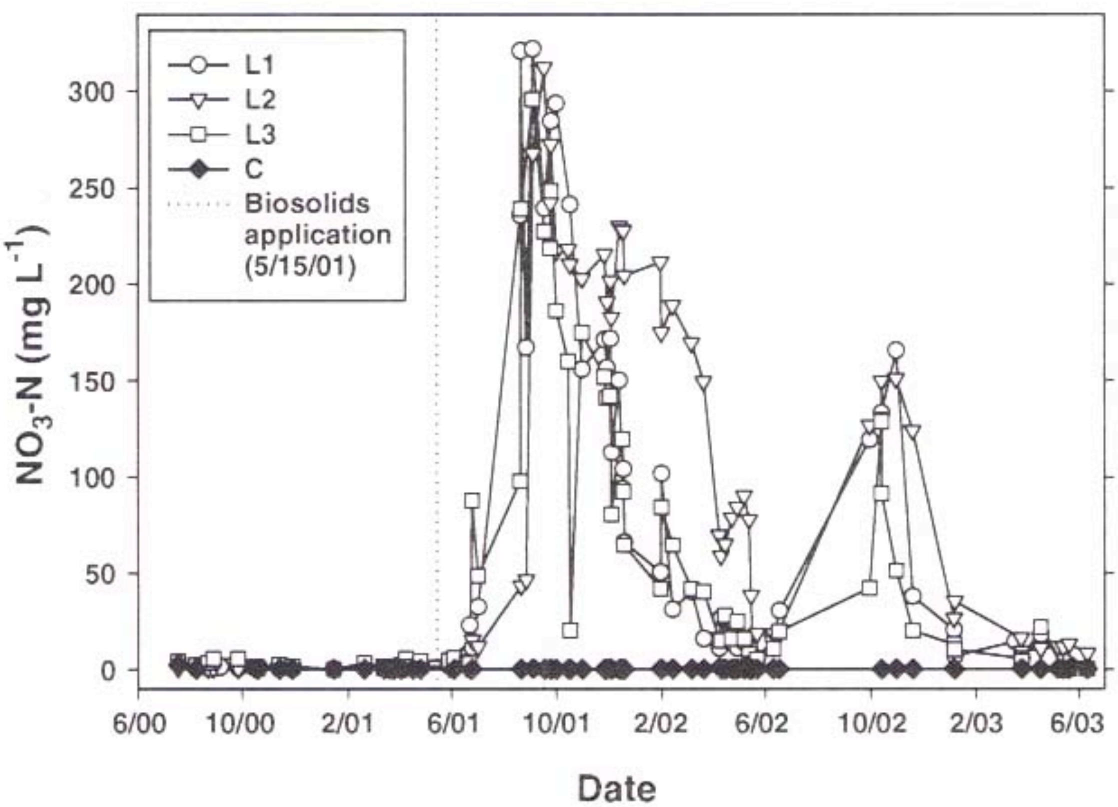

Figure 3. Nitrate concentration in pan lysimeters before and after application of 50% anaerobically digested biosolids and 50% composted anaerobically digested biosolids (Stehouwer et al., 2006).

The high rates of biosolids application in the above experiment reflect the philosophy involved in reclamation. By adding a large amount of organic matter, the one to two year leaching of nitrate is sacrificed for an ecosystem that can become established and sustain itself on the strip mine spoil, something that does not usually occur if only mineral fertilizer is used.

Denitrification

Ryden (1981) measured denitrification rates between 47 and 69 kg N/ha/year (42 and 62 lbs. N/ac./year) from pasture land irrigated with municipal waste water. Russel et al. (1993) measured rates between 12 and 240 g N/ha/hr (0.011 and 0.214 lbs. N/ac./hr) immediately following irrigation of a forest site with meat processing effluent. This corresponds to about 100 to 2,000 kg N/ha/year (89 to 1,784 lbs. N/ac./year).

Nitrate in Surface Waters

A study by Tian et al. (2006) reported on surface application of biosolids at and average rate of 28.2 dry Mg/ha/yr (10.4 dry ton/ac/yr) for a 31 year period. Water was sampled for nitrogen from 20 reservoirs and 6 creeks. Sampling rate was monthly for the first decade and then 3 times per year following that. Nitrate in the receiving waters rose from 0.86 mg NO_3-N/L at time zero to just below 2.52 mg NO_3-N/L after 360 months. The difference between the control and the treated waters was 0.13 mg NH_4-N/L. The authors state that "the application of biosolids for land reclamation at high loading rates from 1972 to 2002, with adequate runoff and soil erosion control, had only a minor impact on surface water quality".

The objectives are to determine the effect of biosolids application rate and tree density on water quality around the deep rows on a gravel mine spoil, the nutrient losses to the vadose zone, nutrient removal by the trees, growth and survival of hybrid poplar, and to continue education of state and local environmental professionals about deep row applications to develop sustainable forest crops and simultaneously rehabilitate disturbed soils.

Methods and Materials

In 1983, ERCO, Inc. developed the deep row application technique in response to the need to utilize large volumes of biosolids from the Washington, D.C. area and reclaim sand and gravel surface mine spoils. The company received a permit from the Maryland Department of Environment (MDE) for application of biosolids to grow nutrient-demanding hybrid poplar trees on nutrient-poor sand and gravel strip mine spoil. The trees were harvested at about 7 years of age when foliar leaf samples were below 3.5 percent nitrogen and total nitrogen mineralization in the biosolids reached 70 percent.

Approximately 10 acres were treated each year starting in 1984. The deep row technique initially involved the application of biosolids at a rate of 383.3 dry Mg/ha (171 dry tons/ac.) and, for a special demonstration plot, at a rate of 659.1 dry Mg/ha (294 dry tons/ac.). Biosolids were placed in trenches that were 30 inches deep, 42 inches wide, and spaced approximately 8 feet on center. Trenches were filled with 18 inches of biosolids. The remaining 20 to 30 cm (8-12 in.) of trench were filled with overburden. The overburden soils were limed to obtain a pH of 6.2 as per permit requirements. Between 1984 and 1996, fast-growing, nitrogen-demanding, hybrid poplar cuttings were planted at a dense spacing of 7,400-9,900 trees/ha (3,000-4,000 trees/ac.) to utilize the nitrogen over a planned 6-year rotation. Since 1996, the tree spacing has been changed to 3 meter by 3 meter (10 ft. by 10 ft.) because the trees were found to grow much more with this spacing. Competing vegetation was controlled by mowing (no herbicides were used). After six or more years, a 4 hectare (10 ac.) section was harvested and subsequently cross-trenched for another biosolid application.

Site Location

The ERCO Beneficial Reuse Tree Farm site is a privately-owned 49.4 ha. (122 ac) sand and gravel mine spoil in Prince George's County, MD within 40 km (25 miles) of many large municipal wastewater treatment plants. The site is in the coastal plains physiographic region, approximately 32 kilometers (20 miles) east of the escarpment region that identifies the piedmont physiographic region. The site is approximately three miles north of Waldorf, MD (Figure 4).

Figure 4. ERCO study site, located in Prince George's County, MD within the Washington, D.C. metro area.

Site Description

The site consists of a plateau with steep banks that fall away to incised streams. The edges of the plateau are bermed and runoff is routed to one of seven detention ponds. All steep banks are covered with permanent forest cover. The plateau has an upper area (two sections) near the entrance on a 0-2% slope. The remaining seven sections have an elevation drop of between 1.5 and 3 m (5-10 ft.), followed by a level section (0-2% slope) to the edge of the plateau.

The research site is an existing reclamation site that has utilized deep row biosolid application with forest trees for 15 years. Prior to any biosolid application, the reclamation site was representative of thousands of acres of sand and gravel mines in the Metro Washington, D.C. area. The entire site has been applied once with biosolids using deep row application. Approximately 25% of the site (13 ha or 32 ac.) is in permanent cover, consisting of either forested steep slope, or detention ponds and buffers. At any one time, only one or two sections (4.05 ha or 10 ac. each) are cleared and replanted. Hence, only 8-16% of the site is subject to significant surface runoff. In addition, the surface water flow on the site is significantly reduced due to the tree crops.

Soils and Geology

There are conventional soils on the steep side slopes that were not disturbed by sand and gravel mining, but there are no soils, as we normally think of them, on the plateau surface. In 1983, following cessation of the sand and gravel mining activity, the spoil consisted of a clay layer with occasional remnants of sand and gravel and some gullies that were filled with spoil during the re-grading process in 1983. The clay layer was 1.5 m to 21.3 m (5' – 70') or more thick. The following description of soils and geology at the ERCO site was derived from Wilson and Fleck (1990) and, to a lesser extent, Tompkins (1983) and begins with the deeper deposit first and concludes with the surface deposit that was removed in the mining operations.

The lower formation is the Marlboro Clay (late Paleocene), a leaky confining unit of dense, reddish silty clay between 4.6 m and 7.2 m (15' – 30') in thickness. The lower Eocene Nanjemoy formation overlies the Marlboro Clay, and predominantly consists of beds of dark green, fine to medium, glauconite-bearing sands in the upper part of the formation and is a water-supply aquifer in many parts of southern Maryland. The thickness of the Nanjemoy at Waldorf ranges from 27.4 m to 38.1 m (90' – 125').

Overlying the Nanjemoy is the lower Miocene Calvert Formation. The Calvert is a light to medium, olive gray to olive green, micaceous, clayey silt which acts as a hydrologic confining unit. The thickness of the Calvert in the Waldorf area is 27.4 m to 30.5 m (90' – 100'). The formation is the basal unit of the Chesapeake Group and it represents deposition in a marine shelf environment.

The Calvert is overlain by the Pliocene Upland Deposits. The Upland Deposits consist of orange-tan, silty, fine to very course sands and gravels, and yellowish to orange, silty clays. The Upland Deposits range from 6.1 m to 15.2 m (20' – 50') thick and crop out throughout the Waldorf area. These materials were removed in the sand and gravel mining process. Hence, the ERCO site has very slight remnants of the Pliocene Upland Deposits over the Calvert clayey silt, over the Nanjemoy.

Experimental Design

This section describes the standard tree farm production implemented since the 1980s, followed by the design of the University of Maryland experiments performed at the site since 2002.

Production Treatments.

The deep-row technique, developed in 1983, involved the application of biosolids, averaging about 20 percent solids, that were lightly amended with lime to control odor (but not lime-stabilized), at a rate of 383.3 Mg/ha (171 dry tons/ac.). The pH of the biosolids ranged from 7.0-8.0. Approximately 4.05 ha (10 acre) sections were treated each year beginning in 1984. The deep row dimensions were 762 mm (30 in.) deep and 1100 mm (42 in.) wide, spaced on or about 2.44 m (8 ft.) centers. The deep-rows were filled with 460 mm (18 in.) of biosolids for the 383.3 Mg/ha (171 dry tons/acre) rate and 560 mm (22 in.) for the 659.1 Mg/ha (294 dry tons/acre) rate. The remaining 200-300 mm (8-12 in.) was filled with overburden. After each section was filled, the site was leveled using a low-ground pressure bulldozer, and disked in preparation for planting. Application rates used at the farm are similar to experimental trenching site applications made from 1974 through 1980 on well-drained, silt loam soils of the Manor and Glenelg soil series (Sikora, et al., 1982).

Experimental Treatments.

The 1.3 ha (3.1-ac.) study site is located on the existing ERCO property and has previously received one biosolids application, as described above (Production Treatments). A replicated treatment design was used to determine the effect of three tree densities: 0, 716, and 1074 trees per hectare (0, 290, and 430 trees per acre) and three deep row biosolid application rates: 4,400, 8,800, and 13,200 kg N per hectare (4,000, 8,000 and 12,000 lbs. N per acre) on water quality and tree production. Unlike past application rates, which were based solely on biosolids weight, the experimental rates are expressed inkilograms of nitrogen per hectare per year (pounds of nitrogen per acre per year). The application rate of biosolids in units of dry Megagrams per hectare (dry tons per acre) required to meet these nitrogen targets will depend on the N content of the biosolids used.

Prior to beginning applications in mid-March 2002, a biosolids sample was collected from a routine delivery at the ERCO site to determine nitrogen content and the corresponding application rates necessary to meet the research requirements. Results showed a total nitrogen content of 1.14% (wet weight basis). Three other samples subsequently collected confirmed this general value, and all four samples together produced an average value of 1.16% total N.

The experimental application rates of 4,400, 8,800, and 13,200 kg N/hectare (4,000, 8,000, and 12, 000 lbs/N per acre) bracket the production level of (4,730 kg N/ha or 4,300 lbsN/ac.), and are designed to discern the most appropriate application rate that results in higher crop production while protecting water quality.

Table 1. Approximate nitrogen treatment rates, depth of biosolids in the trench, total trench depth, and approximate biosolids application rate.

Application Rate kg N/ha (lbs. N/ac..)	Depth of Biosolids cm (in.)	Total Trench Depth cm (in.)	Biosolids Rate Mg/ha (dry tons/ac)
4,400 (4,000)	31.8 (12.5)	61 (24)	386 (172)
8,800 (8,000)	63.5 (25.0)	94 (37)	773 (345)
13,200 (12,000)	95.3 (37.5)	124 (49)	1159 (517)

Plot Layout.

Beginning in spring 2002, plots were established at the ERCO site. The site was partitioned into three blocks based on a north-south gradient of changing soil composition and slope. Each block contains each biosolids application rate/tree density combination. The project funded by WSSC required 18 plots (2 tree densities)(3 biosolids rates)(3 replications). Funds from the McIntyre Stennis grant provided for an additional 12 plots that consisted of: three biosolids rates with no trees replicated three times (9 plots) plus control plots with no biosolids and no trees, replicated three times (3 plots). The result is an incomplete split block experimental design.

Each plot that received biosolids is 22 meters wide (72 feet wide) with 11-12 rows of biosolids. Plots that were planted with 1,074 trees/hectare (435 trees/acre) are 21.3 meters (70 feet) long to accommodate 3 meters x 3 meters (10 foot x 10 foot) tree spacing (8 rows x 8 columns of trees). Plots that are planted with 716 trees/hectare (290 trees/acre) are 32.0 meters (105 feet) long to accommodate 3 meters x 4.6 meters (10 foot x 15 foot) tree spacing (again 8 rows x 8 columns of trees). The no-tree biosolids plots are 11 meters (35 feet) wide. Figure 5 provides a layout of the relative locations of the three blocks and the treatments within each block as they were installed at ERCO. The total area depicted is 12,406 square meters or 1.24 hectares (133,540 square feet or 3.11 acres).

Within each plot the outer two rows of trees around the perimeter were designated as buffers to isolate treatments and provide access routes, thereby reducing disturbance of soil and vegetation in the plots. The sample collection areas within each plot consist of the innermost 16 trees, to reduce possible edge effects. The central area of four rows by four columns of trees contains all soil water sample collection equipment. The three control plots (no trees, no biosolids) are 11 meters x 11 meters (35 feet x 35 feet) with instrumentation in the central portion of the plots.

Biosolids application rates were randomized assigned within each block. Tree plantings were not randomized due to logistical considerations associated with the equipment and labor used.

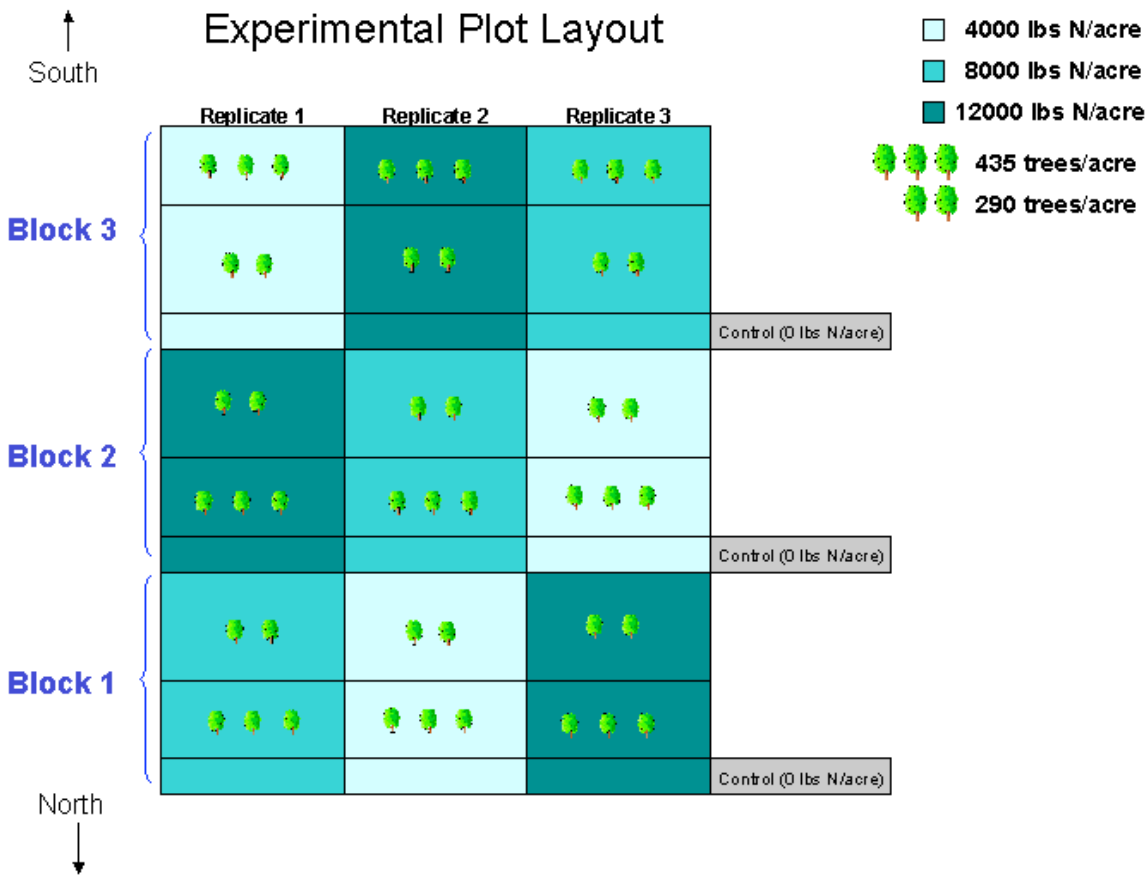

Figure 5. Schematic layout of all three blocks showing total area, nitrogen treatment rates (4,400 kg N/ha, 8,800 kg N/ha, and 13,200 kg N/ha or 4,000 lbs. N/ac., 8,000 lbs. N/ac., and 12,000 lbs. N/ac.), and tree densities (716 trees/ha and 1074 trees/ha or 290 trees/ac. And 435 trees/ac.).

Water Quality Instrumentation and Measurement.

Each treatment (application rate x tree density combination) within each block contains several types of sampling instrumentation to evaluate hydrology and/or nutrient transport: 1) two shallow stand-pipe wells were installed in the 4,400 kg N/ha x 1074 trees/ha (8000 lbs. N/ac. x 435 trees/ac.) plot, with one well positioned through the deep row and the other well in the surrounding soil/gravel spoils profile; 2) in each of the 30 plots, one zero-tension lysimeter positioned 30 centimeters (12 in.) directly under a deep row; and 3) in each of the 30 plots, suction lysimeters located under and around the deep row.

Each plot has one zero-tension lysimeter installed 25 cm (10 in.) below the bottom of the trench. Water collected from zero-tension lysimeters (a.k.a, pan lysimeters) is predominantly macropore flow. Where macropores are minimal or non-existent, as may be the case in this area, the flow represents gravity-drained water. This flow is estimated to account for anywhere between 10 to 85 percent of the percolating water. Because the water percolates relatively rapidly, and does not have prolonged contact with the soil matrix, it is reasoned that there is less time for nutrient uptake from the surrounding soil matrix. Hence, concentrations from the pan lysimeters provide an estimate of the lower limit of nutrient loss. A schematic depicting the pan lysimeter is provided in Figure 6 below.

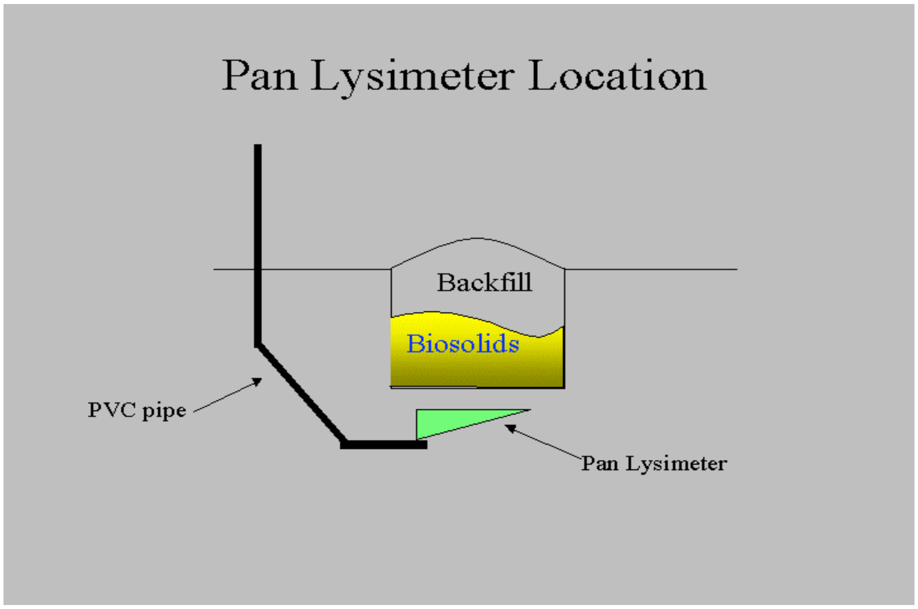

Figure 6. Pan Lysimeter Installation Schematic

Each plot also contains two sets of suction lysimeters installed under and around the biosolids rows. Where water flows a great distance vertically to the water table, nutrients leaving a source generally create plumes that migrate downward. Therefore, one set of suction lysimeters were installed 15 cm, 30 cm, and 60 cm (6, 12, and 24 in.) directly below a biosolids row to monitor long-term migration of any plume in the vertical direction.

The second suction lysimeter nest is located on either side of the row in the soil level with the bottom of the trench. Because this site has a thick clay subsoil layer overlain with gravel and mixed clay loam backfill, lateral flow on top of the horizon interfaces (sometimes referred to as locally perched water) is a possibility. Two suction lysimeters were therefore installed 25 cm and 50 cm (10 and 20 in.) from the side of a row to monitor lateral movement. A schematic of the position of all five suction lysimeters in relation to a biosolids row is presented in Figure 5. Suction lysimeters collect soil water that may contains nutrient levels elevated above that of free flowing sub-surface water. Hence, concentrations provide an estimate of the upper limit of nutrient loss.

Pan lysimeters were installed just after the deep row was filled with biosolids. Suction lysimeters were installed after the trench was filled with biosolids, after the ground was leveled, but before planting. Pan lysimeters were installed from July 2002 through March 2003. Suction lysimeters were installed after the area was leveled and disked. Water quality sampling began in April 2003.

The term “groundwater” will be used to denote water in the zone of saturation (Bear, 1972). More specifically, this is water in the geologic formations that are completely saturated (Freeze and Cherry, 1979). Overall water quality in the groundwater has been assessed by regular measurement from previously installed groundwater monitoring wells already resident in the top of the Nanjemoy formation, which is the first water supply aquifer beneath the site (Wilson and Fleck, 1990).

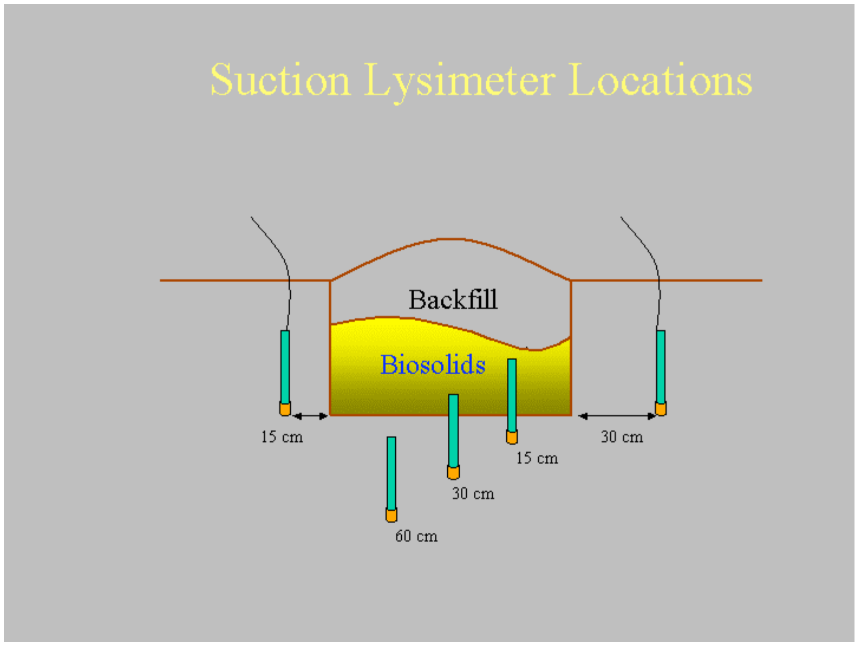

Figure 7. Suction lysimeter installation schematic showing the lysimeters' depths below biosolids 15, 30, and 60 cm (6, 12, and 24 in.) and lateral distances to the biosolids at 15 and 30 cm (6 and 12 inches).

Sampling frequency.

Water samples from pan and suction lysimeters were collected on a monthly basis for the first year. For the following years, samples were collected every other month. These routine collections amount to 7,560 sampling attempts since April 2003. Due to dry weather conditions and other climatic factors, however, there were instances in which water was not present or could not be extracted.

Parameters.

All subsurface water samples have been sampled for pH, nitrate, nitrite, total nitrogen, sulfate, and chloride. At the ERCO site, subsurface water flow is greatly restricted by the clay. This restricted flow provides any aqueous phosphorus with ample opportunity to adsorb onto charged sites, which are plentiful in the clay subsoil. For this reason, ortho-phosphate and total phosphorus was analyzed for the first twelve months, but analysis was discontinued because phosphorus was not detected in these samples.

Results

Outliers

Not all data collected was logical. In Figure 8 below, one lysimeter out of 27 lysimeters with application rates of 4,400 kg N/ha (4,000 lbs. N/ac.) has nitrate values that were orders of magnitude higher than all the others. This lysimeter, SL-1E-2 was installed 15 cm (6 in.) below the biosolids pack. During the installation, biosolids may have gotten into the borehole and been interred with the lysimeter. Hence, these results may reflect biosolids conditions, not soil water 15 cm (6 in.) below the biosolids pack. We don't have a good explaination for why this one lysimeter had such values, but we treated it as an outlier and removed it from all subsequent calculations.

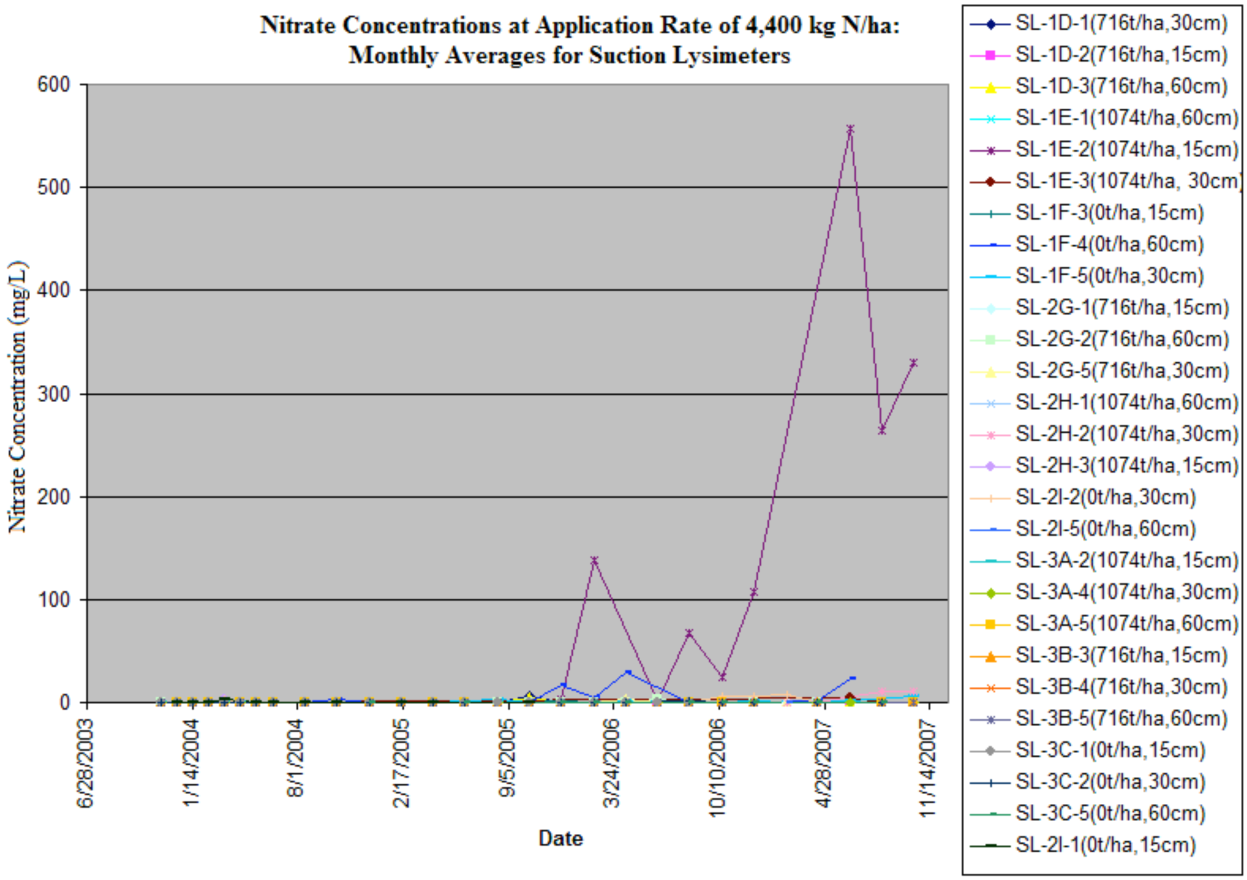

Figure 8. Nitrate concentrations for all suction lysimeter installed in areas that received 4,400 kg N/ha (4,000 lbs. N/ac.) biosolids applications. In the legend, suction lysimeter names are followed by the tree densities (0, 716, and 1,074 trees/ha or 0, 290, and 435 trees/ac.) and lysimeter depth below the biosolids (15, 30, and 60 cm or 6, 12, and 24 in.).

The controls received zero biosolids and had no trees planted on the plots. Weeds took advantage of the treeless areas. Figure 9 represents the nitrate concentrations found in the control plots. In Figure 9 below, there are three values that are much higher than all other values. These values occurred on different days and from different lysimeters. It is possible that samples could have been contaminated. Since they are controls, these three high values were removed from all subsequent calculations.

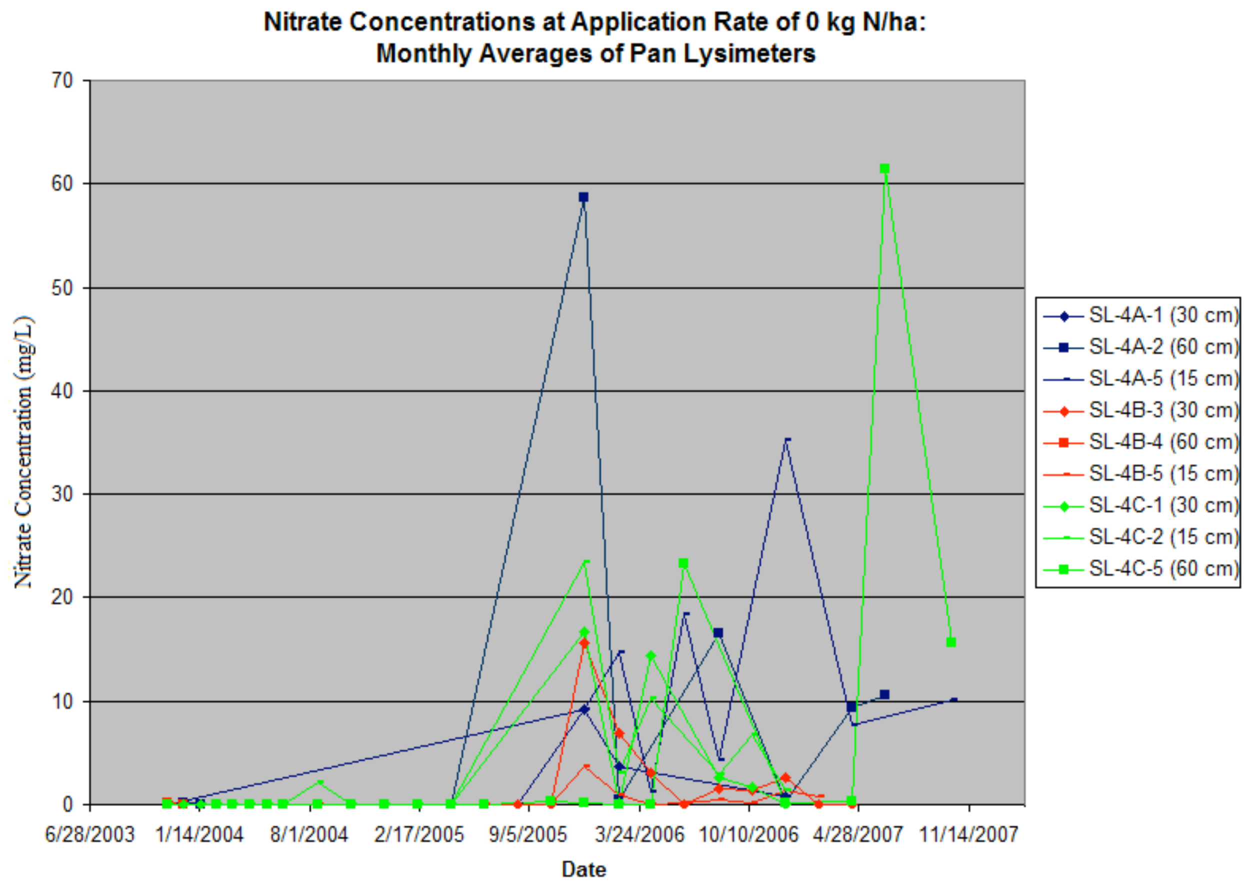

Figure 9. Nitrate concentrations for all suction lysimeter installed in areas that received 0 kg N/ha (0 lbs. N/ac.) biosolids applications. In the legend, suction lysimeter names are followed by lysimeter depth below the biosolids, 15, 30, and 60 cm (6, 12, and 24 in.).

Figure 10 represents the ammonium concentrations found in the control plots. One lysimeter (SL-4A-2) is consistently and significantly higher than the other ammonium values for all other control suction lysimeters. While these are controls and one would at first expect zero ammonium, these plots had had biosolids applied twice before the initiation of this experiment. Hence, some residual biosolids that were either seven or fourteen years old may have been pulled to the surface during reconstruction of the plots and instrumentation installation. If the residual biosolids ended up near the lysimeter, it could generate ammonium that would not represent an untreated area. We treated lysimeter SL-4A-2 as an outlier and removed it from all subsequent calculations. Note that one of the three points in Figure 9 above was also from lysimeter SL-4A-2, which strengthens our suspicion that some contaminant exists near this lysimeter.

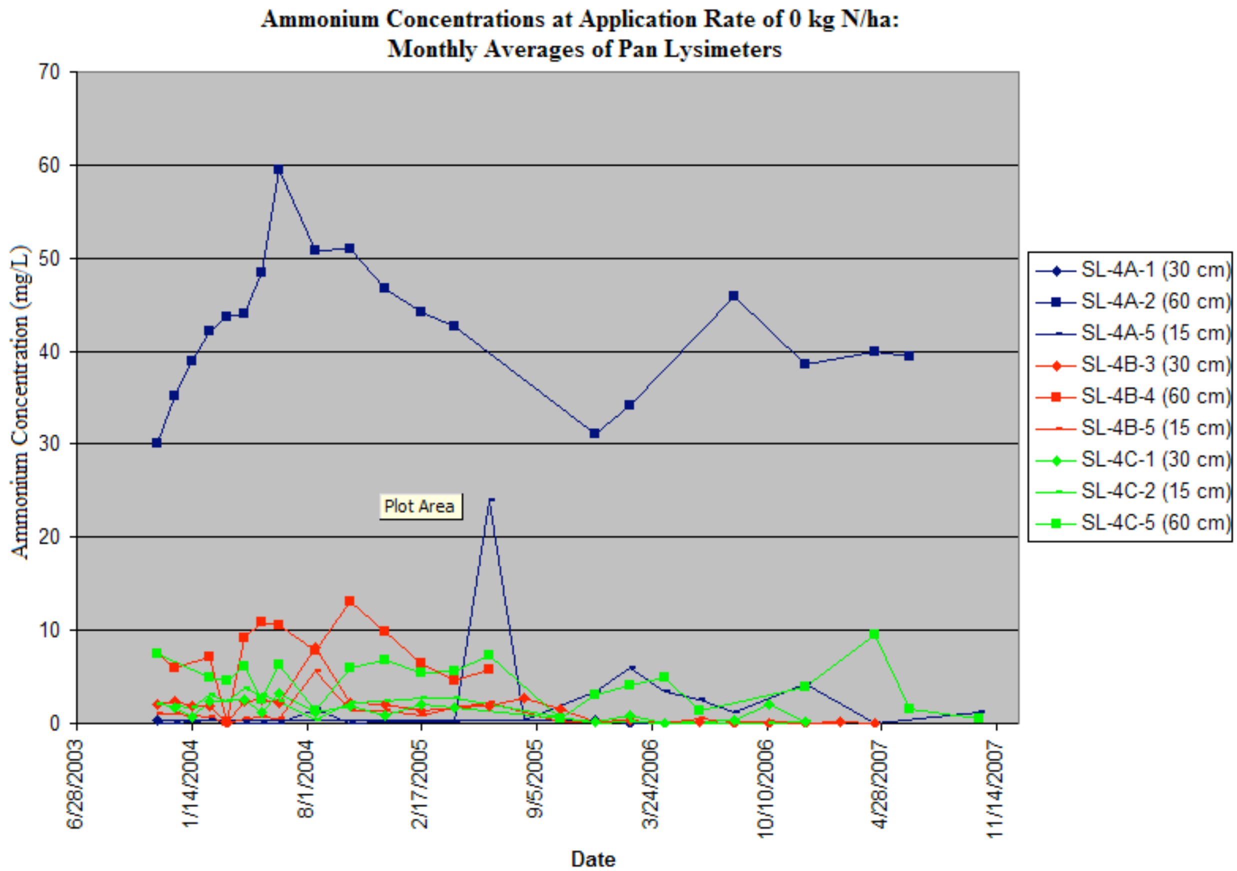

Figure 10. Ammonium concentrations for all suction lysimeter installed in areas that received 0 kg N/ha (0 lbs. N/ac.) biosolids applications. In the legend, suction lysimeter names are followed by lysimeter depth below the biosolids, 15, 30, and 60 cm (6, 12, and 24 in.).

Lysimeter SL-1B-3 located 15 cm (6 in.) laterally from the biosolids pack with an application rate of 8,800 kg N/ha (8,000 lbs. N/ac.) and tree density of 1,074 trees/ha (435 trees/ac.) does not follow the general trends of the other 27 lysimeters with similar properties shown in Figure 11. Also, SL-1B-3 has not produced a sample since December of 2006. For these reasons, SL-1B-3 has been removed from all subsequent calculations.

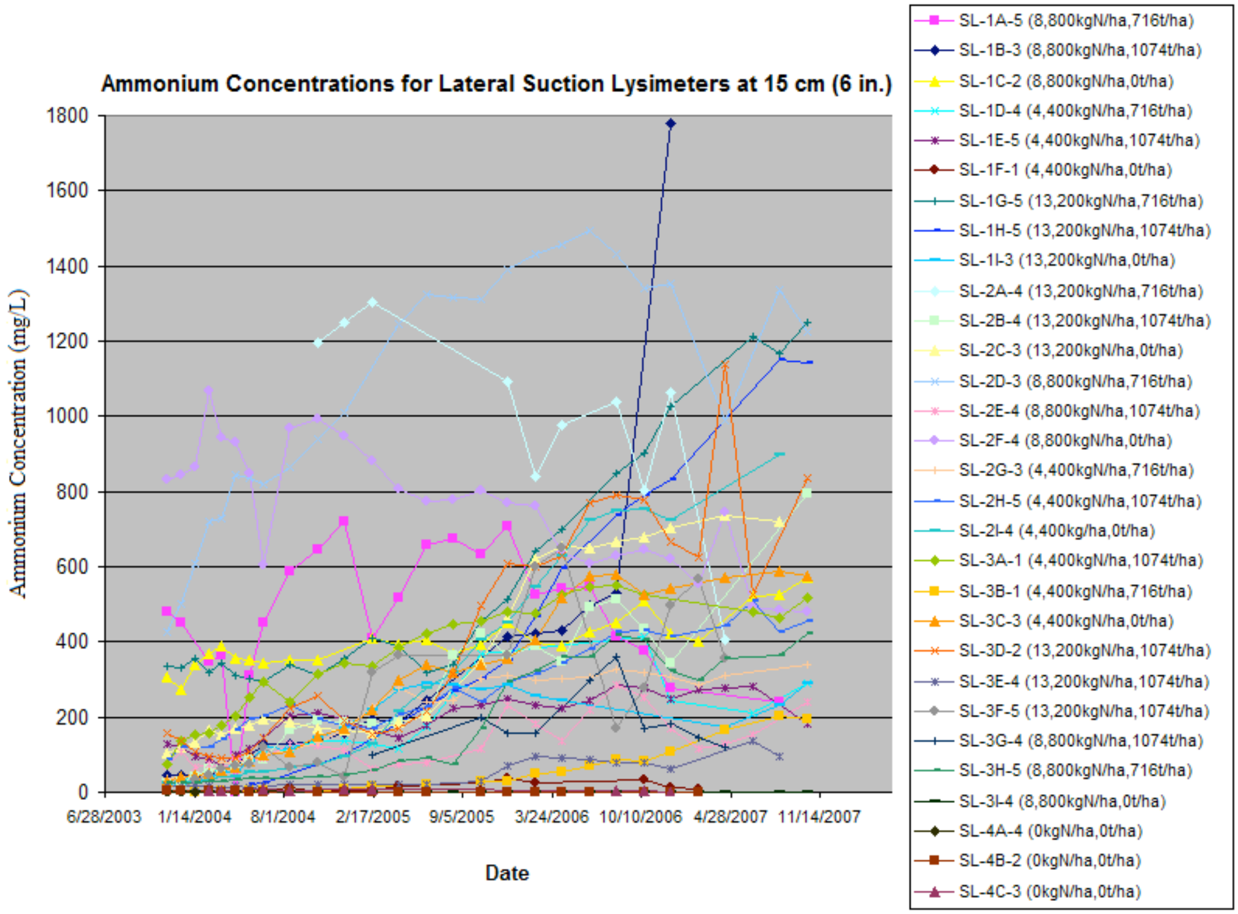

Figure 11. Ammonium concentrations for all suction lysimeters installed 15 cm (6 in.) laterally from the biosolids. In the legend, suction lysimeter names are followed by biosolids application rates (0, 4,400, 8,800, and 13,200 kg N/ha or 0, 4,000, 8,000, and 12,000 lbs. N/ac.) and tree densities (0, 716, and 1,074 trees/ha or 0, 290, and 435 trees/ac.).

Nitrate from the Deep Row Forestry System

The data in Figure 12 represent nitrate concentrations in the pan lysimeters between November 2003 and October 2007. The control data points are the average of three data points and the levels are the average of nine data points. During July 2005, nitrate began to appear in the water sampled by the pan lysimeters. The levels typically have been below 10 mg/L. At this point, variation in the biosolids application rate is not clearly related to the nitrate concentration. The control (no biosolids and no trees) also went up during this time. This suggests that there may be a meteorological change that is affecting the nitrate loss from the surface. However, the control reached levels of 2-3 mg/L while the treatments reached levels of 10 mg/L.

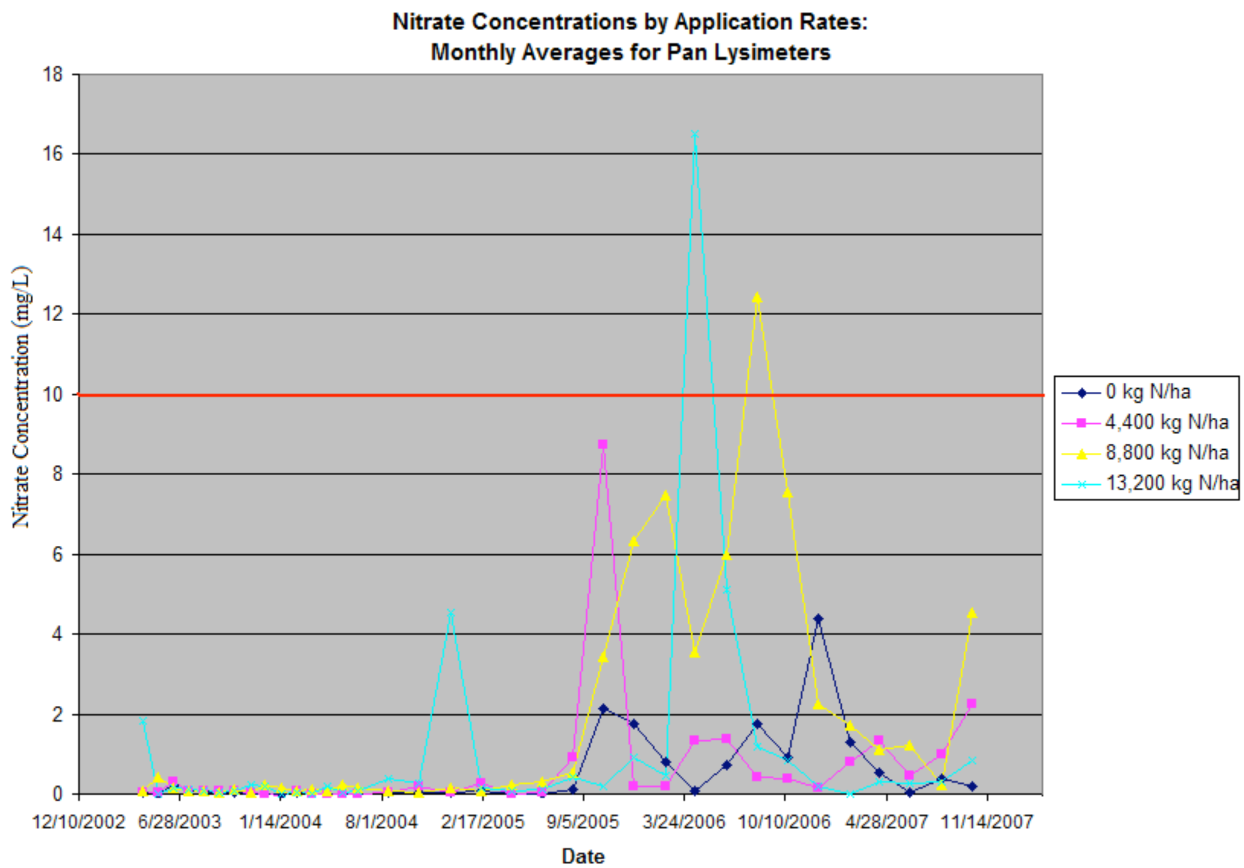

Figure 12. Monthly average nitrate concentration in pan lysimeters from November 2003 to October 2007.

At the same time, nitrate in the soil water, as sampled by the suction lysimeters, increased (Figure 13). The highest value was observed in the control (13.9 mg/L) for the November 2005 sampling. The highest two application rates (13,200 kg N/ha and 8,800 kg N/ha or 12,000 lbs. N/ac. and 8,000 lbs. N/ac.) had the lowest nitrate concentration and the lower two application rates had the highest nitrate concentrations, hovering around 10 mg/L. Between July 2005 and October 2007, all nitrate levels seemed consistent at between 1.0 and 10.0 mg N/L except for the highest application rate, which seems to be starting an increase toward 10 mg N/L from June 2006 to October 2006. The explanation for the lowest application rate increasing first may be that as application rate increases, depth increases. Hence temperature, oxygen, and microbial activity are all delayed in the heavier rates and, as a result, the formation of nitrate may be delayed in the deeper trenches.

Nitrate concentration draining from corn on agronomic soils using fertilizer or biosolids as the N source is significantly (as much as triple) higher than nitrate found beneath the deep row forestry system, even for the highest biosolids application level.

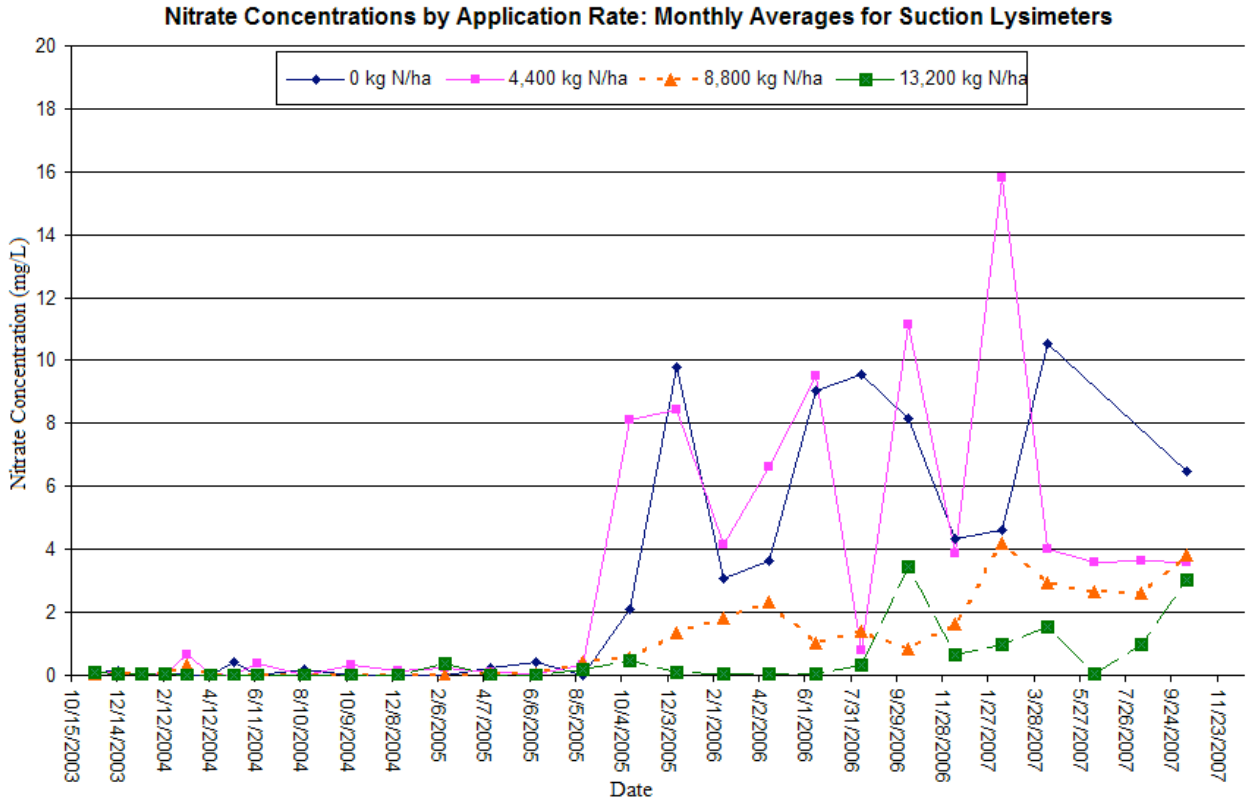

Figure 13. Monthly average nitrate concentrations in suction lysimeters from November 2003 to October 2007 at application rates of 0, 4,400, 8,800, and 13,200 kg N/ha (0, 4,000, 8,000, and 12,000 lbs. N/ac..).

Ammonium in Pan Lysimeters

Ammonium (mg NH_4-N/L) concentrations found in the pan lysimeters are plotted in Figure 14. Clearly, the control was well below the treatments. This was not true for the nitrate values. At about February of 2004, the three application rates had ammonium concentrations of 450-600 mg NH4-N/L. By September 2006, these values had dropped to 175- 320 mg NH_4-N/L. This is clearly a downward trend. The heavy rate of application exhibited a peak from January 2005 through August 2005 that was not exhibited by the other two application rates. This suggests that something in the system is not responding the same in the heaviest rate. A preliminary assessment is that the heaviest rate may not be appropriate.

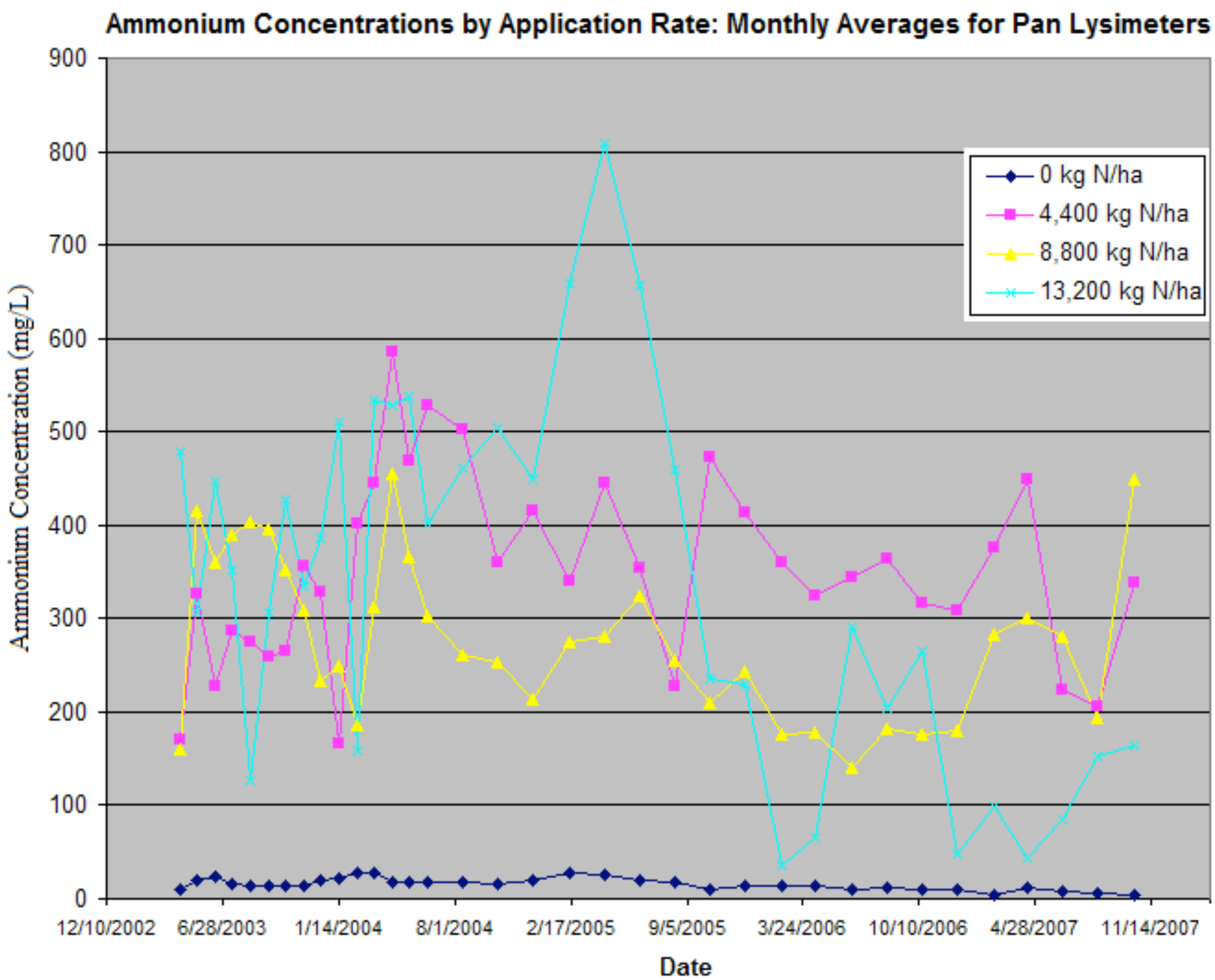

Figure 14. Monthly average ammonium concentration in pan lysimeters from November 2003 to October 2007 at application rates of 0, 4,400, 8,800, and 13,200 kg N/ha (0, 4,000, 8,000, and 12,000 lbs. N/ac..).

Ammonium in Suction Lysimeters

Ammonium (mg NH_4-N/L) concentrations found in the suction lysimeters are plotted in Figure15. Consistently, the control (no biosolids application and no trees planted) had ammonium concentrations in the suction lysimeters just above zero. For the treatments, between October 2003 and December 2004, the ammonium level rose from approximately 400 mg/L to approximately 625 mg/L. This rise was independent of the application rate. Then in 2005 and from there on, the lowest application rate was fairly consistently at a lower concentration level than the other two rates.

There was a jump in December 2004 in the concentration from the middle application rate (8,800 kg N/ha or 8,000 lbs. N/ac..). The concentration rose from approximately 590 mg/L to approximately 775 mg/L. The middle application rate concentration remained elevated above the lowest rate and the highest rate until March 2006. Between August 2005 and August 2006, both the low application rate (4,400 kg N/ha or 4,000 lbs. N/ac..) and the middle application rate (8,800 kg N/ha or 8,000 lbs. N/ac..) dropped to a low in February to April 2006 and then rose to a temporary high in August 2006. This is most likely a seasonal variation. However, the levels dropped by about 100 mg/L over that year. From March 2006 through October 2006, the concentration mirrored the application rate.

Additionally, the increase in concentration has either leveled off or has started to decrease since sometime in mid to late 2005. This was the case for all application rates.

Effects of time may be different for each application rate because the heavier rate is deeper than the lower rate. The lowest rate peaked in August 2005. The middle rate peaked in October 2005, and the highest rate peaked in August 2006. This may be due to the time required for tree roots to grow deeper for the heavier application rates. When the tree roots appear, oxygen is introduced and microbial action may change dramatically.

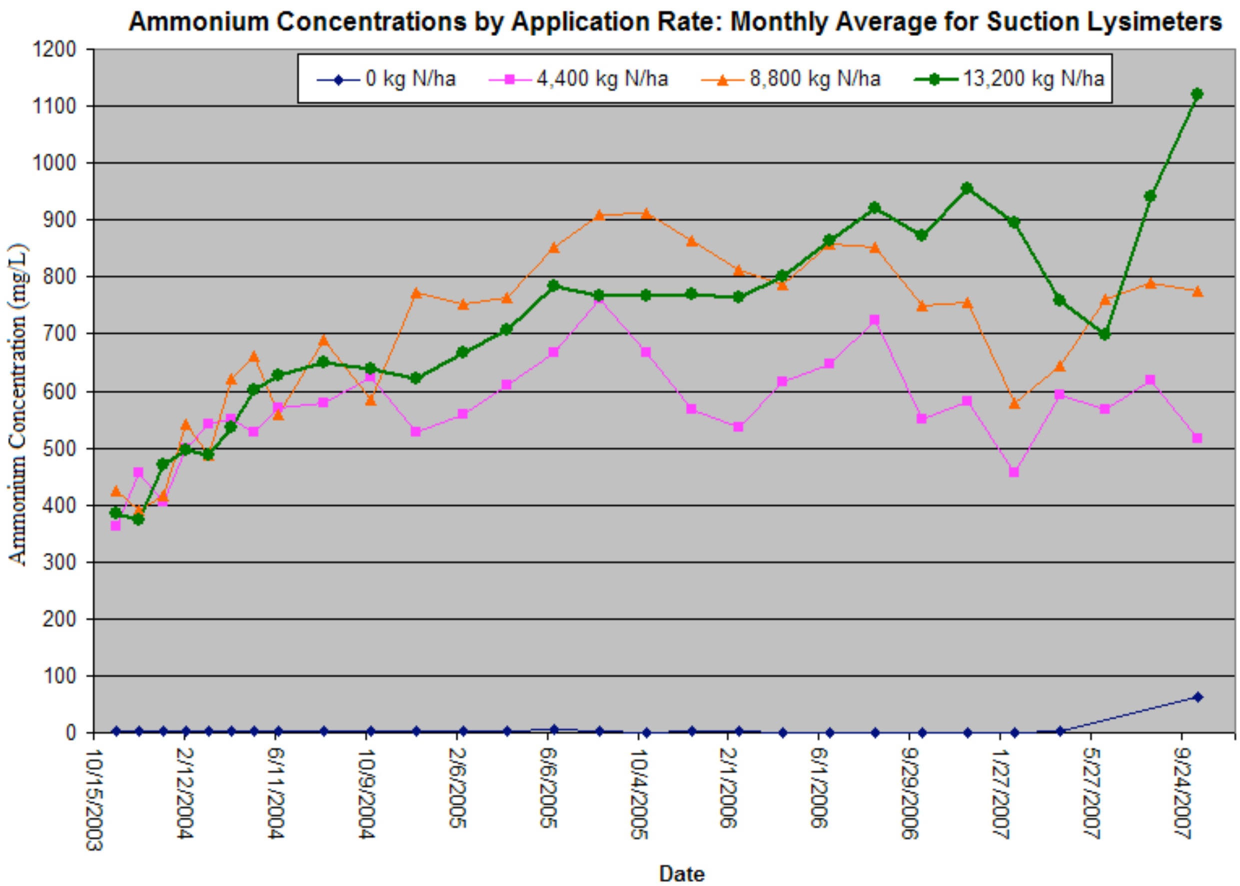

Figure 15. Monthly average ammonium concentration in suction lysimeters from November 2003 to October 2007 at application rates of 0, 4,400, 8,800, and 13,200 kg N/ha (0, 4,000, 8,000, and 12,000 lbs. N/ac..).

Ammonium with depth

Figure 16 represents the effect of depth or the distance ammonium is migrating. Each data point represents 27 observations (3 replicates X 3 application rates X 3 tree densities). The controls (no biosolids and no trees) were not plotted.

First, ammonium is clearly increasing over time. At 15 cm (6 in.) below the trench, the ammonium level has increased from approximately 1000 mg N/L in November 2003 to approximately 2000 mg N/L in November 2005. At the 60 cm (24 in.) depth, the increase was from approximately 175 to 400 mg N/L. The 30 cm (12 in.) depth was uniformly in between. The ammonium levels at the 30 and 60 cm (12 and 24 in.) depth may have leveled off. The ammonium levels at the 15 cm (6 in.) depth may be decreasing. Additional data are necessary to draw any meaningful conclusions.

Because of the findings here, suction lysimeters were installed at 120 cm (48 in.) below the biosolids trench in all plots during summer of 2007. Data collection began in August 2007 for these lysimeters. As with most new installations, it may be six months before the readings have any meaning.

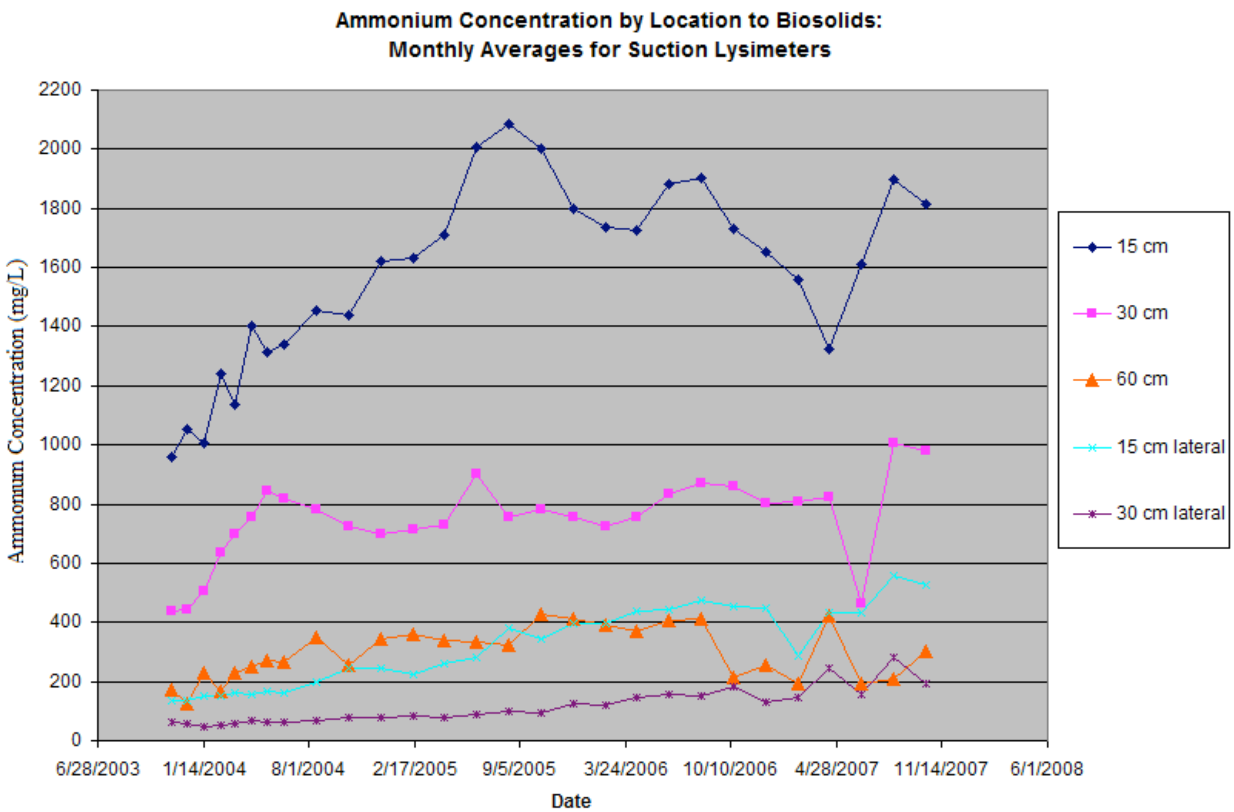

Figure 16. Monthly average ammonium concentration in suction lysimeters, sorted by depth below (vertical) at 15, 30, and 60 cm (6, 12, and 24 in.) and to the side of (lateral) at 15 and 30 cm (6 and 12 in.) the biosolids trench from November 2003 to October 2007.

Orthophosphate

Orthophosphate was monitored heavily during the first 18 months. Of approximately 220 samples, eight samples were above 1.0 mg P/L. There was no trend. It was expected that phosphorus would bind to soil particles.

Anecdotal Observations

As the additional lysimeters were being installed this past summer, the effects on the biosolids that had taken place in the past four years were obvious. Where there were trees, the biosolids were drier, not as thick, and essentially odor-free. In the treatment plots with biosolids and no trees, the biosolids were about the same consistency as the day they went in and the odor of ammonium was very strong. It is clear that the trees are an important moisture sink and are instrumental in removing the ammonium.

Synthesis – Nitrogen Fate and Transport

Hydraulic potential data were not collected in this study. Hydraulic potential data is the component that is necessary to calculate mass flow using Darcy's equation or the Richards equation. In the absence of such data, inferences may still be made about mass transport, based on hydraulic conductivity (Ks).

Typical silt loam soils used in agricultural operations have hydraulic conductivities ranging between 1 x 10^{-4} cm/s and 1 x 10^{-6} cm/s (Fetter, 1994). The sub-surface soils in the plots had hydraulic conductivities averaging 1 x 10^{-6} to 1 x 10^{-5} cm/s. The surface soils had Ks between 1 x 10^{-7} up to 1 x 10^{-3} cm/s but the average was between 1 x 10^{-6} and 1 x 10^{-4} cm/s, depending on the plot (Buswell, 2006). Therefore, the subsoils are one to two orders of magnitude lower than agricultural soils and the surface soils are the same or one order of magnitude less than agricultural sols classified as silt loams or silty clay loams, (sandy agricultural soils have much larger Ks than silt loams, on the order of one to 3 orders of magnitude).

Because the nitrate concentrations in the research plots are lower than published values found for agricultural fields, the nitrate loss (mass transport) logically will be less than lost from a typical corn crop. Because the hydraulic conductivity in the research plots is the same to two orders of magnitude less than agricultural soils, the mass transport will be the same order of magnitude as a corn crop to two orders of magnitude less than a corn crop.

There is an additional benefit from the slowing of subsurface flow caused by the relatively low hydraulic conductivities found in the research plots. Nitrogen that enters the soil water stays in the upper layers of the profile for a longer time (orders of magnitude longer). Hence, there is a much greater opportunity time for uptake by plant roots and denitrification by microbes to occur. This is the logical reason that the system does not lose nitrogen in large amounts.

Conclusions

- For the first three years, zero nitrate left the deep-row tree system.
- Ammonium was immediately released into the soil surrounding the biosolids.
- Ammonium concentrations decrease dramatically with distance from the biosolids, falling from 2100 mg N/L at 15 cm (6 in.) from the biosolids to 400 mg N/L at 60 cm (24 in.) from the biosolids.
- Overall, nitrate concentrations are lower than those found beneath corn crops while utilizing a great deal higher application of nitrogen.

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