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The Role of Forest Soils in a Northern New England Effluent Management System

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THE ROLE OF FOREST SOILS IN A NORTHERN NEW ENGLAND EFFLUENT MANAGEMENT SYSTEM

By

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The Carrabassett Valley Sanitary District in Carrabassett Valley, Maine has utilized both a forest spray irrigation system and a Snowfluent™ system for the treatment of their wastewater effluent. This study was designed to evaluate potential changes in soil properties after approximately 20 years of treatment in the forested spray irrigation site and three years of treatment in the field Snowfluent™ site. In addition, grass yield and composition were evaluated on the field study sites. After treatment with effluent or Snowfluent™, soils showed an increase in soil exchangeable Ca, Mg, Na, and K, base saturation, and pH. While most constituents were higher in treated soils, available P was lower in treated soils compared to the controls. This difference was attributed to higher rates of P mineralization from soil organic matter due to an irrigation effect of the treatment, depleting available P pools despite the P addition with the treatment. Most of the differences due to treatment were greatest at the surface and diminished with depth.

Depth patterns in soil properties mostly reflected the decreasing influence of organic matter and its decomposition products with depth as evidenced by significantly

higher total C in the surface compared to lower horizons. There were decreasing concentrations of total N, and exchangeable or extractable Ca, Mg, Na, K, Mn, Zn, and P with depth. In addition, there was decreasing BS with depth, driven primarily by declining exchangeable Ca and Mg.

Irrigation with Snowfluent™ altered the chemical composition of the grass on the site. All element concentrations were significantly higher in the grass foliage except for Ca. The differences were attributed to the additional nutrients and moisture derived from the Snowfluent™.

The use of forest spray irrigation and Snowfluent™ as a wastewater treatment strategy appears to work well. The soil and vegetation were able to retain most of the applied nutrients, and do not appear to be moving toward saturation. Vegetation management may be a key tool for managing nutrient accumulation on the grass sites as the system ages.

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INTRODUCTION

For many communities wastewater treatment is becoming an increasingly important problem due to expanding industrialization, population growth and increasingly stringent legislation requiring a higher standard for wastewater treatment. Since the enactment of the Water Quality Act of 1972, communities have been forced to assess the impacts that waste disposal has on the environment. The unregulated discharge of wastewater into streams, lakes and oceans is no longer acceptable, due to the degradation of aquatic environments. Subsequently, wastewater must be treated before it can be discharged into the environment. Conventional wastewater treatment requires primary and secondary treatment, and more recently tertiary chemical treatments or filtration, an extensive and often costly process. Currently, land application of wastewater through spray irrigation has helped to deter some of the cost of tertiary treatment. Wastewater irrigation provides nutrients and water that can be taken up by plants and soils and benefit plant growth. In addition, wastewater can recharge groundwater supplies after it has filtered through the soil (Richenderfer et al., 1975). Research has shown that spray irrigation of wastewater effluent has been a successful method of wastewater treatment (Sopper, 1986, Struchtemeyer and David, 1980, Wagner, 1995).

For many regions, cold climates and varied topography can limit the use of land applications of effluent during the colder months. In response to this limitation, an alternative system called Snowfluent™ was developed by Delta Engineering of Ottawa, Canada. The Snowfluent™ process involves making snow from wastewater effluent (the

final step in the treatment process) with equipment similar to that used by ski areas. Snowfluent™ is a direct treatment method. The only pretreatment required for the influent is a short period of storage to allow the settling of the heavier solids. The wastewater is then emitted from a high-pressure snow gun. Chemical changes in the snowpack facilitate the elimination of some nutrients such as excess nitrogen (N), while soil and vegetation are able to absorb the other nutrients (White and Frere, 1994).

The Carrabassett Valley Sanitary District (CVSD) in Maine is utilizing the Snowfluent™ process. CVSD provides wastewater disposal services to the Sugarloaf Mountain ski resort and surrounding communities. It received over 50,000,000 gallons of wastewater per year during the early 1990's, much of which was produced during the winter months. Before the establishment of the Snowfluent™ system the wastewater had to be stored over winter, and then spray irrigated in the summer. Since the district's seven storage lagoons were near full capacity, CVSD decided to explore other options. In February and March of 1994, a temporary Snowfluent™ plant was installed at CVSD to assess the feasibility of the system. Testing by the Maine Department of Environmental Protection (DEP) through an independent laboratory indicated no significant environmental hazards (White and Frere, 1994). Hence, a permanent system was installed and was operational in January 1995.

If proven environmentally sound on a continuing basis, the Snowfluent™ process has the potential to help many communities in colder climates. In conjunction with conventional spray irrigation, storage volumes required for wastewater would be significantly reduced, since the process would allow for year-round effluent treatment.

This is especially important for seasonal resort areas such as ski resorts, where wastewater volumes are highest in the winter. Also, the cost of operation and implementation of frozen effluent systems is relatively low compared to the cost of continually expanding storage lagoons.

The economic advantages of a year-round effluent treatment system must be balanced with environmental quality concerns. Preliminary testing indicated that Snowfluent™ could be successful in treating the effluent (Delta Engineering, unpublished data). Analysis of the data collected in the initial two years of the Snowfluent™ site at CVSD by DEP and Delta Engineering indicated that groundwater and soil quality has not been significantly impacted by the system. Because the system was newly established, little information existed on the biogeochemical responses of the ecosystem to these treatments.

The purpose of this study was to better understand how the Snowfluent™ process performs, and to begin to assess how soils are responding to the frozen effluent system after several years of treatment. In addition, soils from the adjacent conventional spray irrigation site, which have been treated with effluent for approximately 20 years, were assessed to provide insight into the long-term effects of conventional effluent irrigation.

The primary objectives of this research were to examine the influences of effluent spray irrigation and Snowfluent™, soil drainage class and vegetative cover on soil inorganic properties.

Chapter 1

LITERATURE REVIEW

History

Public sanitation has been a social and environmental issue dating back as far as ancient Greek and Roman times. Most of the early developments in land treatment occurred in Britain in the mid- 1800's. The installation of sewers began in Britain around 1850 to help alleviate the pollution of surface waters. In the United States, land treatment was considered the most effective method for waste disposal from 1890 to 1905. After this period the practice came to be regarded as only useful for areas in the southwestern U.S. due to the warmer climate, and it wasn't until the 1970's that it was again regarded as a viable option for wastewater treatment. Starting in the early 1950's, the first major wastewater treatment legislation was passed in the U.S. This act (Water Pollution Act, P.L. 84-660) came about in response to concerns for freshwater eutrophication that arose as partially treated wastewater was discharged into waterways (USEPA, 1979). Since then, this act has been amended many times. A significant change occurred in 1972, with the passage of P.L. 92-500. This modification showed that concern for water pollution was becoming a national priority, and reflected the desire of the nation to control water pollution quickly and efficiently. The Clean Water Act required municipalities to treat their wastewater prior to discharging into the environment. One of the technologies proposed at this time was land application of wastes (USEPA, 1984).

General Considerations

Wastewater treatment involves a series of steps. The wastewater generated from residential areas and industry is brought to a wastewater treatment plant. The first step in the process is primary treatment where the solids that will readily settle out are removed. Next is the secondary stage, often called the biological stage because it includes activated sludge systems and attached growth systems such as trickling filters. The tertiary treatment is the most advanced, and includes such processes as chemical precipitation and filtration (USEPA, 1984).

Traditionally, there have been five major sludge use/disposal options: land application, distribution and marketing of sludge products, landfilling, incineration, and ocean disposal (USEPA, 1984). There are many factors to be considered when land application is used. These include the land application process, preapplication treatment, land suitability, distribution techniques, climatic factors, storage availability, surface runoff control, public health considerations, and monitoring requirements (Knezak and Miller, 1978).

One of the difficulties in assessing the success of land application of wastewater is the tremendous variability of the wastewater itself. The characteristics can fluctuate depending on the initial wastewater composition and the types of treatment processes utilized. Site characteristics are also extremely variable: climate, topography, textures and composition of soil all must be considered when evaluating the effectiveness of land application of wastewater.

Effects on Soil

Many studies have been conducted to assess the feasibility of land application of municipal effluent. Several studies on soil physical properties in response to wastewater treatment revealed a decrease in bulk density, due to a dilution effect caused by the addition of organic matter and a mixing with a denser fraction of soil (Khaleel et al., 1981, Mathan, 1994). Mathan (1994) also concluded the longer the period of irrigation with sewage effluent within limits, the lower the bulk density values. Water holding capacity of the soil was increased with wastewater applications. This was attributed to two factors: (i) the increases in total pore space resulting from increased aggregation, and (ii) as a result of decreased bulk density, the pore size distribution was altered and the relative number of small pores increased, especially for coarse textured soils (Khaleel et al., 1981, Mathan, 1994). Soil pH also tended to increase due to the alkalinity of wastewater in some studies (Stewart et al., 1989, David and Struchtemeyer, 1980).

The influence of effluent treatments on soil exchangeable base cations (K, Ca, Mg, Na) varies depending on site characteristics. A study done on forest soils in Pennsylvania irrigated with municipal sewage effluent for nine years showed significant increases in Ca and Mg in the soil up to 30.5cm depths, possibly due to the strong adsorptive capacity soil colloids exhibit for cations (Richenderfer et al., 1975). David and Struchtemeyer (1980) reported similar results for changes in soil Ca on forest soils in Maine. A four-year study in Australia on medium textured floodplain soils also showed increases in exchangeable Ca and Mg with effluent application (Stewart et al., 1990). Despite increases in soil Ca and Mg in these studies there were no indications that

treatment with sewage effluent was having significant detrimental effects on the soil and vegetation.

A study in Georgia conducted on a sandy clay loam fertilized with liquid sewage sludge, then seeded with rye, showed a depletion in Ca and Mg as compared to unfertilized soils (King and Morris, 1973). Since the applied sludge had a high NO_3-N content, the loss of Ca was attributed both to increased Ca mobility due to the presence of NO_3-N, and low pH. The loss of Mg was attributed to leaching due to low pH as well. It was recommended that dolomitic limestone would have to be administered to maintain productivity of the soils.

The addition of wastewater effluent caused increases in the concentration of soil exchangeable Na in several studies. A study in Australia in which seven tree species were sprinkler irrigated with effluent at an annual rate of 1191-1752mm increased exchangeable Na percentage from 3.2% to 9.8% after 4 years of irrigation (Stewart et al., 1990). At this rate of increase, the soil is becoming sodic, which could eventually impair soil physical properties and necessitate remedial action. Similar increases in soil exchangeable Na were found in several other studies, involving both forest and agricultural soils (King and Morris, 1973, Richenderfer et al., 1975, Mancino and Pepper, 1992, David and Struchtemeyer, 1980, Neilson et al., 1991). Sodium accumulation was most often mentioned as the reason for remedial strategies in effluent treated soil systems. Mancino and Kopec (1989) found that calcium sulfate applied annually at a rate of 2240 kg ha^{-1} could be effective in reducing soil Na levels through cation exchange by

approximately 200 to 300 mg kg^{-1} in soils irrigated with effluent. In addition, the SO_4^-S in the wastewater may also help to control Na levels.

The effects of effluent irrigation on exchangeable K appear small and inconsistent in the literature. Stewart et al., (1990), after four years of effluent irrigation of tree crops in Australia, found no significant increases in K to a soil depth of 55 cm. At depths below 100 cm, the soils showed a significant decrease in K. In a study on forest soils in Maine, David and Struchtemeyer (1980) showed significantly higher levels of exchangeable K in both the O horizon and at depths of 0-20 cm in the sprayed group as compared to control soils. Sampling in the second year of their study showed no significant differences due to treatment, although there were numerical increases between sprayed and control sites. Depths of 20-40 cm showed no significant differences in either year. Total K in the O horizon was found to decrease in the sprayed group compared with the unsprayed group. Richenderfer et al., (1975) did not find significant and consistent changes in exchangeable K after 9 years of irrigation with municipal effluent on forest soils in Pennsylvania.

The effects of effluent irrigation on total N accumulation in the soil profile were not statistically significant after nine years of irrigation in the Pennsylvania study (Richenderfer et al., 1975). Researchers in Maine, Florida, Georgia and Michigan have also reported no accumulations of N from wastewater effluent irrigation in soils (David and Struchtemeyer, 1980; Wagner, 1995; Barnett and Arnold, 1986; Nutter, 1986; Burton and Hook, 1979; Hook and Kardos, 1978).

If N is not accumulating in the soil, it must either be lost to the atmosphere in the gaseous phase, leached from the system or taken up by vegetation. A study on the effects of land application of wastewater on denitrification rates revealed that denitrification was an effective method for removing N, especially for soils that have limited renovation capacity (Monnett et al., 1995). Significant denitrification is helpful in reducing the chance of NO_3-N leaching from surface soils and subsequent contamination of groundwater. To maximize this conversion of NO_3-N to N_2, split application (2 or 3 times a day) of wastes was recommended by the authors of this study. Hauck (1984a) reported denitrification was responsible for the loss of an average of 30% of the NO_3-N found in soils.

Studies of groundwater contamination with N as a result of effluent irrigation show varied results. A study by Reed and Crites (1986) on three forested sites in New England did not find any significant increases in total N levels in groundwater following effluent irrigation for several years, but the authors recommended a tree harvest program at some point in the future to sustain high N removal. Sopper (1986) found slightly elevated concentrations of NO_3-N in the groundwater after several years of effluent irrigation, but levels were still well below the EPA Safe Drinking Water Standards of 10 mg l^{-1}. In addition, Sopper reported that soil type and vegetation influence the extent to which leaching occurs. Examination of three different forest ecosystems at the Penn State facility showed sandier soils had lower N retention than did finer textured soils. In addition, the site with invading pioneer species of herbaceous and tree vegetation was much more effective at renovating wastewater than the sites with mature pine stands.

Another study in Australia showed leaching of N to be insignificant (Polglase et al., 1995). They showed that the weeds that were allowed to grow around the trees acted as a buffer against leaching. The importance of leaving vegetation around the trees (as opposed to plots with no vegetation around the trees) was also emphasized in a study by Smethurst and Nambier (1989). They found the amount of N leached was threefold greater when weeds were excluded.

Stewart et al., (1990) reported that 29% of the N applied in effluent irrigation was accumulated in the biomass of trees in the study plot, and irrigation had no measurable effect on total soil N. Assuming no volatilization, the remaining 71% would have leached through the soil and into groundwater. A similar finding was reported by Hook and Kardos in 1978, who were studying effluent spray irrigation in a mixed hardwood and conifer forest in Michigan. The assumption of no N lost through volatilization may not be valid. Such losses may have occurred because conditions conducive to volatilization, such as sprinkler irrigation and alkaline effluent with high concentrations of ammonium, were experienced throughout the study. Therefore, leaching losses may not have been as high as 71%, but the loss of N as NO_3-N to groundwater represents a real risk to environmental quality. Further research is warranted to determine an irrigation schedule that would minimize leaching and maximize N accumulation in the biomass.

Vegetation is an important sink for applied N, and the uptake of N depends on vegetation characteristics and growth conditions. Sopper and Kardos (1973) determined that at a 5 cm per week level of effluent irrigation, two varieties of corn removed 166 and

179 kg N ha^{-1} respectively, and reed canary grass removed 457 kg N ha^{-1}. They also calculated the efficiency of crops as renovation agents, expressed as the ratio of the weight of the nutrient removed in the harvested crop to the weight of the nutrient applied in the wastewater. For both corn varieties and reed canary grass, the renovation efficiencies were more than 100 percent of the applied N. It is also important to schedule the addition of N to coincide with the period of active vegetative growth and uptake. As growth rates decrease, plants take up less N, leading to either an accumulation of N in the soil or leaching (Feigin et al., 1984).

A study by Kim and Burger (1997) on a mature, upland, hardwood forest in the central Appalachians reported that their forest had a very low wastewater bioremediation capacity. The N input into the forest was increased by as much as six to ten times the atmospheric N inputs, but N retained in aboveground vegetation showed no treatment effect compared to a control site. This site showed that N leaching was the dominant form of N loss from the system, apparently caused by enhanced nitrification rate and limited NO_3-N storage in forest vegetation.

Jordan et al., (1997) conducted a study in which three different ecosystem types were irrigated with effluent: a successional pitch pine woodland 26 years in age, a mixed oak-pine forest >70 years in age, and cleared areas revegetated with grasses and old-field weeds. The authors concluded that the amount of N that forest ecosystems can retain is limited. Although much of the N was retained in the soils during the early stages of effluent irrigation in their study, storage in wood was an important mechanism for long-term N retention. As trees approached maturity, their growth rates and subsequent N

uptake decreased. They concluded that eventually whole tree harvesting would become necessary to remove excess N from the system. In contrast, old field and perennial grasses can be effective at removing N from wastewater. Up to 88% of the N applied to Reed Canary grass at loadings as high as 443 kg $ha^{-1}yr^{-1}$ was either retained or removed by harvesting (Vaccaro et al., 1979). Mowing without harvesting was found to be effective for several years (Burton, 1978), but removal would eventually have to be performed to prevent excessive N release from decomposing litter.

The fate of phosphorus (P) has also been the focus of research involving effluent irrigation, since excess P can lead to eutrophication. A study at Sugarloaf Mountain in Maine found increased concentrations of soil available P in an effluent irrigated mixed hardwood forested site when compared to the control, but only at depths shallower than 20 cm below the soil surface (David and Struchtemeyer, 1980). They suggested that soil was contributing to the removal of P from the effluent. Similarly, in Pennsylvania, a study found no significant change in the concentration of P below 30 cm depth, suggesting that the upper soil profile was effectively removing P (Richenderfer et al., 1975). Another study involving marginal soils (shallow depth to restrictive layers) reported low concentrations of PO_4-P in subsurface waters, and no statistical difference between control and irrigated plots (Monnett et al., 1996). At 60 cm depths, the PO_4-P removals from applied effluents were between 96% and 98%, suggesting high adsorption of P to soil particles. Hook et al., (1973) found that soils differed in their ability to adsorb P from effluents. In heavy textured soils high in sesquioxides, P from effluent irrigation did not increase in the soil below a depth of 30 cm after 7 years of irrigation.

In contrast, the P content of soils increased to a depth of 91 cm after 6 years of treatment in coarse textured soils that had lower concentrations of sesquioxides. A study on sandy soils in a sweet cherry (*Prunus avium* L.) orchard in Canada found elevated concentrations of soil P after effluent irrigation at the deepest soil sampled, suggesting that these soils had a limited ability to sorb P, and P leaching occurred (Neilson et al., 1991).

Effect on Vegetation

An important decision is whether the disposal site should be forested or agricultural land. Since the disposal of sewage effluent will result in the uptake of nutrients and possibly toxic materials by the vegetation, different factors need to be taken into account when assessing success or failure of the irrigation. Several studies conducted in a northern hardwood forest in Ontario, Canada found that heavy leachate loadings were responsible for adverse impacts on forest microbial communities (McBride et al., 1989, Gordon et al., 1988). McBride concluded that transformations in soil water and soil morphology due to the spray irrigation of leachate was leading to a rapid decline in forest health due to excess soil moisture and micronutrient imbalance.

Studies in Australia showed no decline in biomass production of several tree species, with increased yields for Flooded Gum (*Eucalyptus grandis*) and Sydney Bluegum (*Eucalyptus saligna*) in response to effluent irrigation (Stewart et al., 1990). They found that the biomass that accumulated in 4 years under wastewater irrigation was similar to 8 year biomass accumulation on sites that received no irrigation. Similar results were found in southwestern Wisconsin, where diameter growth of Jack pine

(*Pinus banksiana*) was significantly increased after two years of spraying with sewage effluent. The Wisconsin researchers suggested spray irrigation may have worthwhile benefits for increased wood production (Tolsted, 1976).

In Maine, a study on sugar maple (*Acer sacharum* Marsh) foliage revealed significant increases in N, Ca, Mg, and P, and a significant decrease in Mn in the effluent sprayed group compared to the control (David and Struchtemeyer, 1980). However, many of the nutrients taken up by a tree are redeposited as leaf litter annually, rather than being removed as is the case for agronomic crops. Therefore, trees may not be as efficient at purification of the percolating effluent over the long-term.

Other potential benefits and problems exist on agricultural land. A long-term study on two agricultural sites, one in Lubbock, Texas, the other in Bakersfield, California showed no significant detrimental effects of effluent irrigation to crops after 35 years of municipal effluent application (Hinesly et al., 1978). In Bakersfield, the primary crop was barley. Low Iron (Fe) concentration in plant tissue was the only difference between the wastewater irrigated plots and the well water irrigated plots. This may be a result of the increased pH of the wastewater irrigated plots, resulting in lower availability of Fe to plants (Thompson and Troeh, 1978). In Lubbock, the primary crop was corn. Leaf Fe and Na concentrations were slightly lower in the wastewater irrigated plots, and higher in Boron (B). Although the concentrations of B were in an acceptable range, it was suggested that plants may suffer from B toxicity if concentrations continued to increase (Hinesly et al., 1978).

Another study conducted in France also found municipal wastewater irrigation beneficial for agronomic yields, but cautioned that B levels must be monitored in the wastewater if B-sensitive crops such as potatoes were planted (Bunel et al., 1995). In addition, this study found that some thermotolerant bacteria were present on the plants, which means that wastewater irrigation could not be used on plants that are to be eaten raw.

A study in southern Portugal used municipal wastewater to irrigate corn. Although the addition of wastewater increased yields, the study cautioned that later in the season, as plant growth was declining, N was accumulating in the soil. This shows a need to consider the capacity of plants to take up nutrients over the growing season, and adjusting wastewater application rates accordingly (Vazquez-Montiel et al., 1996).

<u>Health Concerns</u>

Potential health concern is the hazard from enteric microorganisms. In general, it has been observed that microbial segregation occurs during conventional wastewater treatment, so that bacteria and viruses, as well as the heavier eggs of certain parasites, tend to become associated with sludge, but a significant amount of parasitic cysts and ova may exit with the liquid effluent (Sorber and Moore, 1986). A study conducted in the southwestern U.S. found no measurable increase in total aerobic bacteria in soils irrigated with effluent as compared to controls (Mancino and Pepper, 1992). Another study suggested the use of chlorination as a method to reduce the health hazard associated with enteric microorganisms (Monnett et al., 1996). A study conducted in a forest clear-cut in Washington determined that in soils fertilized with sewage sludge (as opposed to

wastewater) bacteria can survive (Edmonds, 1976). Survival rates were influenced by sunlight, temperature, moisture, organic matter, and the presence of competitive organisms. Few bacteria were recorded in groundwater, suggesting the removal of bacteria was achieved via percolation through the soil and sedimentation on soil grain surfaces. However, stormwater runoff, or direct contact with the sludge, could pose a potential health hazard.

Another study by Hickey and Reist (1975) summarized the literature on the health significance of spray irrigation aerosolizing effects on pathogens, and showed that there is significant evidence for the spread of pathogenic aerosols through aerated spray irrigation. They discovered bacterial recoveries immediately downwind from the source ranging from 320-30,700 viable particles/m^3, as opposed to 0-1100/m^3 at 91.4 m downwind. Therefore, distance to groundwater, the existence of confining strata, and buffers around the application site are important for preventing contamination since they reduce the efficacy of aerosol transport over long distances.

Effect on Wildlife

Another aspect of wastewater irrigation to be considered is the effect it has on wildlife. In general, wildlife collected from effluent irrigated forests have shown little evidence of higher accumulations of trace metals (Urie, 1986). A study on white footed mice in the Penn State Wastewater Project showed that although lead (Pb) and cadmium (Cd) concentrations were higher in tissues of mice inhabiting spray irrigated sites, the concentrations were not high enough to be considered toxic. Heavy metals were not accumulating in meadow voles inhabiting a reed canary grass field irrigated with sewage

effluent in Pennsylvania. This was attributed to plant uptake of the heavy metals and subsequent harvest of the crop, as well as to the voles' ability to excrete the metal contaminates derived from dietary intake (Taminga, 1995).

Irrigation with effluent can also cause changes in vegetation, which can either benefit or detract from habitat value for a specific wildlife species. For example, a study by Anthony and Wood (1979) found songbird populations to be less diverse on sewage irrigated lands, while forage production for deer and rabbits improved in irrigated aspen, pine and shrub types. Dressler and Wood (1976) found that the spraying of effluent did not significantly deter deer from using the irrigated site as forage.

Conclusion

A review of the scientific literature indicates that land application of effluent has had primarily favorable results. In order for land applications to be more fully utilized, careful planning of land application systems must be employed. Site characteristics such as climate, topography, texture and soil composition, as well as the type of vegetation that will be used at the site and its management, all must all be considered. In addition, careful monitoring of the site must be done, with adjustments to loading rates and time of application performed when necessary. With careful management, land application appears to provide an environmentally sound method of managing wastewater effluent.

Chapter 2

MATERIALS AND METHODS

A field experiment was carried out in 1996 at the Carrabassett Valley Sanitary District in Carrabassett Valley, Maine. The study site was located 1.6 km north of the junction of Sugarloaf Mountain Ski Resort in Township 4, Range 2 (45° 2' N, 70°19' W). The soils throughout the study site were primarily coarse loamy, mixed, frigid aquic Haplorthods from the soil series Marlow, Dixfield and Colonel.

The wastewater treatment system consisted of seven lagoons with a total storage volume of approximately 151,400 m^3. The first lagoon in the series was aerated, and provided the majority of the treatment. The remaining lagoons were used for storage.

The irrigation area was broken up into three distinct spray fields, identified as the East, West, and North fields. The East and West fields were installed in 1973, and covered an approximate area of 14.86 hectares, with 59 sprinkler heads in the West field and 44 in the East field. The North field was constructed in 1985, consisting of 101 sprinkler heads on approximately 9.3 hectares. The irrigation network was all above ground, with sprinkler heads located at intervals approximately 24 to 27 meters apart (Brutsaert and Anderson, 1995).

The primary vegetation covering the spray area was a mixed hardwood forest with a minor component of conifers. The understory included sugar maple (*Acer saccharum* Marsh.), striped maple (*A. pennsylvanicum* L.) and balsam fir (*Abies balsamea* (L.) Mill.). Some brush cutting had been performed along the irrigation

network.

Adjacent to the conventional effluent irrigation site was an 81.5 hectare site for the Snowfluent™ system. The site had slopes from 3% to 15%. The treatment area was clearcut prior to the installation of the temporary system in 1994, and had several berms to prevent surface runoff. The site had been planted with fescue-northern grasses (Festuca spp.).

To characterize the effluent, historical data from 1996 and 1997 (Table 1) were obtained from the CVSD for ammonia (NH_4N) and nitrate (NO_3-N), biological oxygen demand (BOD), phosphorus (P), settleable solids, total kjeldahl nitrogen (TKN), total suspended solids (TSS) and pH. These data were collected monthly.

Soil sampling was conducted in August and September 1996, after the snowpack had melted. The Snowfluent™ system had been in operation for three years, and the forest spray irrigation system for approximately twenty years. Due to limitations of resources, only one intensive soil sampling was conducted.

This study utilized soil samples from four compartments; grass field control without effluent irrigation (GC), grass field treated with Snowfluent™ (GE), forest control without effluent irrigation (FC), and forest treated with effluent (FE). Each compartment was further divided into three sampling zones by soil drainage class that included well drained, moderately well drained and poorly drained. Within each drainage zone a central soil sampling site was established.

For each soil sampling site, one intensive soil pit was dug. For each intensive pit, soil was sampled from the surface horizon (treated as an O horizon in the forest sites and

Table 1. Mean monthly effluent concentrations in 1996/1997. Data obtained from CVSD.

PARAMETERS	UNIT	1996	1997
Ammonia Nitrogen	mg/l	11.9	10.5
BOD	mg/l	26.8	21.3
Nitrate Nitrogen	mg/l	<0.5	0.2
pH		7.3	NA
Phosphorus	mg/l	3.8	2.8
Settleable Solids	mg/l	<0.01	0.1
Total Kjeldahl Nitrogen	mg/l	15.0	13.6
Total Suspended Solids	mg/l	10.1	17.1
Organic Nitrogen	mg/l	2.6	3.7

*Refer to Appendix A for the entire data set.

Ap horizon in the field sites), and then starting at the top of the B horizon depth increments 0-2 cm, 2-7 cm, 7-27 cm, 27-47 cm, 47-67 cm, and 67-87 cm depth increments when possible. In addition, at each site four satellite pits were dug equidistant from each other and two meters from the intensive pit. At these pits, only the surface horizon, 0-2 cm and 2-7 cm of the B horizon were sampled.

After collection, soil samples were placed in canvas bags and dried at approximately 60° C. The mineral soil samples were then passed through a 2 mm sieve and stored in cardboard quart containers. Organic soils were treated similarly, but passed through a 6mm sieve.

The pH of the mineral soils was determined using a 2:1 dilution, and organic soils

using a 10:1 dilution, for pH in both deionized water and 0.01M $CaCl_2$. Organic matter content was estimated by loss-on-ignition in a muffle furnace at 450° C for 12 hours.

Calcium (Ca), magnesium (Mg), phosphorus (P), zinc (Zn), manganese (Mn), aluminum (Al) and iron (Fe) were extracted using 1 N NH_4Cl, according to Kraske and Fernandez (1989). Organic P was determined using an ignition method (Saunders and Williams 1955, as modified by Walker and Adams, 1958). Concentrations in the extracts were measured using a Jarrell-Ash® Model 975 inductively coupled plasma spectrometer (ICP). Sodium (Na) and potassium (K) were also extracted using 1 N NH_4Cl, with concentrations measured using a Thermo Jarrell-Ash® Scan 1 flame atomic absorption spectrophotometer (AA). Exchangeable acidity was determined using 1N KCl soil extracts as described by Roberge and Fernandez (1986). Total carbon (C) and nitrogen (N) were determined using a LECO CN-2000® combustion analyzer.

Adjacent to each of the satellite pits in the field sites (GC and GE), all vegetation was harvested in a 1 m^2 area. The vegetation was dried at approximately 60° C, and then ground in a Wiley® mill to pass through a 20 mesh screen. Samples were then stored in ziplock bags at room temperature.

Foliage was analyzed by ICP for Ca, Mg, P, Al, Fe, Mn, and Zn after dry ashing using 50% HCl and concentrated HNO_3, according to methods of Kalra and Maynard (1991). Na and K in these digests were analyzed by AA. Total N was determined by the LECO CN-2000® combustion analyzer. Mass balance for N and P in the field sites was calculated using the following assumptions: i) the amount of atmospheric deposition of N obtained from precipitation was based on chemistry data from the NADP/NTN station

in Greenville, Maine (1997), and dry deposition was estimated to be two times wet deposition based on similar estimates in literature (Rustad et al., 1994), ii) the volume of effluent sprayed onto the field and the TKN/P effluent concentrations were averaged over the winter treatment months for years 1996/1997, and iii) the depth of the surface horizon of the soil was averaged across all pits in the field, and the bulk density was assumed to be 0.5g cm^{-3} in the surface Ap horizons and 1.0 g cm^{-3} in the subsurface mineral soil horizons. Total soil depth utilized in the calculation was 13 cm (using the average Ap horizon depth of 5cm, and the upper 7cm of mineral soil).

Statistical analyses were conducted using analysis of variance (ANOVA), with means separation by the Student Newman Keuls test at an alpha level of 0.05. All data were tested for normality by univariate analysis and transformed (logarithmically or squared) where necessary. Statistical analyses were performed using the Statistical Analysis System (SAS Institute Inc., 1988) on the University of Maine mainframe computer.

Chapter 3

RESULTS AND DISCUSSION

Vegetation (Forest vs. Field) Main Effects

A comparison of the forest control (FC) to the field control (GC) across all depths and drainage classes to assess the influence of the contrasting vegetation revealed the forest soil to have significantly higher C and N, exchangeable acidity, exchangeable H, CEC, available P, exchangeable Na, K, Al and Fe (Table 2). Soil pH and BS were significantly lower in the forest control. In addition, the percent Na saturation was significantly higher and the percent Mg saturation was significantly lower in the forest. No other significant differences were identified among the exchangeable cations.

Differences between controls were most likely due to the differences in the amount and type of organic matter in the soil as a function of vegetation type. The field area had been clear-cut approximately eight years prior to this study. This caused significant disturbance of the upper soil horizons due to tree harvesting and the use of heavy equipment. Not enough time had passed for large amounts of organic matter to have accumulated, and the organic matter that had accumulated was primarily from herbaceous plants as opposed to the thick organic horizon of softwood and hardwood litter in the undisturbed forest control. The harvesting of the trees also resulted in more open and warmer environmental conditions, causing a subsequent increase in the rate of decomposition. In a study of soil chemical changes after clear-cutting a forest in New Hampshire, Johnson et al., (1997) reported that much of the soil chemical changes observed in their study could be attributed to organic matter composition.

Table 2. Mean soil properties by treatment. For each parameter, values followed by the same letter among treatments are not significantly different at alpha=0.05 using Student Newman Keuls test.

		TREATMENT			
PARAMETER	**UNITS**	**FC**	**FE**	**GC**	**GE**
DI pH	stu	4.73a*	5.31c	5.15b	5.31c
CaCl pH	stu	4.17a	4.80b	4.67b	4.80b
Total C	%	6.76a	6.92a	3.47b	2.63c
Total N	%	0.26b	0.31a	0.14c	0.12c
C/N Ratio		26.1	22.7	24.4	21.2
Exch. Acidity	$cmol_c\,kg^{-1}$	3.96a	2.02bc	2.53b	1.82c
Exch. H	$cmol_c\,kg^{-1}$	2.33a	1.00b	1.38b	0.99b
CEC	$cmol_c\,kg^{-1}$	5.62b	9.55a	4.07c	3.89c
Ca	$cmol_c\,kg^{-1}$	0.90b	3.37a	0.69b	0.94b
Mg	$cmol_c\,kg^{-1}$	0.15c	0.68a	0.13c	0.20b
Na	$cmol_c\,kg^{-1}$	0.09c	0.35a	0.04d	0.21b
K	$cmol_c\,kg^{-1}$	0.27b	0.33a	0.16c	0.15c
BS	%	25.2d	55.0a	30.9c	43.7b
% Ca. Sat.	%	15.9c	35.6a	16.8c	24.2b
% Mg Sat.	%	2.58d	7.14a	3.24c	5.34b
% Na Sat.	%	1.51c	3.66b	1.01d	5.40a
% K Sat.	%	2.71a	2.99a	3.37a	3.48a
P	$mg\,kg^{-1}$	5.00a	4.07a	1.39b	1.15b
Al	$cmol_c\,kg^{-1}$	2.21a	1.37bc	1.57b	1.06c
Fe	$mg\,kg^{-1}$	13.3a	5.96b	5.96b	7.05b
Mn	$mg\,kg^{-1}$	2.91b	4.02b	2.88b	12.0a
Zn	$mg\,kg^{-1}$	1.17a	0.58bc	0.69ab	0.34c

* Data averaged over drainage class and depth.

The tendency for lower pH, greater exchangeable acidity, and more labile Al to be found in forest soils reflects the higher organic matter quantity and lower organic matter quality of forest vegetation compared to herbaceous grass materials (Johnson et al., 1997). The FC site had the highest C/N ratio and the lowest pH, which resulted in slower organic matter decomposition, and the production of greater organic acidity in soil solutions percolating through these soils (Sopher and Baird, 1978, Melilio et al., 1982, McClaugherty et al., 1985).

Treatment Main Effects

There were few significant effects of soil drainage class in this study and results are not presented here. Exchangeable Ca and K were significantly lower in the well drained soils as compared to the more poorly drained soils, reflecting greater leaching losses.

Table 2 summarizes the treatment effects for the four treatments included in this study, averaged across drainage class. Numerous significant differences existed among treatments attributable both to the application of effluent and vegetation type. In general, the application of either effluent or Snowfluent™ increased exchangeable base cation concentrations and the percent saturation of these cations on the exchange complex. This also resulted in a significant decrease in exchangeable acidity between both FC and FE, as well as GC and GE. However, only the forested site exhibited a significant increase in CEC. The greatest absolute and relative increase in soil exchangeable cations was for

Ca. Similar results were reported by Sopper and Kardos (1973), Burton and Hook (1979) and David and Struchtemeyer (1980).

Total N was significantly higher in FE than FC, indicating that N may have accumulated in the treated soil through additions from effluent. Total C was not significantly different between FE and FC. Consequently, the C/N ratio was lower in FE. In contrast, there was no difference in total N between GC and GE. This may reflect the differences in the Snowfluent™ irrigation process as compared to the conventional spray irrigation process. More N may be converted to NH_3 and volatilized as the snowpack ages, thus reducing the amount of N infiltrating into the soils. Total C was significantly lower in GE, thus lowering the C/N ratio in the Snowfluent™ treated site. This may have been due to an increase in moisture due to effluent irrigation, increasing the rate of breakdown of organic matter. This difference may be significant in the field sites and not in the forested sites because of the differences in amounts and types of organic matter on the site, with lower quality soil organic matter in the forested site being less responsive to treatments (Johnson et al., 1997, Melilio et al., 1982, McClaugherty et al., 1985).

Exchangeable Al was significantly lower in both FE and GE when compared to FC and GC, respectively (Table 2). Extractable Fe was significantly lower in FE than in FC, but not significantly different between GE and GC. An increase in BS, as well as the pH effects on Al and Fe solubility, would explain these exchangeable Al results. Similar results have been reported in a study by Metzner et al., (1987) in which concentrations of Al and Fe decreased as a result of liming. Exchangeable Zn showed a similar response and was significantly lower in both the effluent treated sites.

No significant differences were observed in available P between the controls and the effluent treated sites (Table 2). Because of possible P accumulations over time, additional characteristics of soil P were studied at this site. Table 3 shows results for the analysis of organic, inorganic and total P, both for overall means for all depths and by depth increment. No significant differences between GC and GE were detected overall for the P variables in Table 3. Organic P showed no significant differences between FC and FE, but there was significantly greater inorganic P and total P in FE when analyzed across all depths. However, when analyzed by depth the differences in P disappeared after a depth of 7 cm, suggesting that the soils are capable of adsorbing the added P relatively quickly. Similar results have been reported by Richenderfer et al., (1975), and David and Struchtemeyer (1980). Sommers et al., (1977) concluded that the concentration of hydrous iron oxides in the soil was important in phosphorus retention after studying P dynamics on soils from the Penn State area that had been irrigated with municipal wastewater.

Depth Main Effects

Because of the marked morphological and biogeochemical stratification due to soil forming processes in Maine soils, it is important to recognize the influences of soil depth in the interpretation of soil response to vegetation and treatment. Total C and N are significantly higher near the surface and decrease with depth due to inputs of organic matter as litter and the products of decomposition (Table 4). This depth trend in soil organic matter in turn affects almost all other soil parameters. Soil CEC decreases with depth due to the influences of the high surface area and reactivity of humic substances.

Table 3. Mean concentrations for soil P measurements by depth and overall for all depths by parameter. Means followed by the same letter are not significantly different among treatments at alpha=0.05 using Student Newman Keuls test.

		TREATMENT			
PARAMETER	**DEPTH**	**FC**	**FE**	**GC**	**GE**
AVAILABLE P (mg kg-1)	SURFACE	125a	49.7b	2.37c	1.48d
	0-2	2.30a	2.61a	1.26b	1.02b
	2-7	1.30a	1.31a	1.15a	1.09a
	7-27	1.04a	1.03a	1.03a	1.02a
	27-47	1.01a	1.02a	1.01a	1.01a
	47-67	1.01a	1.01a	1.01a	1.01a
	67-87	1.01a	1.01a	1.01a	1.01a
	Overall	5.00a	4.07a	1.39b	1.15b
TOTAL P (mg kg-1)	SURFACE	1069a	1158a	771b	1037a
	0-2	629b	1069a	874ab	685b
	2-7	598a	761a	761a	725a
	Overall	749b	967a	790ab	795ab
INORGANIC P (mg kg-1)	SURFACE	342b	399b	481b	678a
	0-2	387c	809a	643ab	487bc
	2-7	401a	539a	587a	545a
	Overall	386b	578a	571a	565a
ORGANIC P (mg kg-1)	SURFACE	728a	759a	290b	360b
	0-2	242a	260a	231a	198a
	2-7	198a	223a	174a	181a
	Overall	362a	389a	219b	230b

Table 4. Mean soil properties by depth. For each element, values followed by the the same letter are not significantly different at alpha=0.05 using Student Newman Keuls test.

DEPTH	pH	Total C %	Total N %	BS %	Tot Acidity cmol$_c$ kg^{-}	CEC cmol$_c$ kg^{-1}
SURFACE	5.16a	18.7a	0.77a	75.2a	3.44a	24.4a
0-2cm	5.06a	4.93b	0.20b	31.6bc	3.24a	5.24b
2-7cm	5.10a	3.80c	0.16c	27.0bcd	2.40ab	3.52c
7-27cm	5.15a	2.31d	0.10d	23.1d	1.70b	2.17d
27-47cm	5.14a	0.81e	0.04e	24.7cd	0.91c	1.21e
47-67cm	5.20a	0.45f	0.02f	26.0bcd	0.68c	0.97f
67-87cm	5.25a	0.23g	0.01g	32.6b	0.61c	0.91f

DEPTH	Ca cmol$_c$ kg	Mg cmol$_c$ kg^{-1}	K cmol$_c$ kg^{-1}	Na cmol$_c$ kg^{-1}	Ca Sat. %	Mg Sat. %
SURFACE	14.4a	2.81a	0.57a	0.29a	58.8a	11.5a
0-2cm	1.09b	0.21b	0.16b	0.13b	20.8b	4.01b
2-7cm	0.57c	0.11c	0.13b	0.10b	16.3c	3.01c
7-27cm	0.27d	0.05d	0.09b	0.07c	12.7d	3.01c
27-47cm	0.10e	0.02e	0.12b	0.05d	8.03e	1.36e
47-67cm	0.08ef	0.02e	0.09b	0.05d	8.46e	2.13d
67-87cm	0.06f	0.02e	0.17b	0.04e	6.94e	1.76d

DEPTH	K Sat. %	Na Sat. %	Available P mg kg^{-1}	Al cmol$_c$ kg^{-1}	Fe mg kg^{-1}	Mn mg kg^{-1}	Zn mg kg^{-1}
SURFACE	1.78d	1.17e	12.2a	0.73c	8.75b	62.7a	3.16a
0-2cm	2.72c	2.43d	1.67b	2.69a	19.9a	4.08b	0.69b
2-7cm	3.33c	2.89cd	1.23c	1.87b	9.68b	2.30c	0.39c
7-27cm	3.90c	3.34bc	1.03c	1.24c	4.17c	0.77d	0.21d
27-47cm	8.37b	4.35ab	1.01c	0.64c	0.62d	0.26e	0.10e
47-67cm	7.96b	5.24a	1.01c	0.40c	0.77d	0.33e	0.10e
67-87cm	15.2a	4.04ab	1.01c	0.40c	0.56d	0.15f	0.10e

* Data averaged across drainage class and treatment.

Similarly Ca, Mg, Na, K, Mn, Zn, and P also follow this trend, reflecting organic matter controls over their availability. BS decreases with depth, driven primarily by declining exchangeable Ca and Mg. Similar results were reported by Fernandez et al., (1993) in a study estimating nutrient content in a spruce-fir forest in eastern Maine.

Percent Na and K show the opposite trend, occupying a lower percentage at the surface and a higher percentage as depth increases. Several factors could help explain this trend: a) K and Na are monovalent cations, are highly hydrated in solution, and thus are not as strongly adsorbed by humic substances or clays as divalent ions (Stewart et al., 1990), b) Na is not an essential plant nutrient even though it is present in plant tissue, consequently it is not biocycled like the other ions, c) K is always ionic in plant tissue and easily leached, and d) there are less weathered aluminosilicate minerals with depth, allowing Na and K to be replenished through primary mineral weathering.

Exchangeable Al and Fe are lower in the surface horizon, with an increase in the 0-2 cm layer, then a gradual decrease as depth increases. The nature of podzolization, which results in Al and Fe mobilization from the surface horizons to the upper B horizons, (Wolt, 1994) may explain this vertical trend.

Figures 1 through 4 demonstrate the interaction between depth and treatment effects for key elements in this study most important in defining soil response to treatments. The interaction of treatment and depth for base saturation showed declining base saturation with depth for all treatments (Figure 1). Exchangeable Mg and Na followed similar trends (data not presented). The FE soils had significantly higher exchangeable Ca (Figure 2) to a depth of 47 cm than all other treatments attributable to

Figure 1 - Percent base saturation depth profiles by treatment

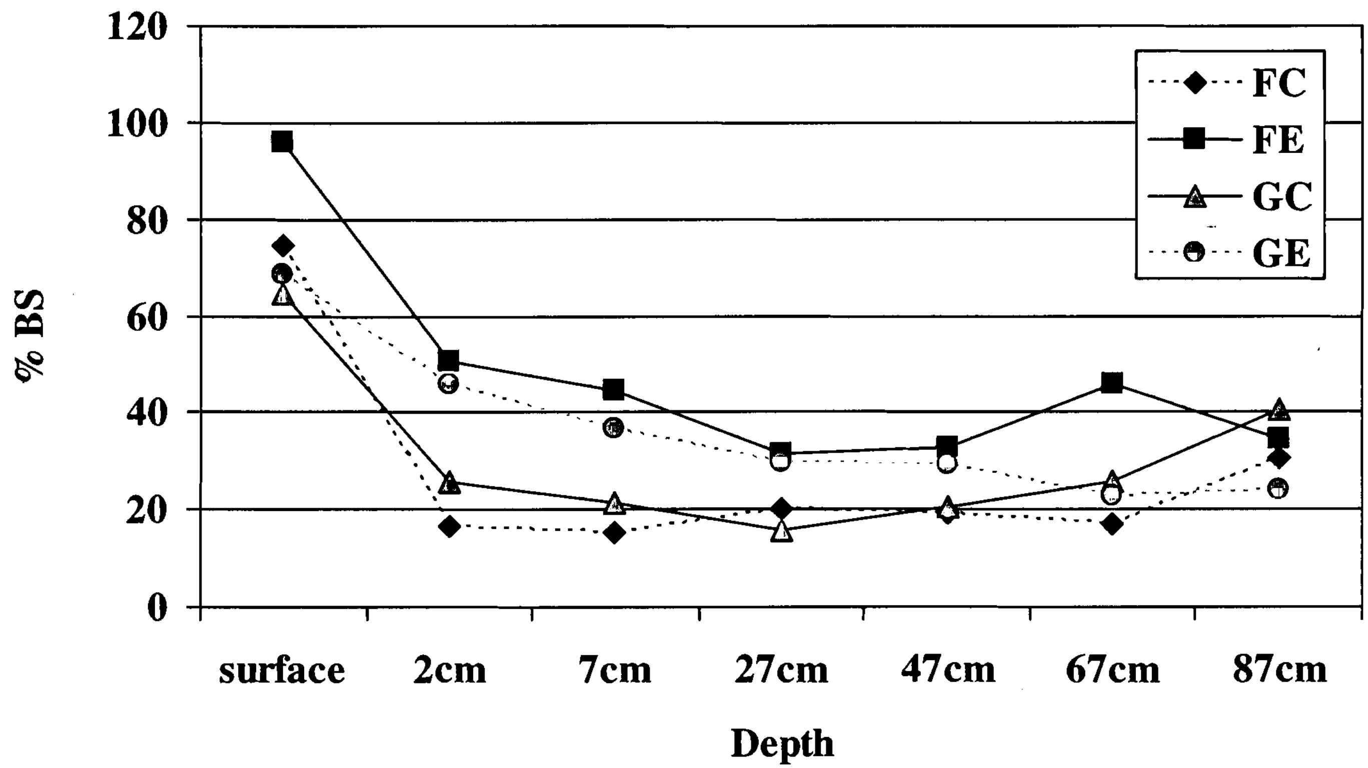


Figure 2 - Exchangeable Ca depth profiles by treatment

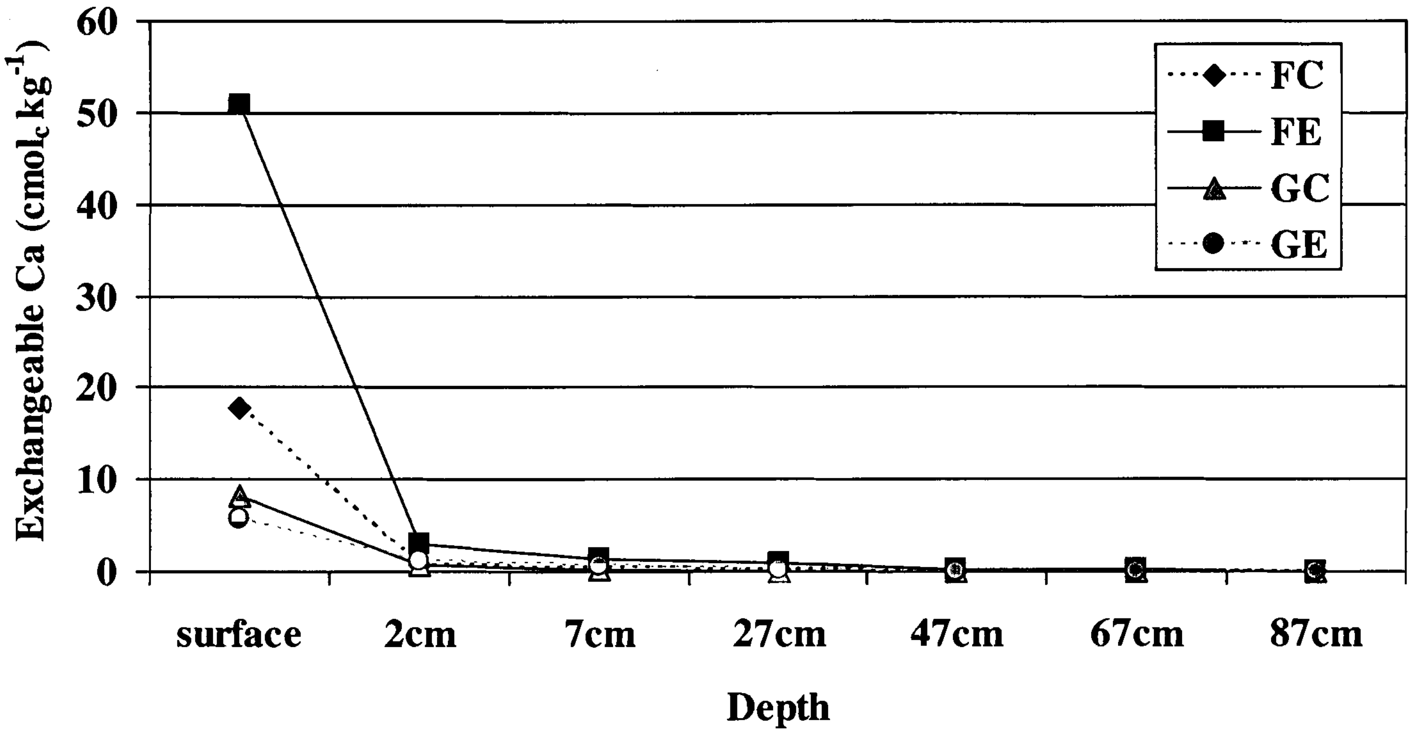


Figure 3 - Exchangeable K/ %K depth profiles by forest treatment

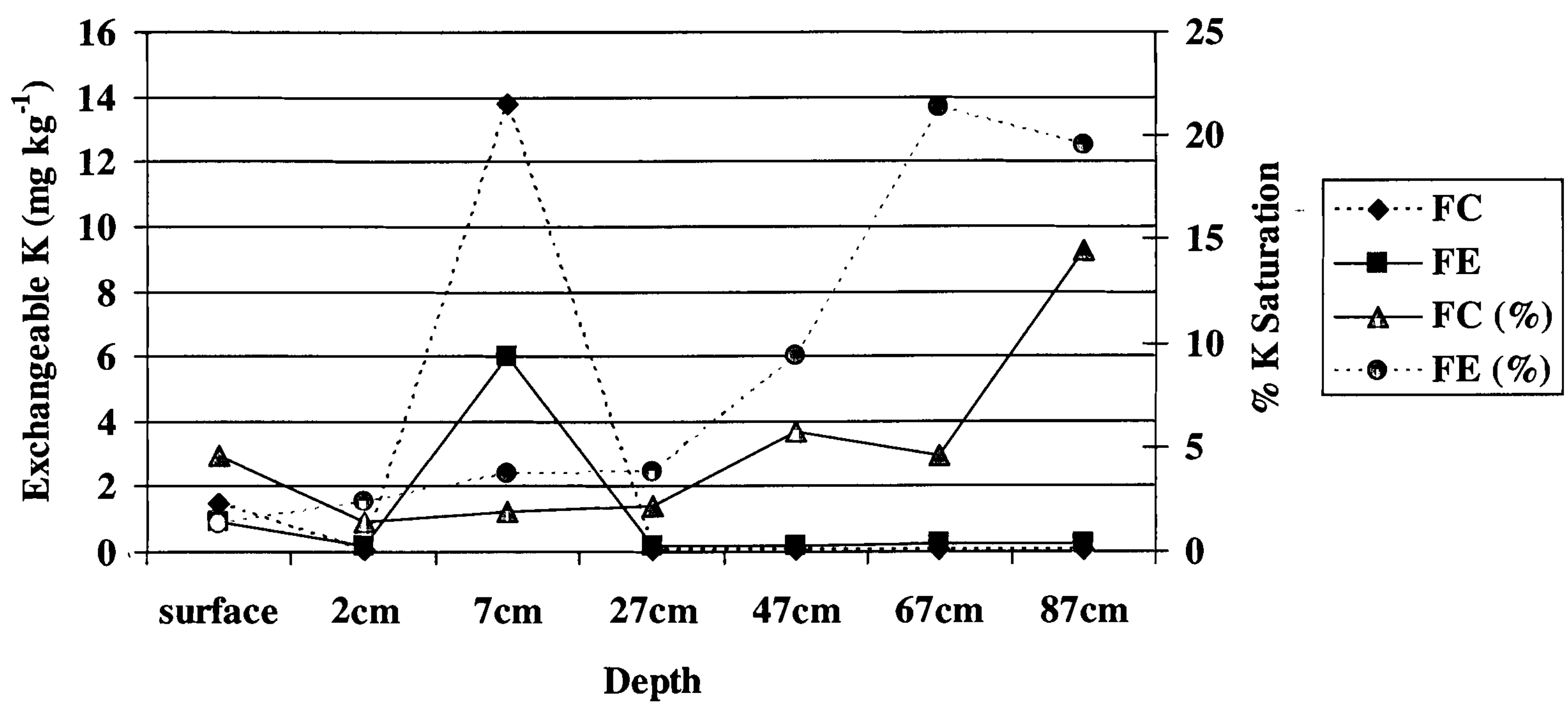
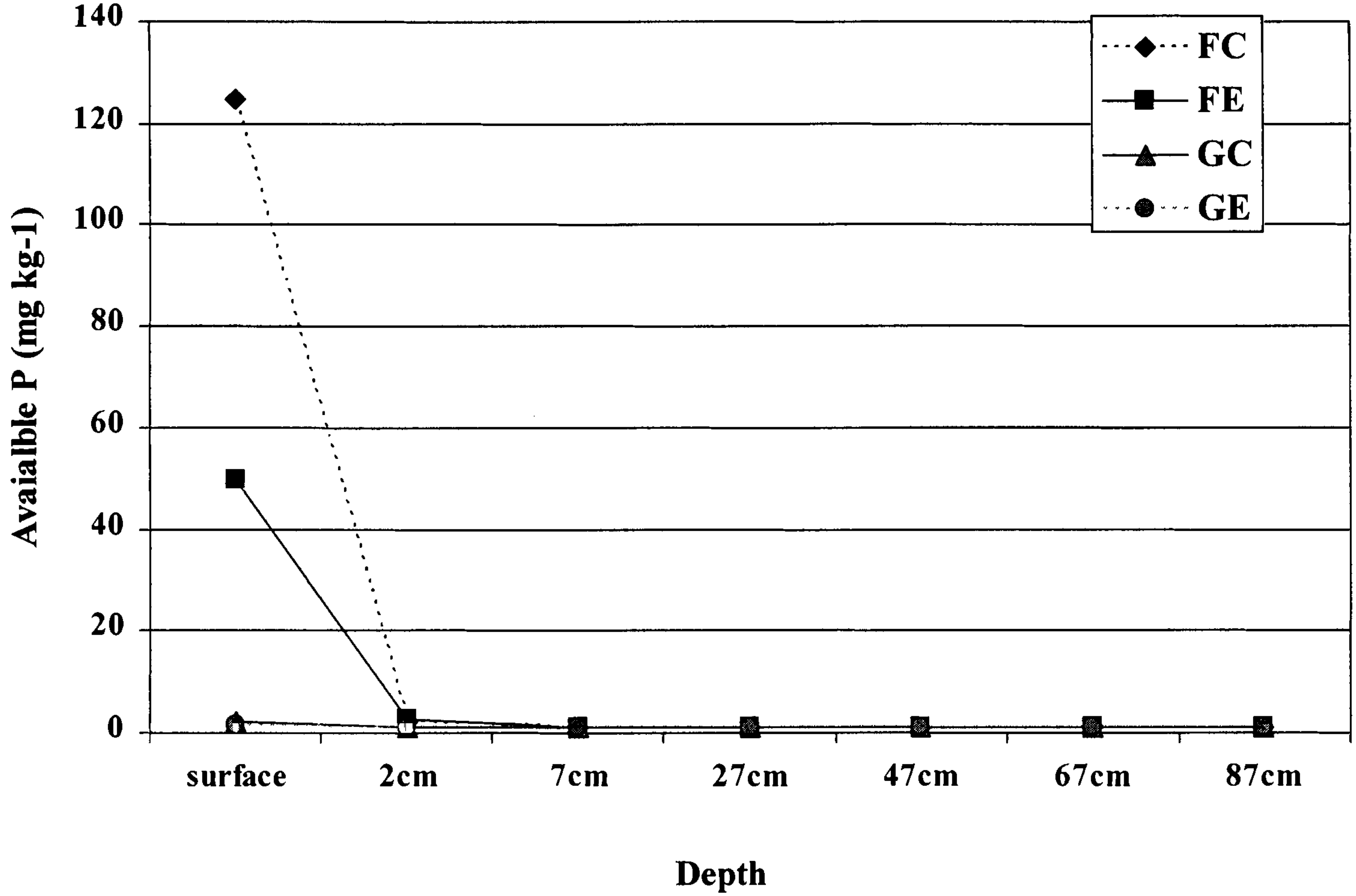


Figure 4 - Available P depth profiles by treatment



the higher CEC in these soils (Appendix B). The Snowfluent™ treated field site (GE) was significantly lower in exchangeable Ca in the surface soils than all other treatments, but significantly higher in Ca concentration than the control in the mineral soils through a depth of 7 cm. Lower exchangeable Ca likely reflects the lower CEC in GE since percent Ca saturation was nearly identical for FC (56.7%), GC (51.9%) and GE (52.3%). Ca saturation for FE was 77.7%. Below 7 cm, there were no significant differences among treatments for exchangeable Ca.

Figure 3 shows the relationship between exchangeable K and % K saturation in the forested sites, and illustrates the importance of assessing both absolute and relative concentrations of exchangeable cations in soils across depth. Exchangeable K showed peak concentrations at a depth of 7 cm for both FC and FE, but no accumulation of K was evident at this depth when the data were expressed as a percentage of the CEC. Both K and Na showed that although the absolute concentrations of the cations were decreasing with depth, the relative concentrations were increasing with depth due to gradients in organic matter and weatherable minerals as discussed above.

Available P concentrations decreased with depth from surface (Figure 4). The surface soils in both control sites, FC and GC, were significantly higher in available P than the effluent treated sites. This may be due to increased uptake by vegetation on effluent amended soils. This explanation is supported by the mass balance estimations of the field sites (Table 6), which show a significant increase in P in the grass biomass. No significant differences among treatments were identified in the deeper soil increments.

Grass Chemical Composition

The chemical composition of grass tissue from the field site was analyzed to determine effects of treatment (Snowfluent™ amended vs. control). This is of particular interest since mowing and harvesting grasses could offer a feasible means of preventing the accumulation of certain nutrients or metals in this type of effluent management system. While forest harvesting offers parallel possibilities, the length of time for crop maturity limits the utility of such an approach to excess nutrient removal. Table 5 compares the composition for selected constituents of grasses grown in the control versus Snowfluent™ areas. All constituents, except Ca, were in significantly higher concentrations in the grasses grown in the Snowfluent™ amended area when compared with controls. These differences are primarily attributable to the additional nutrients derived from the Snowfluent™ and perhaps increased moisture availability in these soils.

Table 6 shows an estimate of mass balance for N and P on the field control and Snowfluent™ amended sites. Grasses were able to take up an amount equal to all of the treatment applied N, as well as some of the N that was stored in the upper soil layers. The additional moisture as well as additional nutrients may account for the large increase in N uptake by the grasses. Grasses were able to take up an amount equal to approximately 75% of the P deposited by Snowfluent™ irrigation. These data show that grasses were able to respond to nutrient and water additions from effluent by increasing nutrient accumulation in the live vegetative biomass. Other researchers have reported similar results (Kardos et al., 1977; Bunel et al., 1995, George et al., 1985, Geber, 2000).

Table 5. Mean grass composition by treatment. For each element, values followed by the same letter are not significantly different between treaments at alpha=0.05 using Student Newman Keuls Test.

ELEMENT		TREATMENT	
		GC	**GE**
N	kg ha-1	17b	56a
Ca	kg ha-1	7.5a	12a
K	kg ha-1	20b	55a
Mg	kg ha-1	3.3b	6.5a
P	kg ha-1	2.4b	6.1a
Al	kg ha-1	0.03b	0.11a
Fe	kg ha-1	0.14b	0.69a
Mn	kg ha-1	1b	1.7a
Na	kg ha-1	0.17b	0.88a
Zn	kg ha-1	0.07b	0.15a

ELEMENT		TREATMENT	
		Field Control (GC)	**Snowfluent (GE)**
N	%	0.624b	1.64a
Ca	mg kg-1	2720a	3396a
K	mg kg-1	7237b	16133a
Mg	mg kg-1	1200b	1906a
P	mg kg-1	870b	1797a
Al	mg kg-1	11.1b	31.0a
Fe	mg kg-1	49.4b	202a
Mn	mg kg-1	378b	486a
Na	mg kg-1	61.3b	257a
Zn	mg kg-1	26.7b	44.1a

Table 6: Mass balance in the field sites for N and P.

	TREATMENT	
	GC	GE
TOTAL BIOMASS (kg\ha)	2760	3410
NITROGEN		
Rain (kg/ha/yr)	6	6
Treatment (kg/ha/yr)		26
Grass (kg N /ha)	17	56
Soil (kg N /ha)	2250	2030
PHOSPHORUS		
Rain (kg/ha/yr)	--	--
Treatment (kg/ha/yr)		8
Grass (kg P /ha)	2	6
Soil (kg P /ha)*	1.5	1.2

* Represents estimate for extractable P

The vegetation acts as a sink for nutrients, and plays an important part in the renovation of applied wastes. Harvesting the grasses from the site could provide a way to remove excess nutrients from the soils in treated areas, and could significantly enhance the capacity of the site to process effluent-derived nutrients like N and P. Grasses treated with effluent removed 39 kg N/ha and 4 kg P/ha more than the control grasses. These numbers appear to indicate that the grasses are more effective at removing N in this study compared with P. However, relative to soil pools, treatment grass nutrient accumulations showed a 1.9% increase in N uptake as opposed to a 300% increase in P uptake. Therefore, grass harvesting may be most effective in managing soil P accumulation since treatment N inputs are small relative to the existing pools.

Soil drainage class did not seem to have a large effect on the ability of the grasses to absorb nutrients, with only Na and K showing significant differences across drainage class (data not presented). Analysis of the dry weights of the grass harvested (averaged across treated and untreated sites) showed the well drained site had significantly less biomass (mean = 170g m^{-2}) as compared to the moderately well and poorly drained sites (mean = 330 and 427g m^{-2}, respectively). Precipitation for the year of this study (1996) was 102.7cm, compared to the thirty year average (1971-2000) of 95.1cm, with the range during this period from 71.3cm to 123.5cm (National Weather Service, 2002). These data suggest the study year was not unusually dry or wet and thus results from this study should reflect representative climatic conditions.

Chapter 4

CONCLUSIONS

A study was conducted to evaluate the response of soils at the Carrabassett Valley Sanitary District to a Snowfluent™ system and an adjacent site that has been treated for approximately 20 years with conventional effluent irrigation.

1) Differences in soil properties between the forest and field control sites reflect differences in the amount and type of organic matter in the soil:
   a) Forest soils had significantly higher C and N, exchangeable acidity, CEC, available P, exchangeable Na, K, Al, and Fe than the field control soils.
   b) Forest soil pH and BS were significantly lower than the field control soils.

2) There were few significant effects of soil drainage class on soil properties or soil response to treatment.

3) Snowfluent™ and conventional spray irrigation of soils with municipal effluent:
   a) Increased soil exchangeable Ca, Mg, Na, K, and base saturation.
   b) Decreased soil available P.
   c) Increased soil pH.
   d) Had its greatest effects on surface soil with decreasing effects with depth that appears to be reduced to background between 27 and 47cm for most parameters.

4) Depth patterns in soil properties reflect the decreasing influence of litter and its decomposition products as evidenced by:

a) Significantly higher total C and N in the surface compared to lower horizons.

b) Decreasing exchangeable Ca, Mg, Na, K, Mn, Zn, and P with depth reflecting organic matter controls over their availability.

c) Decreasing BS with depth, driven primarily by declining exchangeable Ca and Mg.

5) Irrigation with Snowfluent™ altered the chemical composition of the grasses grown on the site:

a) All element concentrations in grass foliage measured were significantly higher in the treated site except Ca.

b) The differences are primarily attributed to the additional nutrients and moisture derived from Snowfluent™.

The use of spray irrigation and Snowfluent™ as a wastewater treatment strategy appears to work well. The soil and vegetation seem to be able to retain most of the applied nutrients, and do not appear to be moving toward saturation. Vegetation management will be a key consideration as the system ages, should any sign of nutrient accumulation begin to develop.

Recommendations for future studies would include further measurements of the soils associated with the Snowfluent™ system, as the system was relatively new during this study. Assessing the environmental benefits and cost feasibility of harvesting the vegetation would also be useful in determining the long term viability of the system.

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APPENDICES

APPENDIX A

Table A1. Mean wastewater effluent constituent concentrations, as provided by Carrabassett Valley Sanitary District.

PARAMETERS	UNIT	1996 JAN	FEB	MAR	APR	MAY	JUNE	JULY	AUG	SEPT	OCT	NOV	DEC
Ammonia Nitrogen	mg/l	15	20	20	25	19	15	<0.5	<0.5	<0.5	4.4		
BOD	mg/l	17	29	24	60	53	63	3	9	3	7		
Nitrate Nitrogen	mg/l	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	0.58		
pH	stu	7.2	6.4	7.2	7	7.1	7.6	7.3	8.3	7.4	7.5		
Phosphorus	mg/l	4.2	4.9	5	12	4.6	4	0.42	0.47	0.65	2.4		
Settleable Solids	mg/l	<0.1	<.01	<.01	<.01	<.01	<.01	0.01	<.01	<.01	<.01		
Total Kjeldahl Nitrogen	mg/l	18	12	25	30	23	19		1.2	<1	6.5		
Total Suspended Solids	mg/l	8	3	6	20	19	25	11	5	1	3		
Organic Nitrogen	mg/l	3	0.6	5	5	4	4	1.2	1.2	<1	2.1		
	No discharge for Nov & Dec 1996												

PARAMETERS	UNIT	1997 JAN	FEB	MAR	APR	MAY	JUNE	JULY	AUG	SEPT	OCT	NOV	DEC
Ammonia Nitrogen	mg/l	18	25	27	25	18	2.4	0.2	<0.1	0.1	1.2	3.8	5
BOD	mg/l	27	34	58	57	11	36	5	<4	4	9	8	4
Nitrate Nitrogen	mg/l	0.05	0.1	0.07	0.19	0.17	0.06	0.06	<0.05	0.09	0.93	0.19	0.19
Phosphorus	mg/l	4.6	5	5.02	6.36	4.46	4.48	0.19	0.21	0.21	0.96	0.71	0.99
Settleable Solids	mg/l	<0.1	<0.1	<0.1	<0.1	<0.1	1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Total Kjeldahl Nitrogen	mg/l	22	29	31	28	20	14	0.8	0.4	0.5	4.6	6.4	6.3
Total Suspended Solids	mg/l	21	13	21	20	25	47	6	7	6	23	10	6
Organic Nitrogen	mg/l		4	4	4	3	2	11.6	0.6	<0.4	3.4	2.6	1.3

APPENDIX B

Table B1. Mean soil properties by treatment and horizon.

	TREATMENT			
	Forest Control FC	**Effluent Forest FE**	**Field Control GC**	**Snowfluent GE**
Total Acidity (meq/100g)				
surface	7.62a	0.91c	4.14b	2.82b
0-2 cm	4.83a	3.73ab	2.98b	1.81c
2-7 cm	3.17a	2.41ab	2.19ab	1.92b
7-27 cm	1.65a	2.98a	1.36a	1.08a
27-47 cm	0.84a	1.12a	1.18a	0..57a
47-67 cm	0.78a	0.54a	0.76a	0.67a
67-87 cm	0.38a	0.83a	0.81a	0.52a
Exchangeable Calcium (meq/100g)				
surface	17.8b	50.8a	8.13c	5.81d
0-2 cm	0.61c	2.95a	0.69c	1.14b
2-7 cm	0.36c	1.34a	0.33c	0.69b
7-27 cm	0.27b	0.87a	0.11b	0.23b
27-47 cm	0.08b	0.25a	0.06b	0.08b
47-67 cm	0.06a	0.15a	0.08a	0.07a
67-87 cm	0.05a	0.1a	0.09a	0.04a
Exchangeable Magnesium (meq/100g)				
surface	3.81b	9.42a	1.75c	0.98d
0-2 cm	0.09c	0.62a	0.13c	0.28b
2-7 cm	0.05b	0.27a	0.06b	0.19a
7-27 cm	0.03a	0.17a	0.03a	0.04a
27-47 cm	0.01b	0.05a	0.01b	0.02a
47-67 cm	0.02a	0.03a	0.01a	0.02a
67-87 cm	0.01a	0.03a	0.02a	0.01a

Available Phosphorus (mg/kg)

surface	125a	49.7b	2.37c	1.48d
0-2 cm	2.3a	2.61a	1.26b	1.02b
2-7 cm	1.30a	1.31a	1.15a	1.09a
7-27 cm	1.04a	1.03a	1.03a	1.02a
27-47 cm	1.01a	1.02a	1.01a	1.01a
47-67 cm	1.01a	1.01a	1.01a	1.01a
67-87 cm	1.01a	1.01a	1.01a	1.01a

Exchangeable Aluminum (meq/100g)

surface	0.99a	0.19b	1.13a	0.62ab
0-2 cm	4.16a	2.55b	2.58b	1.48b
2-7 cm	2.57a	1.71ab	1.74ab	1.47b
7-27 cm	1.48a	1.55a	1.14a	0.79a
27-47 cm	0.56a	0.79a	0.70a	0.48a
47-67 cm	0.44a	0.38a	0.42a	0.36a
67-87 cm	0.29a	0.65a	0.40a	0.39a

Iron (mg/kg)

surface	31.0a	2.47c	8.85b	8.63b
0-2 cm	33.0a	23.9ab	14.3b	13.9b
2-7 cm	13.5a	9.14a	6.76a	10.5a
7-27 cm	4.34a	5.95a	3.1a	3.79a
27-47 cm	1.0a	1.11a	0.78a	0.95a
47-67 cm	0.74a	0.67a	0.71a	0.95a
67-87 cm	0.50a	0.50a	0.67a	0.50a

Manganese (mg/kg)

surface	205a	111b	18.2d	37.4c
0-2 cm	1.53c	2.34bc	4.31b	17.9a
2-7 cm	0.75c	1.12c	2.2b	15.2a
7-27 cm	0.1b	0.77a	0.85a	5.56a
27-47 cm	0.1b	0.20b	0.16a	1.53a
47-67 cm	0.29a	0.50a	0.16a	0.50a
67-87 cm	0.10a	0.40a	0.16a	0.14a

Zinc (mg/kg)

surface	33.4a	0.78c	2.50b	1.54bc
0-2 cm	0.87a	0.83a	1.0a	0.31b
2-7 cm	0.33ab	0.65a	0.51ab	0.21b
7-27 cm	0.10b	0.74a	0.25b	0.1b
27-47 cm	0.10a	0.10a	0.10a	0.10a
47-67 cm	0.10a	0.10a	0.10a	0.10a
67-87 cm	0.10a	0.10a	0.10a	0.10a

Exchangeable Sodium (meq/100g)

surface	0.24c	1.43a	0.05d	0.38b
0-2 cm	0.09b	0.30a	0.04c	0.25a
2-7 cm	0.06b	0.23a	0.04c	0.22a
7-27 cm	0.05b	0.16a	0.03c	0.11ab
27-47 cm	0.03b	0.09a	0.04b	0.07a
47-67 cm	0.03a	0.07a	0.06a	0.56a
67-87 cm	0.02b	0.06a	0.03ab	0.06a

Exchangeable Potassium (meq/100g)

surface	1.47a	0.91b	0.17c	0.20c
0-2 cm	0.08b	0.21a	0.17a	0.20a
2-7 cm	13.8a	5.98b	7.58b	6.11b
7-27 cm	0.04b	0.17a	0.08ab	0.09ab
27-47 cm	0.06a	0.16a	0.25a	0.06a
47-67 cm	0.04a	0.25a	0.75a	0.06a
67-87 cm	0.09a	0.25a	0.32a	0.06a

pH (stu)

surface	4.14a	5.71c	5.32b	5.48bc
0-2 cm	4.87a	5.07ab	5.00a	5.30b
2-7 cm	4.86a	5.22b	5.12b	5.21b
7-27 cm	5.04a	5.16a	5.08a	5.30a
27-47 cm	5.16a	5.15a	5.09a	5.16a
47-67 cm	5.17a	5.18a	5.20a	5.23a
67-87 cm	5.33a	5.28a	5.21a	5.20a

Nitrogen (%)				
surface	1.55a	1.63a	0.39b	0.36b
0-2 cm	0.24a	0.26a	0.19ab	0.15b
2-7 cm	0.18ab	0.20a	0.14ab	0.13b
7-27 cm	0.13a	0.15a	0.10a	0.05a
27-47 cm	0.04a	0.06a	0.04a	0.03a
47-67 cm	0.02a	0.02a	0.02a	0.01a
67-87 cm	0.02a	0.03a	0.01a	0.01a
Carbon (%)				
surface	41.9a	36.6b	10.7c	7.55d
0-2 cm	6.48a	6.08a	4.67a	3.17b
2-7 cm	4.92a	4.64a	3.33ab	2.79b
7-27 cm	3.20a	3.50a	2.23ab	1.15b
27-47 cm	0.95a	1.30a	0.69a	0.49a
47-67 cm	0.45a	0.45a	0.68a	0.29a
67-87 cm	0.33a	0.58a	0.17b	0.16b
CEC (meq/100g)				
surface	31.5b	65.4a	15.7c	11.1d
0-2 cm	5.48b	8.24a	3.99b	4.17b
2-7 cm	3.64ab	4.58a	2.67b	3.46ab
7-27 cm	1.94a	4.43a	1.61a	1.59a
27-47 cm	1.02a	1.68a	1.59a	0.80a
47-67 cm	0.94a	1.06a	1.00a	0.88a
67-87 cm	0.56a	1.27a	1.36a	0.67a
% Base Saturation				
surface	74.8b	95.9a	64.8b	68.3b
0-2 cm	16.4c	50.8a	25.9b	46.0a
2-7 cm	15.4c	44.6a	21.4b	36.8a
7-27 cm	19.9a	31.3a	15.6a	29.5a
27-47 cm	19.0a	32.8a	20.5a	29.3a
47-67 cm	17.0b	45.8a	25.8b	22.8b
67-87 cm	30.5a	34.6a	40.8a	24.2a

% Ca Saturation				
surface	56.7b	77.7a	51.9b	52.3b
0-2 cm	11.2c	35.8a	17.2b	27.3a
2-7 cm	9.90c	29.3a	12.3c	19.8b
7-27 cm	13.6a	19.6a	6.54a	14.6a
27-47 cm	7.54ab	14.8a	3.55b	10.4ab
47-67 cm	7.56a	13.7a	6.61a	7.48a
67-87 cm	7.93a	7.87a	6.89a	5.77a
% Mg Saturation				
surface	12.1ab	14.4a	11.2b	8.82c
0-2 cm	1.61c	7.55a	3.20b	6.61a
2-7 cm	1.23c	5.89a	2.10b	5.43a
7-27 cm	1.35a	3.93a	1.63a	2.23a
27-47 cm	0.98bc	2.81a	0.63c	1.99ab
47-67 cm	2.43a	2.80a	1.44a	2.08a
67-87 cm	2.51a	2.36a	1.41a	1.49a
% Na Saturation				
surface	0.76c	2.19b	0.33d	3.41a
0-2 cm	1.56c	3.67b	1.03d	5.94a
2-7 cm	1.70b	4.95a	1.35b	6.18a
7-27 cm	2.54b	3.63ab	2.05b	6.58a
27-47 cm	3.25ab	5.53ab	2.24b	8.93a
47-67 cm	3.07a	6.17a	6.22a	6.39a
67-87 cm	4.34a	4.72a	2.29a	8.17a
% K Saturation				
surface	4.57a	1.35cb	1.01c	1.62b
0-2 cm	1.44c	2.40b	3.68a	4.29a
2-7 cm	1.95b	3.74a	4.38a	3.84a
7-27 cm	2.22a	3.81a	5.08a	5.38a
27-47 cm	5.71a	9.42a	12.2a	7.46a
47-67 cm	4.58b	21.4a	6.84b	5.60b
67-87 cm	14.5a	19.5a	21.4a	8.17a

BIOGRAPHY OF THE AUTHOR

Leslie Bourassa Nelson was born on March 11, 1970. She was raised in Winthrop, Maine and graduated from Winthrop High School in 1988. Leslie attended the University of Maine, and graduated in 1992 with a Bachelor's degree in Biology.

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