

## Abstract

The effect that plants (*Typha latifolia*) as well as root-bed medium physical and chemical characteristics have on the treatment of primary treated domestic wastewater within a vertical flow constructed wetland system was investigated. Five sets of cells, with two cells in each set, were used. Each cell was made of concrete and measured 1.0 m X 1.0 m and was 1.3 m deep. Four different root-bed media were tested : Queenston Shale, Fonthill Sand, Niagara Shale and a Michigan Sand. Four of the sets contained plants and a single type of root-bed medium. The influence of plants was tested by operating a Queenston Shale set without plants. Due to budget constraints no replicates were constructed. All of the sets were operated independently and identically for twenty-eight months. Twelve months of data are presented here, collected after 16 months of continuous operation.

Root-bed medium type did not influence BOD<sub>5</sub> removal. All of the sets consistently met Ontario Ministry of Environment (MOE) requirements (<25 mg/L) for BOD<sub>5</sub> throughout the year. The 12 month average BOD<sub>5</sub> concentration from all sets with plants was below 2.36 mg/L. All of the sets were within MOE discharge requirements (< 25 mg/L) for suspended solids with set effluent concentrations ranging from 1.53 to 14.80 mg/L. The Queenston Shale and Fonthill Sand media removed the most suspended solids while the Niagara Shale set produced suspended solids. The set containing Fonthill Sand was the only series to meet MOE discharge requirements (< 1mg/L) for total phosphorus year-round with a twelve month mean effluent concentration of 0.23 mg/L. Year-round all of the root-bed media were well below MOE discharge requirements (< 20 mg/L in winter and < 10 mg/L in summer) for ammonium. The Queenston

Shale and Fonthill Sand sets removed the most total nitrogen.

Plants had no effect on total nitrogen removal, but did influence how nitrogen was cycled within the system. Plants increased the removal of suspended solids by 14%, BOD<sub>5</sub> by 10% and total phosphorus by 22%. Plants also increased the amount of dissolved oxygen that entered the system. During the plant growing season removal of total phosphorus was better in all sets with plants regardless of media type.

The sets containing Queenston Shale and Fonthill Sand media achieved the best results and plants in the Queenston Shale set increased treatment efficiency for every parameter except nitrogen. Vertical flow wetland sewage treatment systems can be designed and built to consistently meet MOE discharge requirements year-round for BOD<sub>5</sub>, suspended solids, total phosphorus and ammonium. This system is generally superior to the free water systems and sub-surface horizontal flow systems in cold climate situations.

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## Introduction and literature review

The use of wetlands, constructed or natural, to cleanse polluted water has been investigated for almost fifty years. A 1953 report from the Max-Planck Institute in Germany by Dr. Kathe Seidal discusses the possibility of using wetland plants to remove nutrients from wastewater prior to discharge into the natural environment (Brix, 1994). Work at this institute eventually led to the Max-Planck process, the first patented constructed wetland treatment system. Based on Dr. Seidal's ideas large scale systems were developed by the IJsselmeerpolders Development Authority in Holland beginning in 1967. In Europe the first full scale wetland treatment facility was constructed in Holland to treat wastewater from a campground accommodating 6,000 people (Brix, 1994).

In the late 1960's N.A.S.A. developed a hybrid wastewater treatment system using anaerobic micro-organisms and reed grass (*Phragmites communis*). Additional work by other researchers was related to the use of natural wetlands for treatment of wastewater or as receivers of treated water. However, it soon became obvious that this practice led to significant changes in the species composition and the function of natural wetlands. It was realized that constructed wetlands systems have a much greater potential for application, because of the opportunity to optimize control over the treatment process and because the use of constructed wetlands does compromise the integrity of natural wetlands (Brix, 1994).

Research since has led to the realization that constructed wetlands for wastewater treatment are capable of providing a low cost wastewater treatment alternative while providing a high level of treatment. Constructed wetlands for wastewater treatment are defined by Hammer (1989) as " a designed and man-made complex of saturated substrates, emergent and submergent vegetation,

animal life, and water that simulates natural wetlands for human use and benefits". These benefits could include: 1) low cost of construction and operation; 2) ease of maintenance; 3) effective and reliable wastewater treatment; 4) tolerance to fluctuating hydrologic and contaminant loading rates; and 5) indirect benefits such as green space, wildlife habitats, and recreational and educational areas (Hammer and Bastian, 1989). Approximately, 1,000 constructed wetlands are used in North America for treating water. They are being used to treat municipal wastewater, storm water, coal and metal mine drain water and wastewater produced from the agriculture, textile, and photography industries (Wren, 1997).

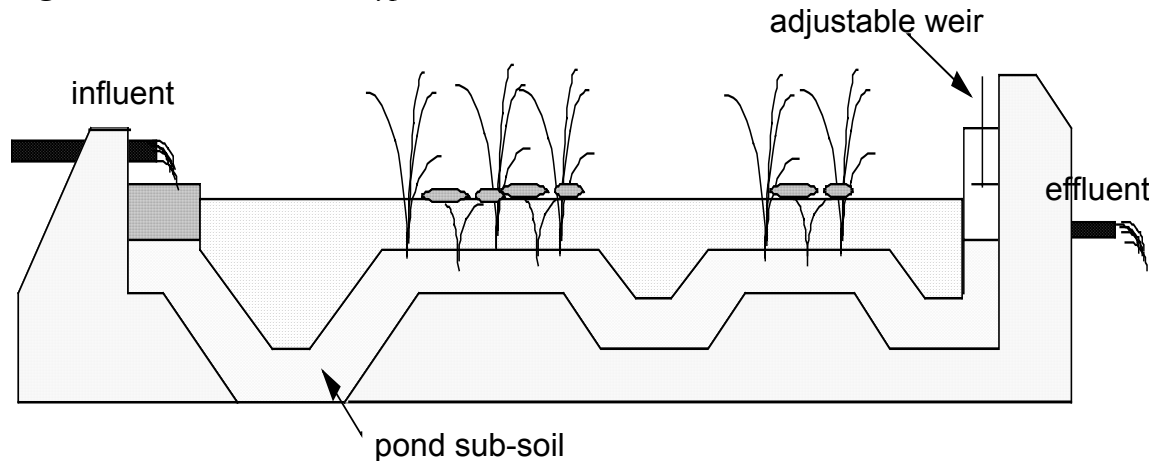
Three constructed wetland designs are currently in use. Bhamidimarri *et al.* (1990) identify these as:

- 1) Surface-flow wetlands or flowing water systems (FWS systems).
- 2) Subsurface horizontal flow wetlands (SF systems).
- 3) Subsurface vertical flow wetlands (VF systems).

## Section 1.1 — Surface-flow wetlands or flowing water systems (FWS systems)

Surface-flow wetlands closely mimic natural wetland systems. The FWS system (Figure 1). contains appropriate vegetation, often *Typha spp.* (common cattail),

**Figure 1.** Cross section of a typical FWS wetland.



*Scirpus spp.* (bulrush) and/or *Phragmites spp.* (reed grass). This vegetation is cultivated in shallow channels or beds containing a root medium such as gravel and/or sand. The wastewater is discharged into the system and allowed to flow freely through the system at a low velocity. Shallow water depth within the wetland system along with aquatic plant stalks and organic litter help regulate water flow. Oxygen is passively made available for biochemical reactions mainly by the diffusion of air into the system. The majority of the biochemical reactions occur on the submerged portions of the plants and within the wetland soils (Rogers *et al.*, 1991). Water depths greater than 0.3 m prevent the establishment and growth of undesirable aquatic vegetation (Pride *et al.*, 1990). In the US some 56 FWS systems process 95 million gallons a day of runoff and wastewater (Reed, 1991).

## FWS systems for domestic sewage treatment

In Ontario, wastewater treatment using design concepts of the FWS systems have been tested in an attempt to treat domestic sewage. From 1980 to 1984 an experimental FWS facility was operated at Listowel, Ontario (Herskowitz, 1986). This facility, however, failed to provide adequate treatment due to anaerobic conditions within the marsh system. Herskowitz (1986) gives three possible reasons for these anaerobic conditions: 1) Reduced alga photosynthesis due to shading by *Typha latifolia* and *Lemna spp.* (i.e. duckweed). 2) Oxygen provided by the *Typha latifolia* was not sufficient to meet the requirements of the system. 3) Ice cover in the winter may have inhibited the diffusion of atmospheric oxygen into the water.

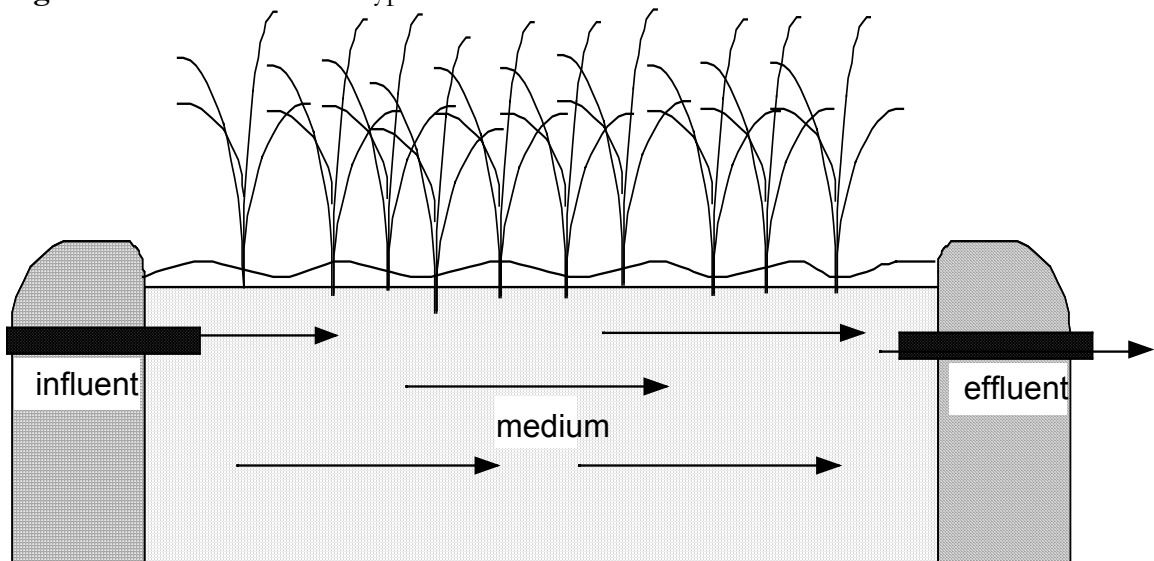
With recommendations made from the Listowel study implemented, a full-scale artificial marsh treatment facility was established at Port Perry, Ontario in 1984. This facility which was based on the FWS design, produced effluents that did not meet Provincial Water Quality Objectives. The draft report (Snell, 1990) gives the primary cause for this failure as anaerobic conditions within the marsh system. As in the study at Listowel, natural aeration was restricted in the summer by *Typha spp.* and *Lemna spp.* cover. Snow and ice cover in the winter resulted in severe anaerobic conditions. Decomposition of alga and duckweed biomass further aggravated the oxygen supply problem. In addition, Snell writes "the overland flow regime and relatively high water depths further limited treatment performance by restricting the contact of the applied wastewater within the sediment/rhizome matrix." An important conclusion reached in this unpublished study is that FWS treatment technology is not suitable to treat domestic sewage in Ontario. Low winter temperatures affected the systems at Listowel and Port Perry mainly by causing ice cover, which in turn caused anaerobic conditions.

## Section 1.2 — Subsurface Flow (SF) systems

The second type of system discussed by Bhamidimarri *et al.* (1990) is the subsurface flow (SF) wetland (Figure 2). SF wetlands first emerged as a wastewater treatment technology in Western Europe in the 1960s. Early developmental work in the US commenced in the early 1980s and continues to date.

The SF wetland consists of a medium (rock, sand or gravel) which is used to support aquatic vegetation, usually *Typha spp.*, *Phragmites spp.*, or *Scirpus spp.* The water levels in these beds is maintained below the top of the medium, thus creating a subsurface flow (Figure 2). Wastewater flows laterally or horizontally through the medium and is purified during contact with surfaces of medium particles and root zones of aquatic vegetation.

**Figure 2.** Cross section of a typical SF wetland.



Because of the increased opportunity of contact between wastewater and root-bed medium, phosphorus removal is more effective than in the FWS system (Reed, 1986). The primary oxygen source for this type of wetland is oxygen transmitted from the leaves to the roots of the aquatic plants (Gersburg *et al.*, 1986). Aside from their potential application to colder climatic conditions because

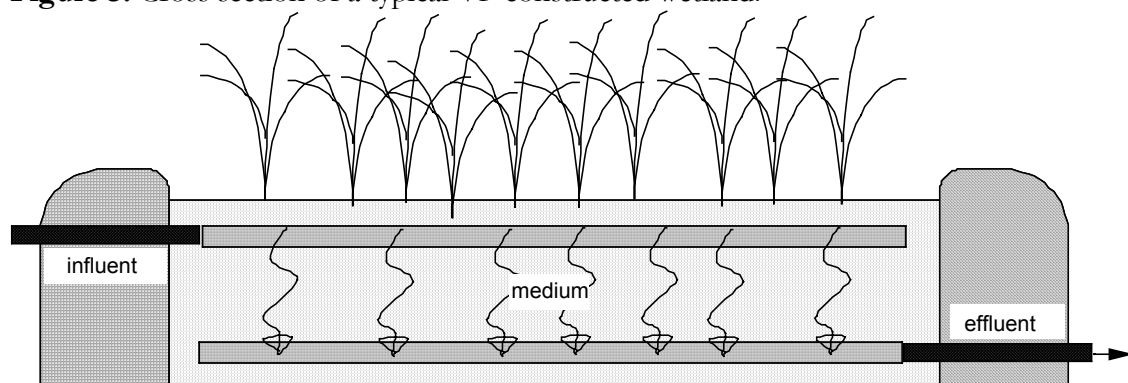
lack of exposed water prevents both *Lemna spp.* growth and ice cover, several other advantages of SF wetlands are realized. Odours are less likely to occur, mosquitoes cannot develop, and evaporation losses are low (Reed, 1986). Because of the large surface area of medium available for contact in a SF system, they generally require less land area than FWS systems for the same degree of treatment (EPA, 1988).

A critical difference between the two wetland types is the way in which oxygen is supplied to the system. FWS wetlands rely on diffusion of atmospheric oxygen as the primary oxygen source. SF wetlands depend on oxygen supplied by the roots of aquatic plants (Reed, 1986). Bowmer (1987) suggests that oxygen in SF systems may also be deficient and that greater oxygenation may be necessary to increase removal efficiencies.

### Section 1.3 — Vertical flow (VF) systems

The vertical flow (VF) system is a third type of constructed wetland that is currently under study. Much of the research on VF systems in Canada has been done in Niagara-On-The-Lake, Ontario. This type of system relies on intermittent applications of wastewater which is allowed to slowly percolate vertically down through a medium (Figure 3).

**Figure 3.** Cross section of a typical VF constructed wetland.



This downward percolation: 1) draws oxygen through the medium (Lemon et al, 1997; Bowmer, 1987), 2) allows for increased contact between medium and wastewater than presently offered by FWS and SF systems (Lemon *et al.*, 1997; Rogers *et al.*, 1991), and 3) reduces the creation of channels (or short circuiting) around the roots of aquatic plants which could minimize medium-wastewater contact (Bowmer, 1987; Bhamidimarri, 1991). In a five year research project at Niagara-On-The-Lake, Ontario, it has been shown that vertical flow systems are efficient in the treatment of domestic sewage even in a cold Canadian climate and that they meet most MOE discharge guidelines throughout the year (Lemon and Smith, 1993; Lemon *et al.*, 1997).

The VF constructed wetlands at Niagara-On-The-Lake are uniformly top-flooded through diffuser pipes buried 30 cm below the surface of the wetland cell. This allows the surface to freeze in winter and act as insulation permitting normal year-round operation. This subsurface flow also ensures the safety of the public, as there is no opportunity for wastewater/human contact. The following is a list of design features peculiar to the VF wetland at Niagara-On-The-Lake that are believed to contribute to their treatment efficiency:

- 1) Uniform sub-surface flooding at 30 cm below and throughout the top of the root-bed, at a rate of 60 L/m<sup>2</sup>/day in flooding pulses of 4 hour cycles.
- 2) Vertical complete drainage to the bottom of the bed between pulses, with a hydraulic conductivity of 10 cm/hr.
- 3) Root-bed medium of a fine shale or coarse limestone sand with a porosity near 40% and a 'working volume' of 10% airspace at field capacity.
- 4) A hydraulic residence time of 2 - 4 days through 2 or 3 beds in series.
- 5) Root-bed medium rich in aluminum, calcium and iron to fix phosphorus.
- 7) Cattail plants (*Typha spp*) to provide improved aeration and insulation against wintertime freezing. (Lemon *et al.*, 1997).

## Section 1.4 — Contaminant removal processes

Physical, chemical, and biological processes combine in wetlands to remove contaminants from wastewater. According to the Canadian Wildlife Service, an understanding of these processes is fundamental not only to designing wetland systems but to understanding the fate of chemicals once they have entered the wetland (Wren, 1997). Theoretically, treatment of wastewater within a constructed wetland occurs as it passes through the wetland medium and the plant rhizosphere. A thin aerobic film around each root hair is aerobic due to the leakage of oxygen from the rhizomes, roots, and rootlets (Hammer, 1989). Decomposition of organic matter is facilitated by aerobic and anaerobic micro-organisms present. Microbial nitrification and subsequent denitrification releases nitrogen as gas to the atmosphere. Phosphorus is co-precipitated with iron, aluminum, and calcium compounds located in the root-bed medium (Brix and Schierup, 1989; Davies and Hart, 1990; Fried and Dean, 1955; Patrick and Reddy, 1976; Sah and Mikkelsen, 1986). Suspended solids are filtered out as they settle in the water column in FWS wetlands or are physically filtered out by the medium within the SF or VF wetland cells. Harmful bacteria and viruses are reduced by filtration and adsorption by biological films on the rock media in SF and VF systems.

Physical attributes of wetland systems that are important in removal of contaminants are centered around the physical and chemical characteristics of the root-bed medium or soil. According to Webber (1990) a soil may be described as a porous medium consisting of finely divided organic and inorganic particles and water and gas-filled pores. Therefore a soil is a three phase system consisting of inorganic (or mineral) and organic solids, water and air. The proportions of each of these components in a soil will determine the physical properties of the soil and will have significant effect on plant growth. The mineral

component of most soils represents about one half of the volume but 95% of the weight of a moisture free soil sample. The mineral portion can be divided into three fractions: sand, silt or clay.

During the formation of clay particles an excess negative charge is acquired (Webber, 1990). The cation exchange capacity (CEC) of a soil is a measure of the concentration of cations required to neutralize all of the negative charges possessed by a unit weight of soil and is expressed in milliequivalents per 100 grams of soil (meq/100 g). The CEC of Ontario soils can range from 6 to 28 meq/100 g. The CEC of a soil is important in wastewater treatment because important nutrients such as  $\text{NH}_4^+$  can be removed via cation exchange (Faulkner and Richardson, 1989; Reed, 1995; Webber, 1990).

The ratio of voids to the total volume of soil is a measure of total porosity and refers to the total pore space available in a soil. In dry soils this pore space is filled with air. In wet soils this pore space is filled with air and water. The porosity of a soil determines the maximum amount of water that the soil can contain when it is saturated. Soils with a coarse texture (i.e. sand soils) are porous, have a low water holding capacity and drain rapidly. The opposite is generally true for finer, clay soils. The total porosity of a soil is expressed as cubic centimetres of air or water filled pores per cubic centimetre of soil ( $\text{cm}^3/\text{cm}^3$ ) Webber (1990).

Surface area of a soil is expressed as  $\text{m}^2/\text{g}$  and is the total surface area per unit weight of dry soil. Generally, the finer the soil, the larger its surface area. Because of the microscopic size of clay particles a unit weight of clay has a much larger surface area than does the same weight of sand or silt (Weber, 1990). A soil with a large surface area will provide many more attachment sites for bacteria than a soil with a small surface area (Steiner and Freeman, 1989). Webber (1990) estimates that one gram of soil may contain anywhere from 10 to 100 million bacteria.

The rate that water flows through soil is referred to as hydraulic conductivity and may be expressed as centimetres per unit time. In a saturated soil, water moves through the soil pores. When a soil is unsaturated water movement occurs via film flow of moisture from one particle to another. Coarse textured soils with larger pores have a faster hydraulic conductivity and *vice versa* (Reed, 1998; Webber, 1990).

The Ontario Ministry of Environment (MOE) assesses water quality by looking at some of the following :

- 1) Five Day Biochemical Oxygen Demand (BOD<sub>5</sub>)
- 2) Total Suspended solids (TSS)
- 3) Total Phosphorus (TP)
- 4) Nitrogen Compounds (For example: total nitrogen, ammonia, nitrates)

In addition, health departments may be interested in the level of total coliforms and fecal coliforms in areas where humans may come in contact with wastewater.

The following sections will define and provide detail on the biogeochemical processes that occur within constructed wetlands. Results obtained from various wetland treatment facilities located across North America will also be discussed.

## Section 1.5 — Five Day Biochemical Oxygen Demand (BOD<sub>5</sub>)

The amount of biodegradable organic contaminants in water, in either dissolved or suspended form is determined by the biochemical oxygen demand (BOD<sub>5</sub>) of that water. The BOD<sub>5</sub> is defined as the mass of dissolved oxygen required by a specific volume of liquid for the process of biochemical oxidation under prescribed conditions over 5 days at 20 °C in the dark. The result is then expressed in milligrams of oxygen per litre of sample (Mitsch, 1986). Simply put, BOD<sub>5</sub> is a physical measure of the oxygen consumed during the decomposition and mineralisation of organic and inorganic compounds.

### **BOD<sub>5</sub> removal in constructed wetlands used to treat domestic sewage**

In the literature the amount of organic matter in water that is applied to a constructed wetland is referred to as “ BOD<sub>5</sub> “ or “BOD<sub>5</sub> loading “. According to Reed (1995) the BOD<sub>5</sub> loading can be the limiting design factor for constructed wetland systems. Efficient BOD<sub>5</sub> removal can only be achieved when aerobic conditions are maintained within the wetland system (Lemon *et al.*, 1997). In wetland systems BOD<sub>5</sub> removal is facilitated by quiescent conditions, deposition, and filtration, with subsequent decomposition and oxidation of organic matter. In FWS systems the primary source of oxygen for these reactions is aeration of the water via diffusion of oxygen from the air into the water column and photosynthetic oxygenation (Reed *et al.*, 1996). SF systems rely on the transport of oxygen into the system via plant roots; VF systems rely on mass movement of oxygen into the system facilitated by intermittent flooding and draining cycles (Lemon *et al.*, 1997).

Perhaps the most efficient method of oxygenation has been demonstrated by Lemon *et al.*, (1997). These researchers found that excellent BOD<sub>5</sub> removal (97-99% in a three cell system) was obtained when VF wetland cells were pulse

flooded and maintained as 'free draining'. These authors reported that effluent concentrations were consistently less than 8 mg/L BOD<sub>5</sub> and as the system aged effluent levels of 1 mg/L BOD<sub>5</sub> were approached. In the FWS system at Listowel, Ontario, BOD<sub>5</sub> removal ranged from 51 - 76% (7.6 - 9.6 mg/L effluent concentration ). A similar system in Benton, Kentucky reported removals from 52 - 65% (8 - 11 mg/L effluent concentration ) (Watson *et al.*, 1989). SF systems are generally more efficient than FWS systems with respect to BOD<sub>5</sub> removal. A SF system operating in Santee, California, obtained 96% removal in a gravel bed planted with Bulrush ( 5.3 mg/L effluent concentration ). The same authors reported 94% removal in a similar system operating in Ringsted, Denmark (11 mg/L effluent concentration) (Watson *et al.*, 1989).

#### **Section 1.6 — Suspended Solids (SS)**

The suspended solids component of wastewater consists of organic and inorganic particles suspended in the water column. Removal of SS is effective in all three types of wetlands discussed.

#### **SS removal in constructed wetlands used to treat domestic sewage**

FWS wetlands rely on the gravitational settling of suspended solids. SF and VF wetlands rely on the filtration of these particles by the sand or gravel medium within the wetland cells (Watson *et al.*, 1989). In the FWS system at Listowel, Ontario removals of 60 - 93% were recorded (8 - 9.2 mg/L effluent concentration ). The FWS system at Benton, Kentucky reported removals from 67 - 88% (7 - 20 mg/L effluent concentration ) (Watson *et al.*, 1989). The SF system operating at Santee, California obtained 94% suspended solids removal in the gravel bed planted with Bulrush ( 3.7 mg/L effluent concentration ). Lemon *et al.* (1997) reported an overall average of less than 5 mg/L suspended solids in

the effluent from a three cell VF system with a removal efficiency of 98%.

### Section 1.7 — Nitrogen

The dominant forms of nitrogen in wetlands that are of importance to wastewater treatment include organic nitrogen, ammonia, ammonium, nitrate, nitrite, and nitrogen gases. Inorganic forms are essential to plant growth in aquatic systems but if scarce can limit or control plant productivity (Mitsch & Gosselink, 1993). The nitrogen entering wetland systems can be measured as organic nitrogen, ammonia, nitrate and nitrite. Total Nitrogen refers to all nitrogen species.

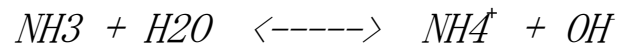
The removal of nitrogen from wastewater is important because of ammonia's toxicity to fish if discharged into water courses. Excessive levels of nitrates in drinking water is thought to cause methaemoglobinaemia in infants, which decreases the oxygen transport ability of the blood. The UK has experienced a significant increase in nitrate concentration in groundwater and rivers (Gray, 1989).

#### Organic Nitrogen

Mitsch & Gosselink (1986) define nitrogen mineralisation as "the biological transformation of organically combined nitrogen to ammonium nitrogen during organic matter degradation". This can be both an aerobic and anaerobic process and is often referred to as ammonification. Mineralisation of organically combined nitrogen releases inorganic nitrogen as nitrates, nitrites, ammonia and ammonium, making it available for plants, fungi and bacteria (Mitsch & Gosselink, 1986). Mineralisation rates may be affected by oxygen levels in a wetland (Patrick & Reddy, 1976).

Ammonia ( $\text{NH}_3$ ) and Ammonium ( $\text{NH}_4^+$ )

The formation of ammonia (NH<sub>3</sub>) occurs via the mineralisation or ammonification of organic matter under either anaerobic or aerobic conditions (Keeney, 1973). The ammonium ion (NH<sub>4</sub><sup>+</sup>) is the primary form of mineralized nitrogen in most flooded wetland soils. The formation of this ion occurs when ammonia combines with water as follows:



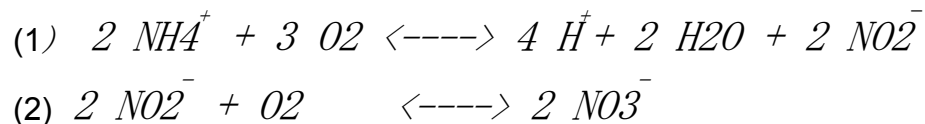
(Mitsch & Gosselink, 1986)

Upon formation, several pathways are available to the ammonium ion. It can be absorbed by the plants and algae and converted back into organic matter, or the ammonium ion can be immobilized onto negatively charged soil particles (Mitsch & Gosselink, 1986). At this point, the ammonium ion can be prevented from further oxidation because of the anaerobic nature of wetland soils. Under these conditions the ammonium ion is stable and it is in this form that nitrogen predominates in anaerobic sediments typical of wetlands (Brock & Madigan, 1991; Patrick & Reddy, 1976).

Most wetland soils have a thin aerobic layer at the surface. As an ammonium ion from the anaerobic sediments diffuses upward into this layer it is converted to nitrite or nitrified (Klopatek, 1978). An increase in the thickness of this aerobic layer results in an increase in nitrification (Patrick & Reddy, 1976). This diffusion of the ammonium ion sets up a concentration gradient across the aerobic-anaerobic soil layers resulting in further nitrification reactions (Klopatek, 1978; Patrick & Reddy, 1976).

## Nitrification

Wetzel (1983) defines nitrification as the "biological conversion of organic and inorganic nitrogenous compounds from a reduced state to a more oxidized state". Nitrification is strictly an aerobic process in which the end product is nitrate ( $\text{NO}_3^-$ ); this process is limited when anaerobic conditions prevail (Patrick & Reddy, 1976). Nitrification will occur readily down to 0.3 ppm dissolved oxygen (Keeney, 1973). The process of nitrification (1) oxidizes ammonium (from the sediment) to nitrite ( $\text{NO}_2^-$ ), and then (2) nitrite is oxidized to nitrate ( $\text{NO}_3^-$ ). The overall nitrification reactions are as follows:

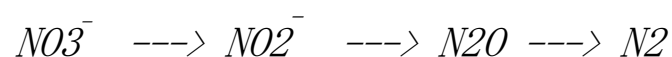


(Davies & Hart, 1990)

Two different bacteria are required to complete this oxidation of ammonium to nitrate. *Nitrosomonas sp.* oxidizes ammonium to nitrite via reaction (1), and *Nitrobacter sp.* oxidizes nitrite to nitrate via reaction (2) (Keeney, 1973).

## Denitrification

According to Wetzel (1983) "Denitrification by bacteria is the biochemical reduction of oxidized nitrogen anions, nitrate-N and nitrite-N, with concomitant oxidation of organic matter." The general sequence as given by Wetzel (1983) is as follows:



The end products,  $\text{N}_2\text{O}$  and  $\text{N}_2$  are gases that re-enter the atmosphere.

Denitrification occurs intensely in anaerobic environments but will also occur in aerobic conditions (Bandurski, 1965). A deficiency of oxygen causes certain bacteria to use nitrate in place of oxygen as an electron acceptor for the reduction of organic matter (Patrick & Reddy, 1976).

The process of denitrification is restricted to a narrow zone in the sediment immediately below the aerobic-anaerobic soil interface (Mitsch & Gosselink, 1986; Nielson *et al.*, 1990). Denitrification is considered by Richardson *et al.* (1978) to be the predominant microbial process that modifies the chemical composition of nitrogen in a wetland system and the major process whereby elemental nitrogen is returned to the atmosphere (Patrick & Reddy, 1976).

To summarize, the nitrogen cycle is completed as follows: ammonia in water, at or near neutral pH is converted to ammonium ions; the aerobic bacterium *Nitrosomonas sp.* oxidizes ammonium to nitrite; *Nitrobacter sp.* then converts nitrite to nitrate. Under anaerobic conditions, nitrate is reduced to relatively harmless nitrogen gas, that is given off to the atmosphere.

#### **Nitrogen removal in constructed wetlands used to treat domestic sewage**

In a review of 19 FWS wetlands (US EPA, 1988) it was found that nearly all reduced total nitrogen. In a review of both FWS and SF wetlands Reed (1995) concluded that effluent nitrate concentration is dependent on maintaining anoxic conditions within the wetland so that denitrification can occur. He found that SF wetlands were superior to FWS wetlands for nitrate removal. The 20 FWS wetlands reviewed reported effluent nitrate levels below 5 mg/L; the 12 SF wetlands reviewed reported effluent nitrate ranging from <1 to < 10 mg/L. Results obtained from the Niagara-On-The-Lake VF systems show a significant reduction in both total nitrogen and ammonia (> 97%) when primary treated effluent was applied at a rate of 60L/m<sup>2</sup>/day. Calculations made showed that over

50% of the total nitrogen going into the system was converted to relatively harmless nitrogen gas. Effective removal of nitrate from the sewage lagoon influent was dependent on medium type used within the VF cell as well as water table level within the cell (Lemon *et al.*, 1997).

### Section 1.8 — The Phosphorus Cycle

Phosphorus occurs naturally in both organic and inorganic forms. Table 1 shows some of these forms. The analytical measure of biologically available orthophosphates is referred to as soluble reactive phosphorus (SR-P). Dissolved organic phosphorus and insoluble forms of organic and inorganic phosphorus are generally not biologically available until transformed into soluble inorganic forms (Mitsch and Gosselink, 1986).

**Table 1.** Phosphorus compounds present in freshwater ecosystems.

phosphorus	soluble forms	insoluble forms
inorganic	orthophosphates ( $H_2PO_4^-$ , $HPO_4^{2-}$ , $PO_4^{3-}$ )	clay-phosphate complexes
	ferric phosphate ( $FeHPO_4^+$ )	metal hydroxide-phosphate minerals, e.g., apatite ( $Ca_{10}(OH)_2(PO_4)_6$ )
	calcium phosphate ( $CaH_2P_2O_7$ )	
organic phosphates, phospholipids	dissolved organics, e.g., sugar phosphates, inositol organic matter	insoluble organic phosphorus bound in

Source: after Mitsch and Gosselink, 1986.

In freshwater aquatic ecosystems phosphorus has been described as the major limiting nutrient. Under undisturbed natural conditions, phosphorus is in short supply. The natural scarcity of phosphorus is demonstrated by the explosive growth of algae in water receiving heavy discharges of phosphorus-rich wastes.

Because phosphorus does not have an atmospheric component as does nitrogen, the phosphorus cycle can be characterized as closed. The removal and storage of phosphorus from wastewater can only occur within the constructed wetland itself. According to Mitsch and Gosselink (1986) phosphorus may be sequestered within a wetland system by the following:

- 1) The binding of phosphorus in organic matter as a result of incorporation into living biomass, and
- 2) precipitation of insoluble phosphates with ferric iron, calcium, and aluminum found in wetland soils.

#### **Incorporation into biomass**

Higher plants in wetland systems may be viewed as transient nutrient storage compartments absorbing nutrients during the growing season and releasing large amounts at senescence (Bernard and Solsky, 1976; Guntensbergen, 1989). Generally, plants from nutrient-rich habitats accumulate more nutrients than plants found in nutrient-poor habitats, a phenomenon referred to as luxury uptake of nutrients (Guntensbergen, 1989; Kadlec, 1989). Aquatic vegetation may play an important role in phosphorus removal and, if harvested, extend the life of a system by postponing phosphorus saturation of the sediments (Breen, 1990; Guntensbergen, 1989; Rogers *et al.*, 1991).

According to Sloey *et al.* (1978) vascular plants may account for only a small amount of phosphorus uptake with only 5 to 20% of the nutrients detained in a natural wetland being stored in harvestable plant material. Bernard and Solsky (1976) also reported relatively low phosphorus retention, estimating that a sedge (*Carex sp.*) wetland retained 1.9 g of phosphorus per square metre of wetland. Bulrushes (*Scirpus sp.*) in a constructed wetland system receiving secondarily treated domestic wastes contained 40.5% of the total phosphorus

influent. The remaining 59.0% was found to be stored in the gravel substratum (Sloey *et al.*, 1978). Phosphorus removal in a FWS wetland treatment system planted with one of *Scirpus* sp., *Phragmites* sp. or *Typha* sp. was investigated by Finlayson and Chick (1983). Phosphorus removal of 60%, 28%, and 46% were found for *Scirpus* sp., *Phragmites* sp. and *Typha* sp. respectively. More recent work by Breen (1990) may prove this to be a low estimate. His work on an artificial wetland indicated that vascular plants are a major phosphorus storage compartment accounting for 67.3% of the influent phosphorus. Thut (1989) attributed plant adsorption with 80% phosphorus removal.

Only a small proportion (<20%) of phosphate removal by constructed wetlands can be attributed to nutritional uptake by bacteria, fungi and algae (Moss, 1988). Swindell *et al.*, (1990) found that the lack of seasonal fluctuation in phosphorus removal rates suggests that the primary mechanism is bacterial and alga fixation. However, Richardson (1985) dismisses this mechanism as temporary saying that although the initial removal of dissolved inorganic phosphorus from the water under natural loading levels is due largely to microbial uptake and adsorption, the microbial pool is small and quickly becomes saturated at which point the soil medium takes over as the major contributor to phosphate removal.

There are more indirect ways in which plants contribute to wastewater purification. Plants create a unique environment at the attachment surface of the biofilm. Certain plants transport oxygen which is released at the biofilm/root interface perhaps adding oxygen to the wetland system (Pride *et al.*, 1990). Plants also increase soil or other root-bed medium hydraulic conductivity. As roots and rhizomes grow they are thought to disturb and loosen the medium increasing its porosity which may allow more effective fluid movement in the rhizosphere. When roots decay they leave behind pores and channels known as

macropores which are effective in channeling water through the soil (Conley *et al.*, 1991).

Whether or not wetland systems act as a phosphorus sink or source seems to depend on system characteristics such as sediment and hydrology. Kramer *et al.*, (1972) indicated that there seems to be a net movement of phosphorus into the sediment in many lakes. In Lake Erie as much as 80% of the total phosphorus is removed from the waters by natural processes and is presumably stored in the sediment. According to Klopatek (1978) marsh sediments high in organic matter act as sinks. He has also shown that phosphorus release from a marsh exhibits a cyclical pattern. Much of the spring phosphorus release comes from high phosphorus concentrations locked up in the winter ice covering the marsh; in summer the marsh acts as a phosphorus sponge. Simpson (1978) found that phosphorus was exported from the system following dieback of vascular plants. It has been demonstrated by Klopatek (1978) that phosphorus concentrations in water are reduced during the growing season due to plant uptake but decomposition and subsequent mineralisation of organic matter releases phosphorus over the winter and accounts for the higher winter phosphorus concentrations in the marsh (Klopatek, 1978; Mitsch, 1986).

#### **Phosphorus retention by soils or root-bed media**

Two types of phosphate retention mechanisms may occur in soils or root-bed media: chemical adsorption onto the medium (Hsu, 1964) and physical precipitation of the phosphate ion (Faulkner and Richardson, 1989). Both result from the attraction between phosphate ion and ions of Al, Fe or Ca (Hsu, 1964; Cole *et al.*, 1953) and terminates with formation of various iron phosphates (Fe-P), aluminum phosphates (Al-P) or calcium phosphates (Ca-P) (Fried and Dean, 1955).

Redox potential (Eh) of soil or water is a measure of its ability to reduce or

oxidize chemical substances and may range between -300 and +300 millivolts (mV) (Hammer, 1992). Though the oxidation state of phosphorus is unaffected by redox reactions, the redox potential is important because of Fe reduction. Severely reduced conditions in the sediments may result in phosphorus release (Mann, 1990). Typical wetland soils may have an Eh of -200 mV (Hammer, 1992). Under these reduced conditions  $\text{Fe}^{3+}$  (Ferric iron) may be reduced to  $\text{Fe}^{2+}$  (Ferrous iron) and may release the bound phosphate ion back into solution (Faulkner and Richardson, 1989; Sah and Mikkelsen, 1986). The introduction of oxygen causes the  $\text{Fe}^{2+}$  to be oxidized to  $\text{Fe}^{3+}$  producing a simultaneous reduction of phosphate (Wetzel, 1983). The solubility of phosphorus may be affected by the amount of oxygen present in the sediment because saturation by water and subsequent loss of oxygen generally cause wetland soils to have negative redox potentials (Hammer, 1992). A well documented occurrence in the hypolimnion of lakes is the release of soluble phosphorus when conditions become anaerobic (Burns & Ross, 1972; Williams & Mayer, 1972). This phenomenon also occurs in natural wetlands (Gosselink & Turner, 1978) and Kramer *et al.*, (1972) report that oxygen concentrations of less than 2.0 mg/l result in the release of phosphorus from sediments.

#### **Phosphorus removal in constructed wetlands used to treat domestic sewage**

Adsorption to binding sites within the sediments was identified as the major phosphorus removal mechanism in the FWS constructed wetland system at Port Perry, Ontario (Snell, unpublished data). Release of phosphorus from the sediments occurred when anaerobic conditions prevailed. The lowest wetland effluent phosphorus levels occurred when oxygen levels of the overlying water column were above 1.0 mg / L. Removal efficiencies for total phosphorus were 54-59% with mean effluent levels of 0.38 mg P/L. Wetland effluent phosphorus concentration was higher than influent levels during the winter months.

Lantzke *et al.*, (1999) investigated phosphorus removal in a VF wetland in Australia and found that the quantity of phosphorus removed over a short term was stored in the following wetland components in order of decreasing importance: substratum > macrophyte > biofilm but over the long term phosphorus storage was located in macrophyte > substratum > biofilm components. They also found that medium iron-oxide adsorption provides additional removal for some years.

Mann (1990) investigated the phosphorus removal efficiency of two large-scale, SF wetland systems in Australia which had a gravel substratum. He then compared these results to laboratory phosphorus adsorption experiments. For the first two months of wetland operation the mean phosphorus removal efficiency of system 1 and 2 was 38% and 22%, respectively. Over the first year a decline in removal efficiencies occurred. During the second year of operation release of phosphorus from the system was often recorded such that more phosphorus came out than was put in. This release was attributed to the saturation of phosphorus binding sites. Close agreement between the phosphorus adsorption capacity of the gravel as determined in the laboratory and the adsorption capacity recorded in the field was found.

The phosphorus adsorption capacity of an SF constructed wetland system containing a predominantly quartz gravel was investigated by Breen (1990). The adsorption characteristics of this gravel as determined by laboratory adsorption experiments and using the Langmuir adsorption isotherm was 25 mg P / g gravel. Close agreement between calculated and realized phosphorus adsorption was found. Because of the poor adsorption capacity of the quartz gravel, plant uptake and subsequent harvesting were identified as the major phosphorus removal mechanism.

Sequestering of phosphorus by the root-bed medium is most probably

dependent not only on medium characteristics but also on the dissolved oxygen content of the fluid passing through the medium. The VF system at Niagara-On-The-Lake, Ontario did remove significant amounts of total phosphorus from domestic sewage ( 48 to 95% reduction or 1.5 - 0.18 mg/L effluent concentration). This study concluded that phosphorus can be removed most efficiently by allowing the cells to drain freely which maintained high dissolved oxygen concentrations and by selecting root-bed media with a large clay surface area high in Fe, Al, and Ca.

## Section 2.0 — Purpose

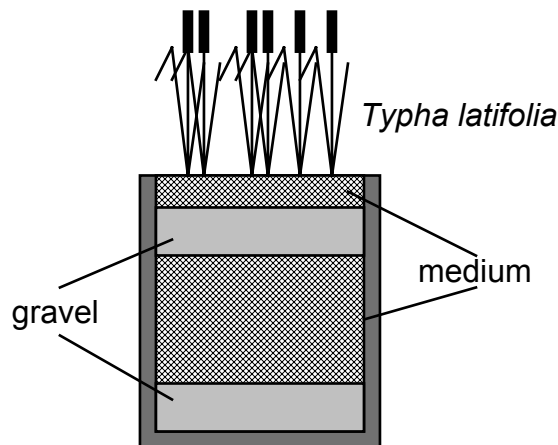
Lemon *et al.* (1996) have already established that a vertical flow constructed wetland sewage treatment system can be made to operate effectively throughout the year in a cold climate. They have also established some general operational and design criteria that, if employed, will enhance the treatment efficiency of these systems. The purpose of this research is to add to this body of knowledge by investigating the influence of root-bed media with different physical and chemical properties on the treatment of primary treated domestic sewage as it passes through a vertical flow constructed wetland system. The reduction of the nutrients nitrogen and phosphorus and reduction of suspended solids and organic matter are assessed by monitoring influent and effluent concentrations. In addition, the role of plants in reducing these parameters is investigated.

### Section 3.0 — Materials and methods

On November 12, 1994 a subsurface, vertical flow constructed wetland experiment was begun consisting of ten cells in all. These ten cells were operated in sets with two cells in each set resulting in five sets of cells. Due to budget constraints no replicates were constructed.

Each cell was made of pre-cast concrete with inside dimensions measuring 1.0m X 1.0m X 1.30m in height ( Figure 7). Layers of root-bed medium and gravel were incorporated into each cell as shown in figure 4.

**Figure 4.** Cross sectional view of the experimental wetland cell showing the location of the gravel and medium layers as well as the location of the plants, *Typha latifolia*.



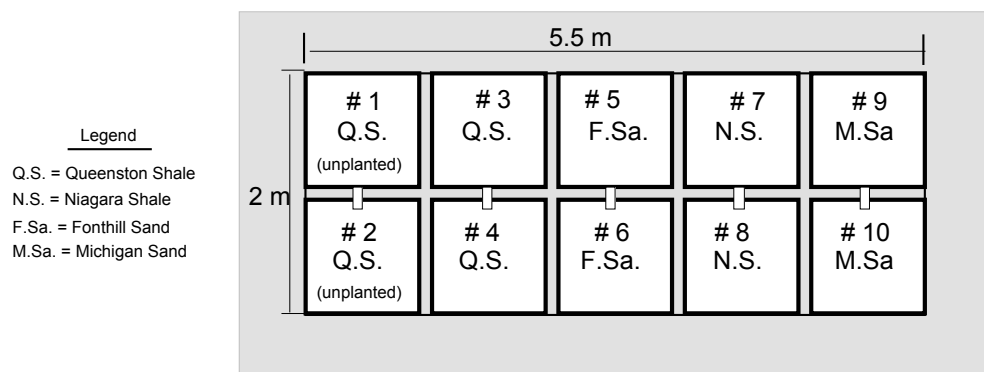
The entire experimental apparatus was situated above ground and protected from freezing by an insulated plywood enclosure (Figure 5). During winter the temperature of this enclosure was kept at above 4 ° C. No attempt was made to regulate summer temperatures.

The experimental layout is presented in Figure 5. The first two sets of cells (cells 1 & 2 and 3 & 4) contained identical Queenston Shale medium . Cells 3 & 4 were planted with the common Cattail, *Typha latifolia*; cells 1 & 2 remained un-

planted. The top of both unplanted cells was insulated with 20 cm of straw in the autumn. This straw was removed in early spring. The two Queenston Shale sets were used to test the influence of plants on the treatment of domestic sewage. The other three sets were planted with *Typha latifolia* and contained Fonthill Sand (cells 5 & 6), Niagara Shale (cells 7 & 8) and Michigan Sand (cells 9 & 10). These sets were used to test the influence of medium type on the treatment of domestic sewage. All five sets of cells were operated identically with regard to wastewater loading, flow direction and sampling.

Data for one full year, collected between March 20, 1996 and March 20, 1997 are presented and analyzed in this thesis. During the time period these data were collected no parameters of the experiment were changed, i.e. feed source or hydraulic loading rate.

**Figure 5.** Top view of the cell layout in the constructed wetland experiment. The shaded area represents the heated gallery enclosure used to prevent the plumbing from freezing during winter months.



### *Hydraulic loading and flow*

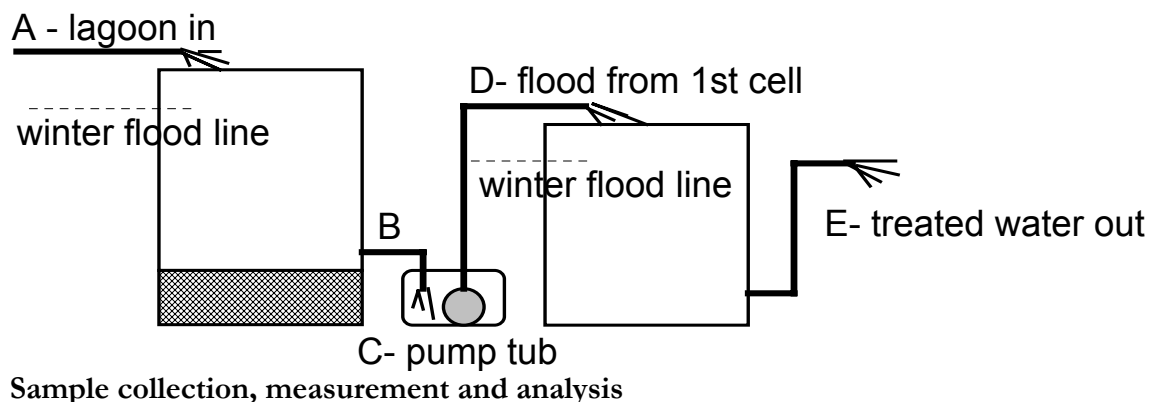
Each set of cells was flooded with domestic sewage that had received primary treatment in sewage lagoons located on the experimental site. The flooding rate was calculated to be 120 L/m<sup>2</sup> of set surface area/day applied to the top of the first cell in each set (Figure 6, A). Each set had a surface area of 2 m<sup>2</sup>, therefore the first cell in each set received 240 L of sewage per day (120 L/day X

2.0 m<sup>2</sup> = 240 L/m<sup>2</sup>/day). Flooding was controlled by an electronic timer in timed pulses every 6 hours. This arrangement allowed the first cell to drain completely before the next flood. The pumps and timers were checked monthly to ensure consistent hydraulic loading. During the period reported on in this thesis no adjustment to either the pumps or timer was required.

Water levels were controlled by simply raising or lowering the drain hose connected to the exit fitting of each cell (Figure 6, E). The water level in the first cell in each set was maintained at 70 cm below surface creating a freely draining cell environment. Effluent from the bottom of the first cell in a set was collected in a plastic tub (Figure 6, B). Located within this tub was an electric pump controlled by a mercury float switch (Figure 6, C). When the water level in the tub reached a critical level, the mercury float switch activated the pump, pumping the water to the top of the second cell in the set (Figure 6, D). Water in this second cell drained vertically down through the cell and was discharged from the bottom. The water level in the second cell in each set was maintained at 30 cm below surface creating saturated conditions within the cell.

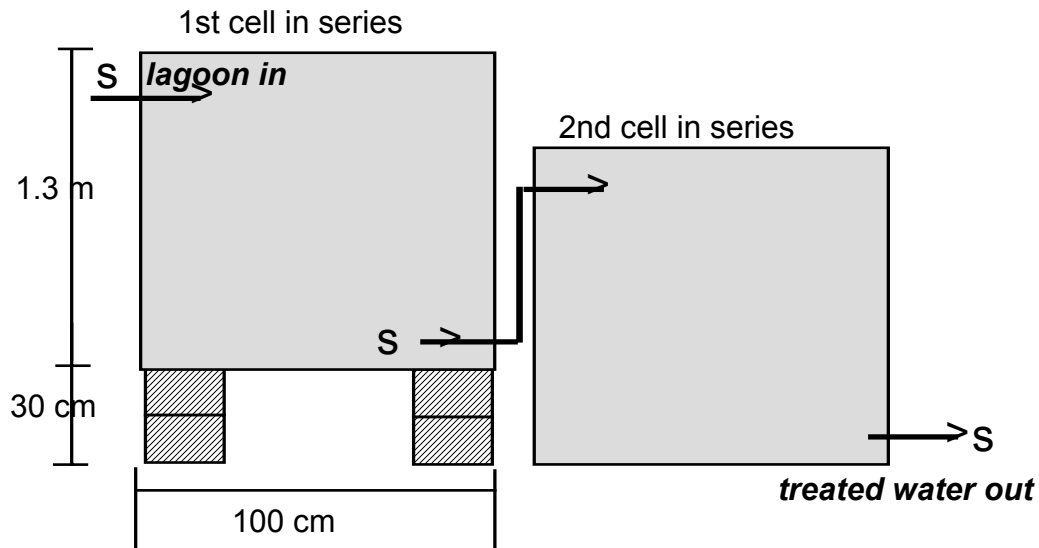
During the winter months all flooding was done from 30 cm below the cell surface via a buried 10 cm perforated pipe. This arrangement allowed the experiment to operate successfully all year round.

**Figure 6.** Side view of one two cell set showing flooding, pumping and draining arrangements.



Water samples were drawn every second week and were collected at points marked 'S' as shown in Figure 7.

**Figure 7** Sideview of a set of cells showing cell arrangement and size, flow direction and sampling points (S) in each set.



Water samples were sent to the MOE lab in Etobicoke to be analyzed for five day biochemical oxygen demand ( $BOD_5$ ), suspended solids, total nitrogen, ammonia and total phosphorous. These analyses were performed in accordance with the MOE's procedure F-10-1 "Procedure for sampling and analysis requirements for Municipal and Private Sewage Treatment Works" Ministry of Environment and Energy, December 31, 1994.

Cell influent and effluent water samples were collected in 500 mL bottles and measured on-site for temperature using a portable Horiba ES-12 conductivity and temperature meter. Dissolved oxygen was measured on-site in  $BOD_5$  bottles using a self-stirring Orion Model 850 dissolved oxygen and  $BOD_5$  meter. Sample pH was measured on-site using the pH probe for the Orion Model 850 dissolved oxygen and  $BOD_5$  meter.

Hydraulic conductivity of each medium type was measured directly from

the second cell of each series. Each cell was flooded to about 5 cm depth of fluid over the cell surface. They were then drained from the bottom in increments measuring the volume drained over time and depth of the water table in the cell at the end of each drain period. Water table depth was determined using a calibrated peiziometer tube connected to the drain valve of each cell. The data collected were then used to calculate the hydraulic conductivity of each medium.

### **Data Analysis**

All of the data were analyzed using the statistical analysis software *Microcal Origin*, version 3.5. The Student's T-test was used to analyze the data. Independent Student's T-tests were used to compare data collected from different sets of cells and dependent Student's T-tests were used to compare data collected from the first and second cells in the same set. The data were analyzed at the 95% level of confidence. For all of the data analyzed N = 25.

### **Media Characteristics**

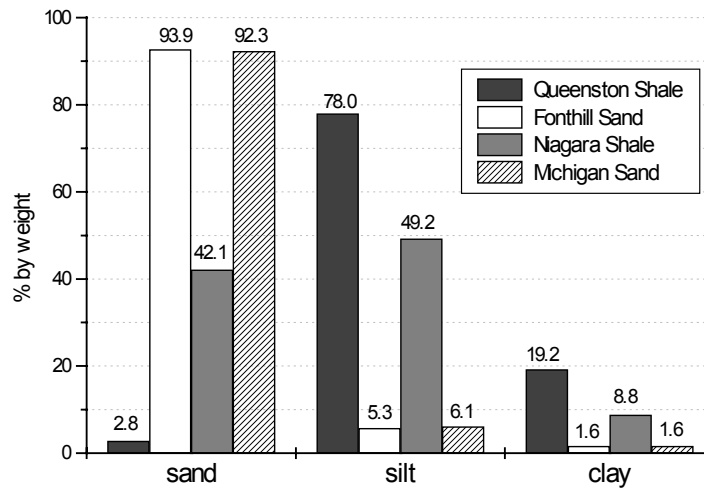
Four different root-bed media were tested : Queenston Shale, Niagara Shale, Fonthill Sand and Michigan Sand. An extensive investigation into the physical and chemical characteristics of this media was made by Lemon *et al.*, (1996). The results of this work are reported in this thesis.

A textural analysis of these media was done by the Analytical Services Laboratory, University of Guelph, Ontario, Canada. The results (Figure 8) show that the Queenston Shale consists of 78.0% silt sized particles, 19.2% clay and 2.8% sand sized particles. The Fonthill and Michigan sand media were found to be primarily sand, 93.9% and 92.3%, respectively, and both had similar amounts

of silt and clay sized particles. The Niagara Shale contained 42.1% sand and

49.2% silt sized particles, with 8.8% clay sized particles.

**Figure 8.** Textural analysis of the Queenston Shale, Fonthill Sand, Niagara Shale, & Michigan Sand media. (Analysis by the University of Guelph, Department of Land Resource Science, Analytical Services Laboratory, Ontario, Canada.).



Some physical characteristics of the root-bed media are presented in Table 2. The Queenston and Niagara Shales have similar porosities which are somewhat higher than the two sand media which also have similar porosities. No data on the cation exchange capacity of the Michigan Sand was available. The Queenston and Niagara Shales both have identical (and higher) cation exchange capacities than the Fonthill Sand.

**Table 2.** Physical and chemical characteristics of the four different root-bed media used in the experiment (*Analysis by the University of Guelph, Department of Land Resource Science, Analytical Services Laboratory, Ontario, Canada*).

	<b>Queenston Shale</b>	<b>Fonthill Sand</b>	<b>Niagara Shale</b>	<b>Michigan Sand</b>
<b>Bulk Density (g/cm<sup>3</sup>)</b>	1.59	1.56	1.64	1.55
<b>Total Porosity (cm<sup>3</sup>/cm<sup>3</sup>)</b>	0.42	0.36	0.43	0.31
<b>Field Capacity (cm<sup>3</sup>/cm<sup>3</sup>)</b>	0.19	0.33	0.08	0.23
<b>surface area (m<sup>2</sup>/gm)</b>	93	53	18	3
<b>Cation exchange capacity (cmol +/kg)</b>	8.7	4.7	8.1	NA
<b>% CaCO<sub>3</sub> (by weight)</b>	4.3	10.4	52.9	0.5

Figure 9 presents data on the phosphorous adsorption capacities of the different root-bed media. No standard errors for these data are available. The data shown in Figure 9 were obtained by using a solution with a phosphorous concentration similar to the lagoon effluent used in the experiment i.e. 5 mg/L. The Niagara Shale has the highest phosphorous adsorption capacity followed in descending order by the Queenston Shale and Fonthill Sand. The phosphorous adsorption capacity of the Michigan Sand is considerably lower than the other three media.

**Figure 9.** P adsorption (mg P /Kg media) of the four media used in the experiment obtained using the Langmuir Adsorption Isotherm. The results presented were obtained at an initial P concentration of 5 mg /L (*Analysis by Dr. K.R. Reddy, University of Florida, Soil and Water Science Department.*).

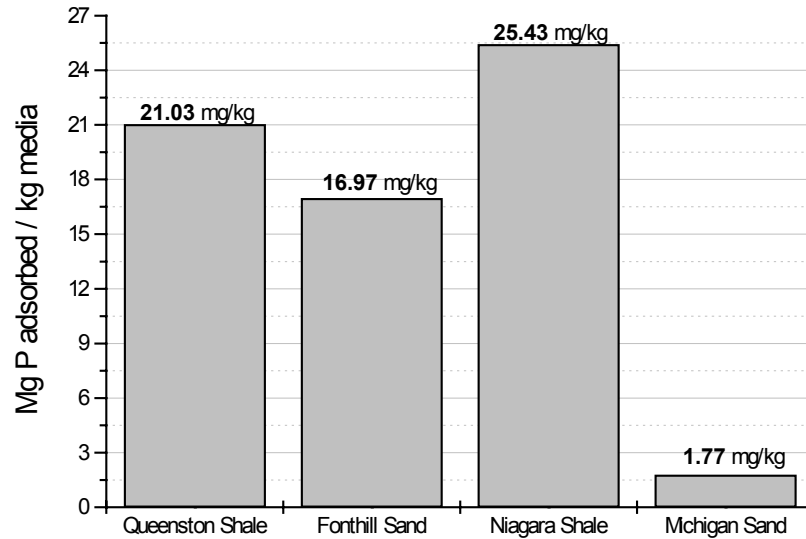
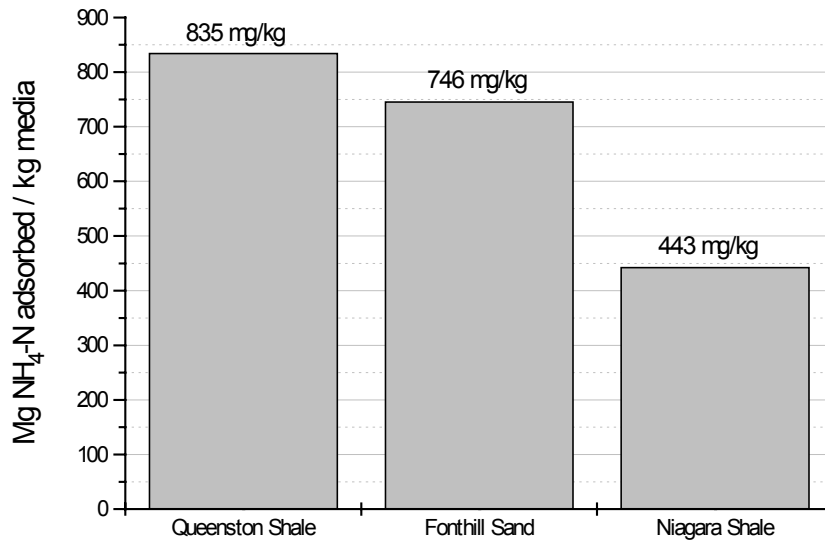


Figure 10 presents data on the ammonium adsorption capacities of three of the four different root-bed media. No standard errors for these data are available. The Michigan Sand did not fit the Langmuir adsorption isotherm and therefore is not included. The Queenston Shale was able to adsorb 846 mg/kg of ammonium. The Fonthill Sand adsorbed 735 mg/kg ammonium, while the Niagara Shale adsorbed 443 mg of ammonium per kg of medium.

**Figure 10.**  $\text{NH}_4\text{-N}$  adsorption (mg  $\text{NH}_4\text{-N}$  /Kg media) of three of the four media used in the experiment obtained using the Langmuir Adsorption Isotherm. (Analysis by the University of Guelph, Department of Land Resource Science, Analytical Services Laboratory, Ontario, Canada).

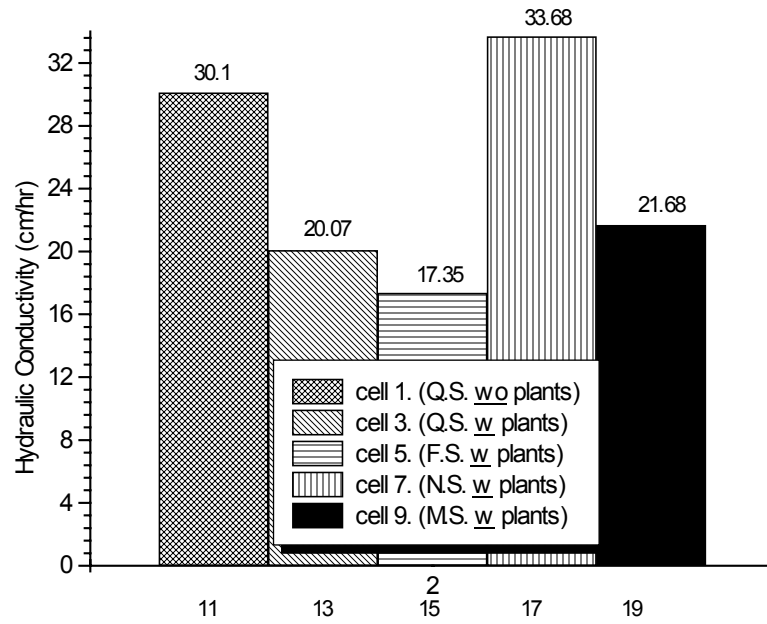


### *Hydraulic Conductivity*

Hydraulic drainage studies were done on cells 1, 3, 5, 7 & 9. Hydraulic conductivity is synonymous with permeability and is defined as the volume of fluid drawn down over a fixed time expressed in centimetres per hour (cm/hr). The results of these studies are presented in Figure 11.

According to the Soil Conservation Service (SCS) classification, all of the cells drained "rapidly" however there were differences between the different root-bed media. The Queenston Shale without plants and the Niagara Shale with plants drained most rapidly; the Queenston Shale with plants drained more slowly in a similar fashion to the Fonthill and Michigan Sand cells with plants.

**Figure 11.** Hydraulic Conductivity of cells 1, 3, 5, 7 & 9 expressed in centimetres per hour.



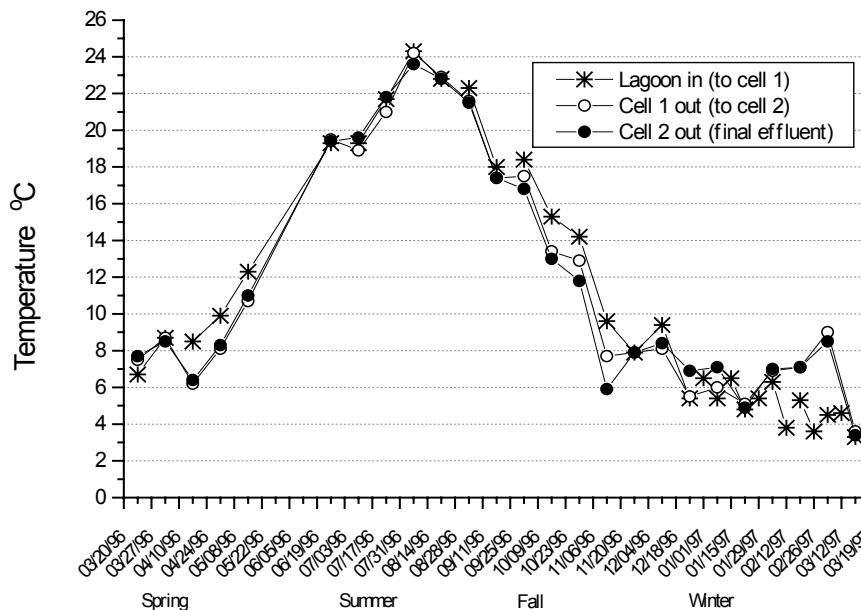
## Section 4.0 — Results

### Section 4.1 — Temperature (°C)

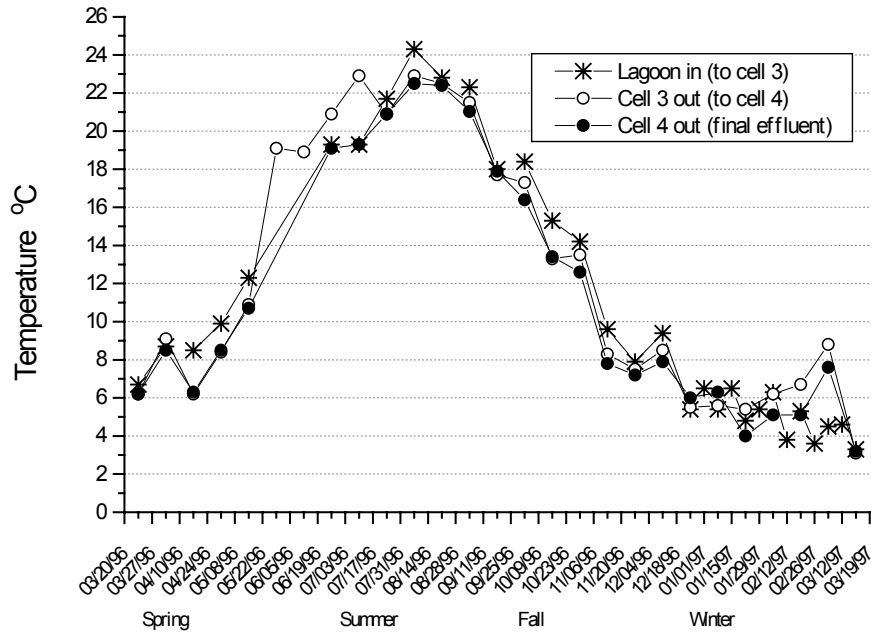
Temperature data on lagoon flood water and individual cell effluent water are shown in Figures 12 through 16. Table 3 presents the results of Student's t-tests comparing these data. These figures show that the influent to each cell and effluent from each cell exhibit a seasonal variation in temperature. Summer temperatures rise past 22 °C while winter temperatures go as low as 3 °C.

Water temperatures of each cell in each set closely follow the temperature of the influent water to that cell. Table 3 shows that no statistical difference in water temperature exists among the different cells of the experiment or the lagoon influent water.

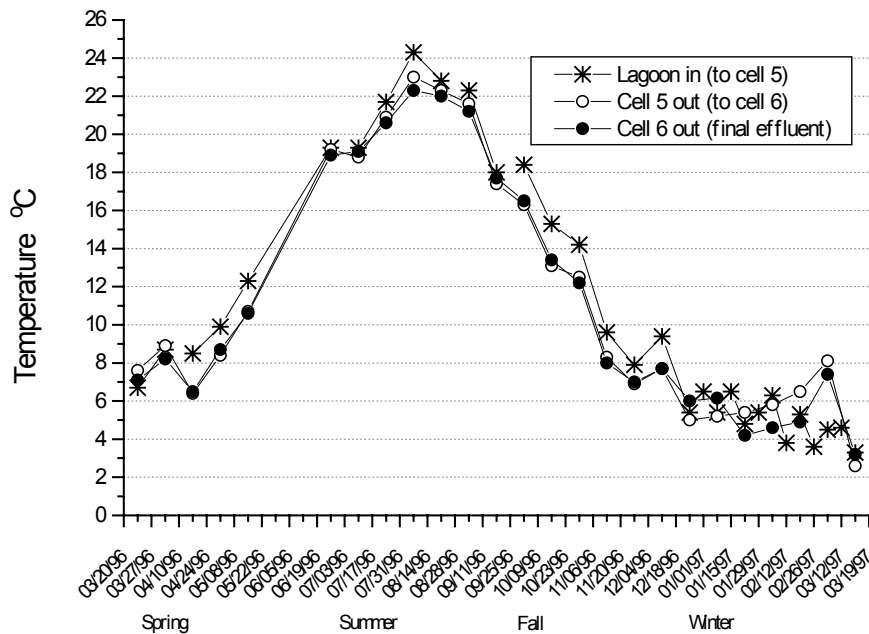
**Figure 12.** Time series plot of lagoon influent temperature and Queenston Shale cells 1 & 2 (without plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.



**Figure 13.** Time series plot of lagoon influent temperature and Queenston Shale cells 3 & 4 (with plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.

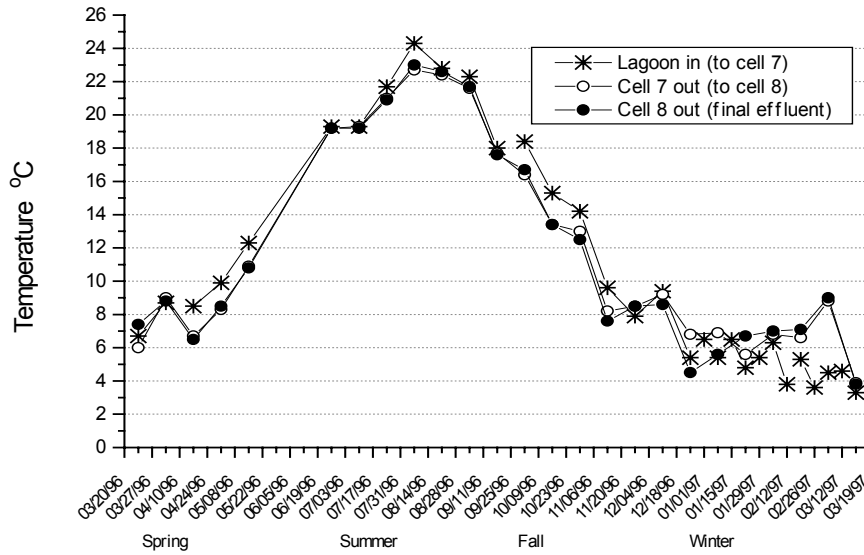


**Figure 14.** Time series plot of lagoon influent temperature and Fonthill Sand cells 5 & 6 (with plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.

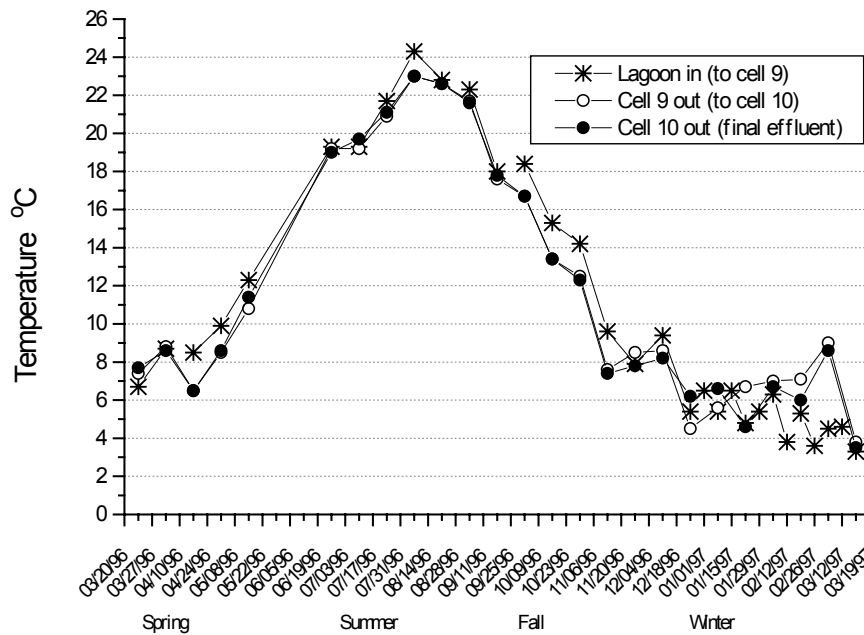


**Figure 15.** Time series plot of lagoon influent temperature and Niagara Shale cells 7 & 8 (with plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.

plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.



**Figure 16.** Time series plot of lagoon influent temperature and Michigan Sand cells 9 & 10 (with plants) effluent temperature from March 20, 1996 to March 19, 1997. All values reported in degrees Celsius.



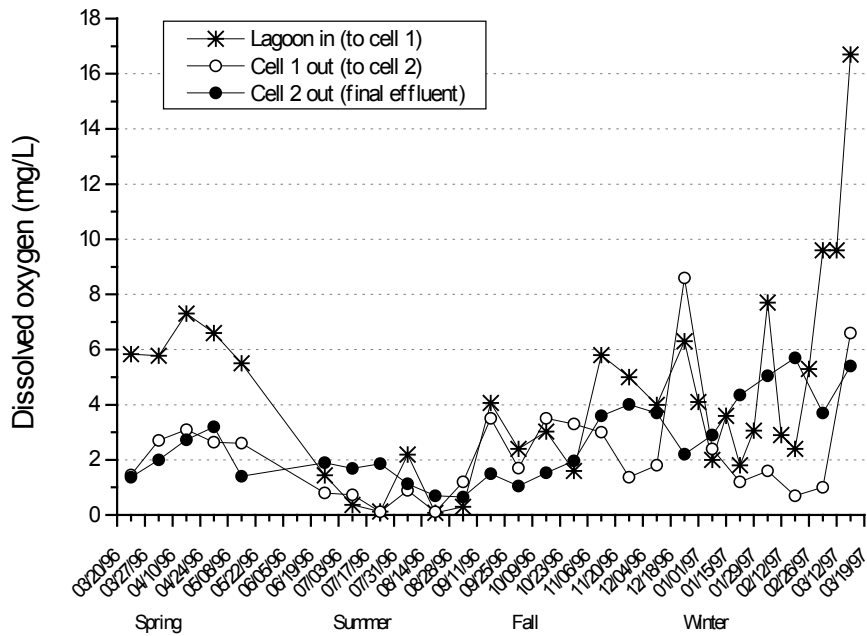
**Table 3.** Results of Student's t-tests comparing the mean lagoon water temperature and cell water temperature from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.



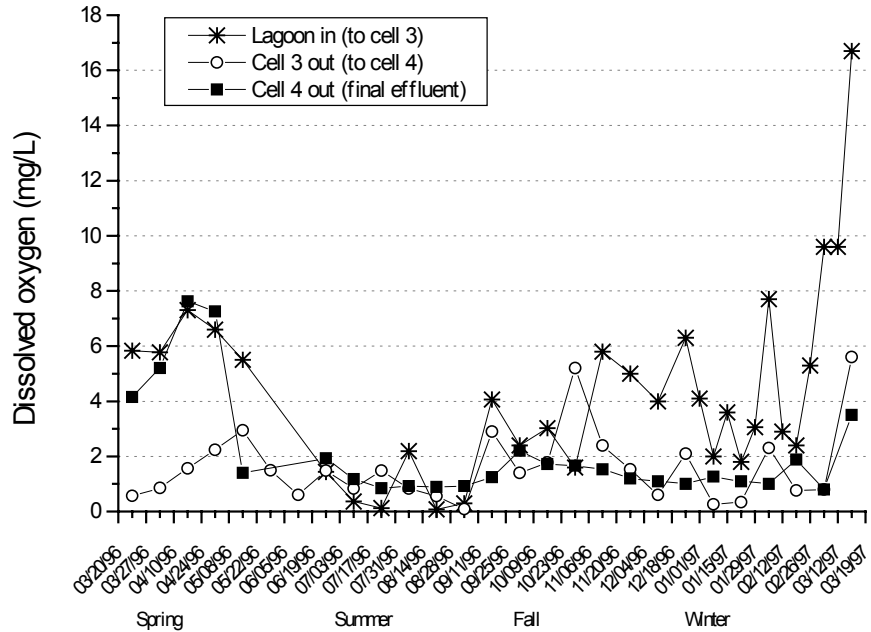
## Section 4.2 — Dissolved oxygen

Time series plots are presented in Figures 17 through 21 showing the lagoon influent and cell effluent dissolved oxygen concentrations for each cell in the experiment from March 20, 1996 to March 19, 1997.

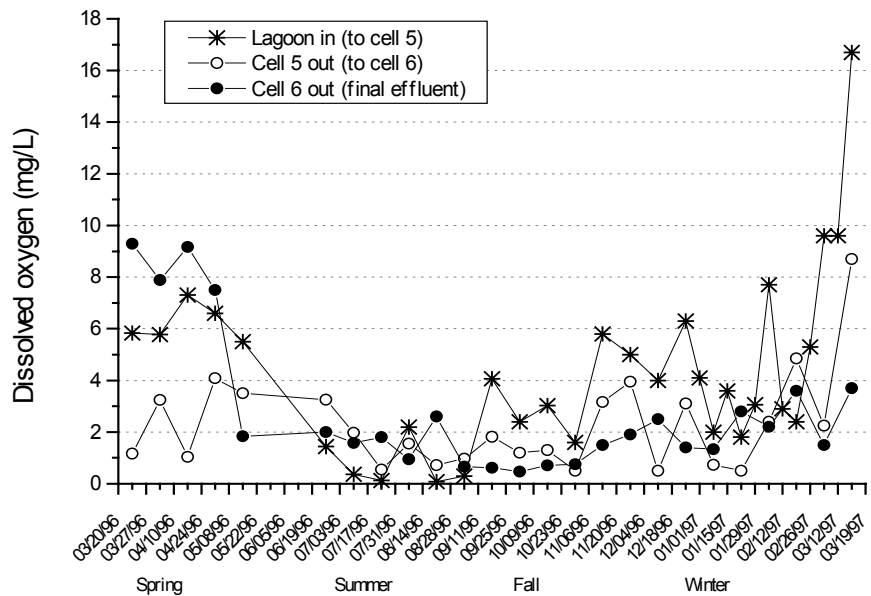
**Figure 17.** Time series plot of lagoon influent dissolved oxygen and Queenston Shale cells 1 & 2 (without plants) dissolved oxygen effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L.



**Figure 18.** Time series plot of lagoon influent dissolved oxygen and Queenston Shale cells 3 & 4 (with plants) dissolved oxygen effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L..

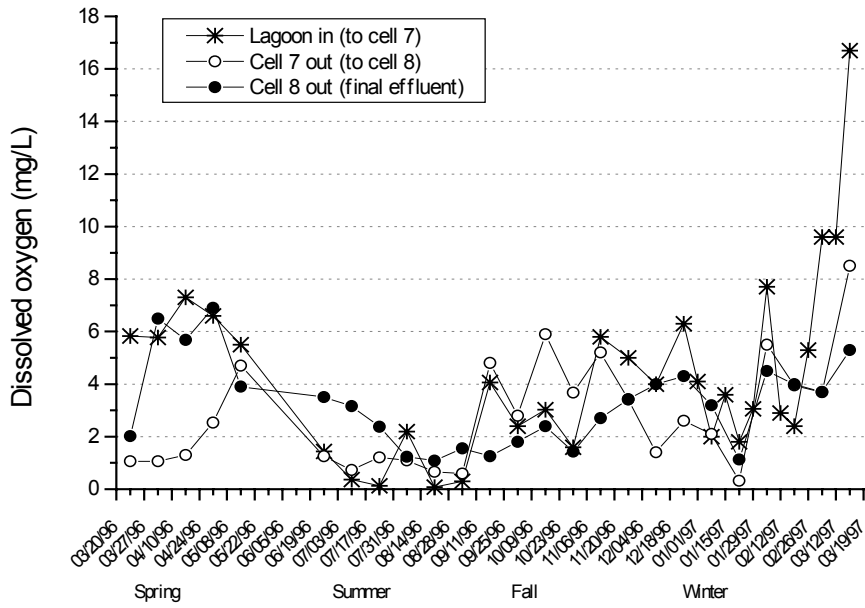


**Figure 19.** Time series plot of lagoon influent dissolved oxygen and Fonthill Sand cells 5 & 6 (with plants) dissolved oxygen effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L.

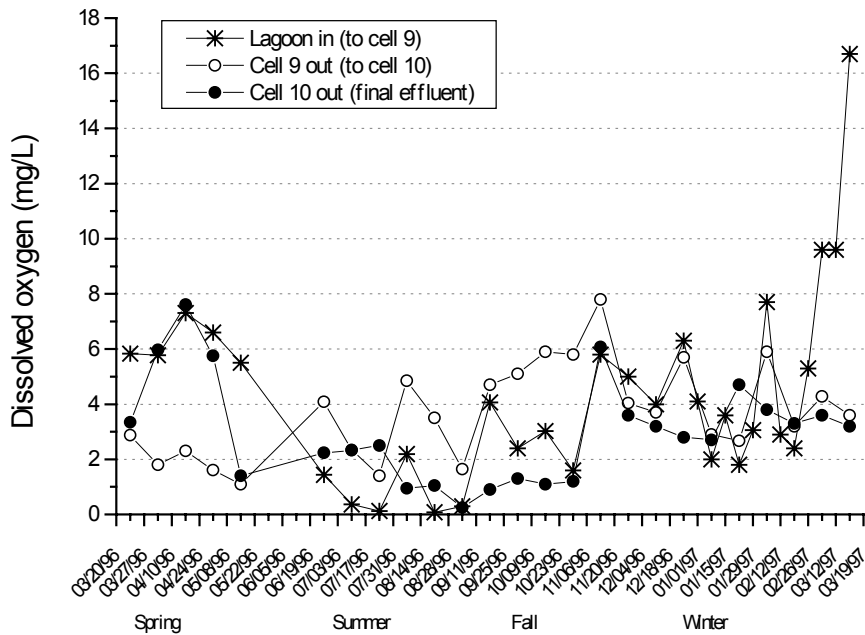


**Figure 20.** Time series plot of lagoon influent dissolved oxygen and Niagara Shale cells 7 & 8 (with plants) dissolved oxygen effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L.

reported in mg/L..



**Figure 21.** Time series plot of lagoon influent dissolved oxygen and Michigan Sand cells 9 & 10 (with plants) dissolved oxygen effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L.



Figures 17 through 21 show that the lagoon influent dissolved oxygen

concentration was lowest in Summer and higher during the Spring, Fall and Winter. The lowest lagoon influent dissolved oxygen concentration was 0.08 mg/L in Summer and the highest was 16.7 mg/L in March of 1997. Cell effluent dissolved oxygen concentration varied considerably, but generally followed the lagoon influent dissolved oxygen concentrations, at lower levels.

The twelve month mean dissolved oxygen concentration of the effluent of each cell is shown in Table 4. Table 5 shows the results of Student's T-tests comparing these data. It is evident from Tables 4 & 5 that with a few exceptions there was little difference among the individual cell effluent dissolved oxygen concentrations. Of note was that the first Michigan Sand cell effluent dissolved oxygen concentration (cell 9) was higher than the effluent dissolved oxygen concentration from cells one through five and the first Queenston Shale cell with plants (cell 3) effluent dissolved oxygen concentration was lower than that of cells six through ten.

**Table 4.** Mean dissolved oxygen concentration in the lagoon influent and effluent from each cell calculated from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	Lagoon	Q Sh	N P	Q Sh	W P	F Sa	W P	N Sh	W P	M Sa	W P
	influent	cell 1	cell 2	cell 3	cell 4	cell 5	cell 6	cell 7	cell 8	cell 9	cell 10
concentration (mg/L)	4.40 (0.62)	2.26 (0.39)	2.61 (0.30)	1.61 (0.26)	2.14 (0.39)	2.28 (0.37)	2.81 (0.53)	2.80 (0.42)	3.24 (0.34)	3.71 (0.34)	2.99 (0.38)



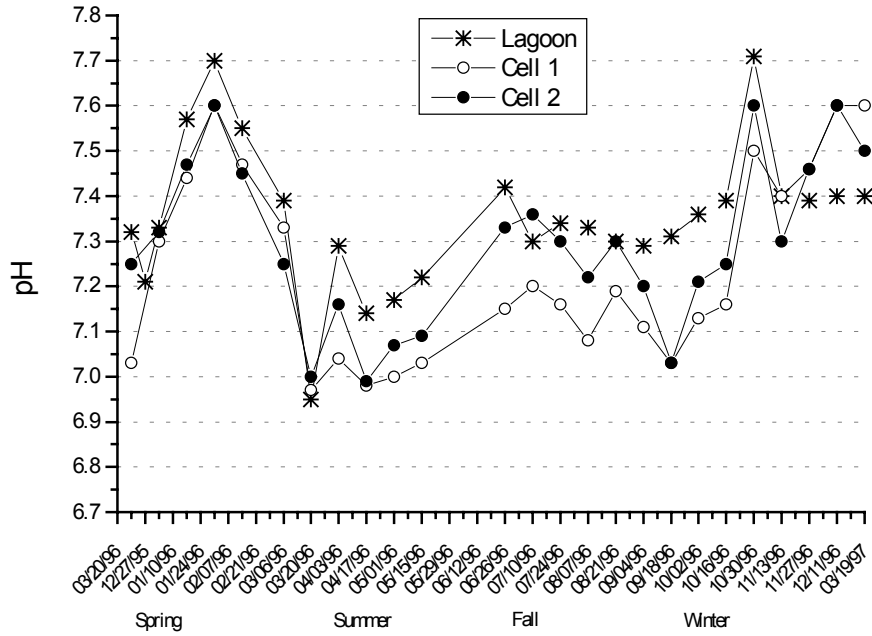
### section 4.3 — pH

The mean pH data (along with the Standard error) of the lagoon influent water and cell effluent water are presented in Table 6. Time series plots of lagoon influent pH and cell 1 through 10 effluent pH are presented below in figures 22 through 26 from March 20, 1996 to March 19, 1997.

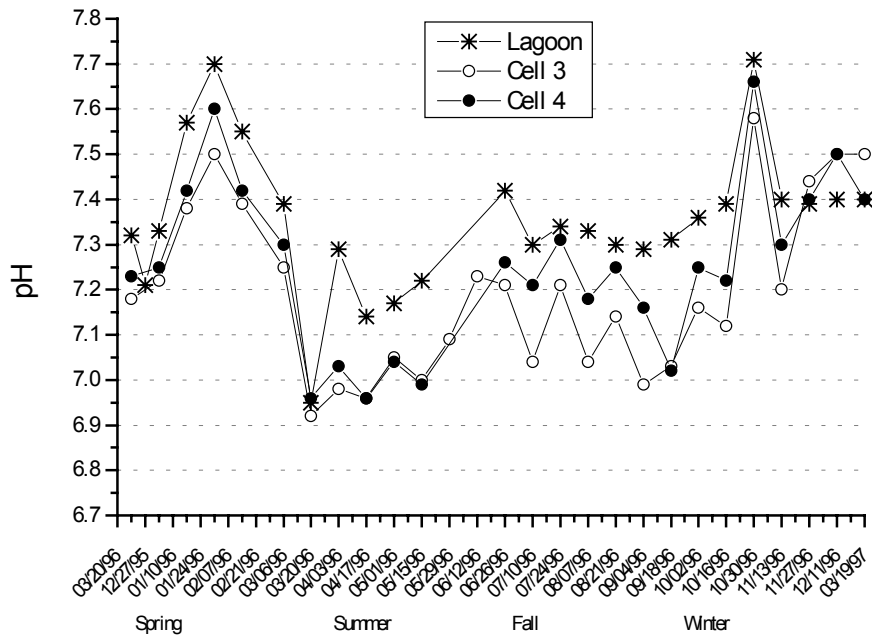
**Table 6.** Mean lagoon influent and individual cell effluent pH from March 20, 1996 to March 19, 1997. Numbers in parenthesis are the Standard Error.

Table 6 shows that all pH values recorded are near neutral. Some obvious seasonal fluctuations and patterns occur. For instance, all pH values increase in Spring and Winter and are lower during the Fall and Summer; in addition cell effluent pH values fluctuate with lagoon influent pH values (figures 22 - 26).

**Figure 22.** Time series plot of lagoon influent pH and Queenston Shale cells 1 & 2 (without plants) effluent pH from March 20, 1996 to March 19, 1997.

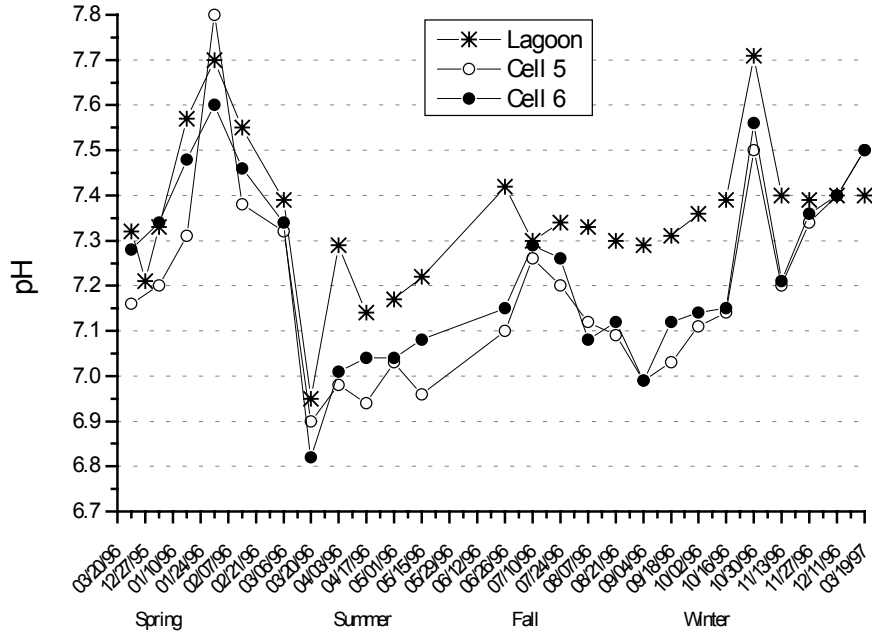


**Figure 23.** Time series plot of lagoon influent total pH and Queenston Shale cells 3 & 4 (with plants) effluent pH from March 20, 1996 to March 19, 1997.

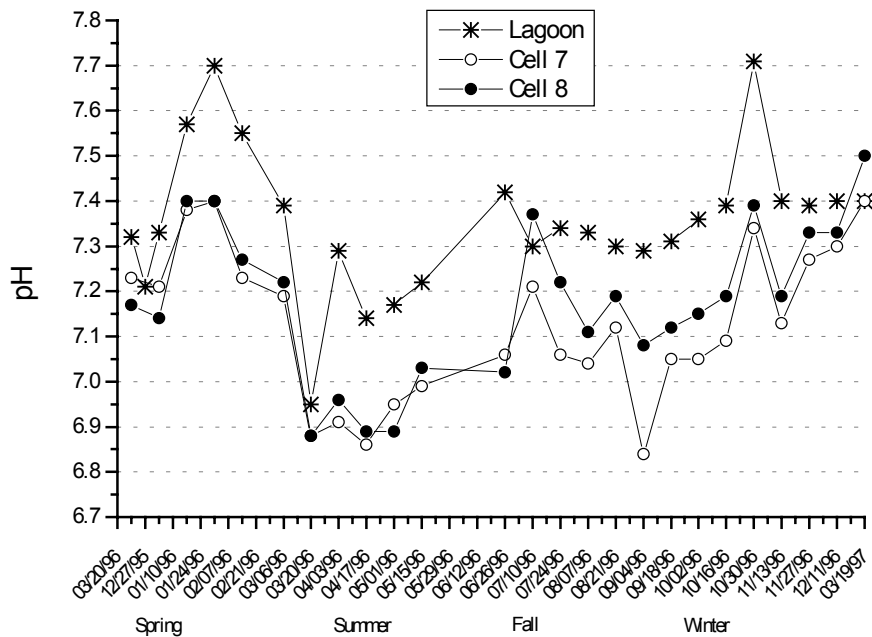


**Figure 24.** Time series plot of lagoon influent pH and Fonhill Sand cells 5 & 6 (with plants)

effluent pH from March 20, 1996 to March 19, 1999.



**Figure 25.** Time series plot of lagoon influent pH and Niagara Shale cells 7 & 8 (with plants) effluent pH from March 20, 1996 to March 19, 1997.



**Figure 26.** Time series plot of lagoon influent pH and Michigan Sand cells 9 & 10 (with plants) effluent pH from March 20, 1996 to March 19, 1997.

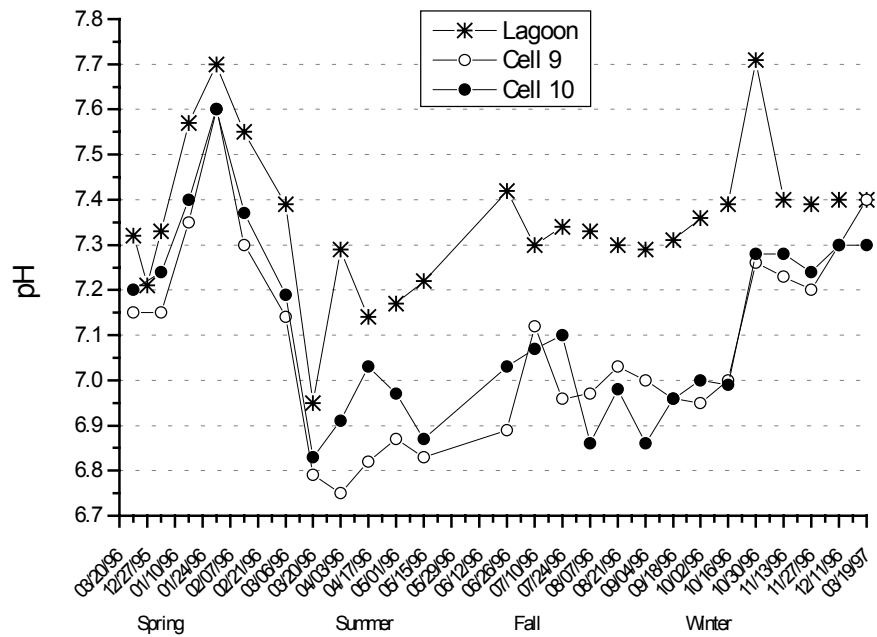


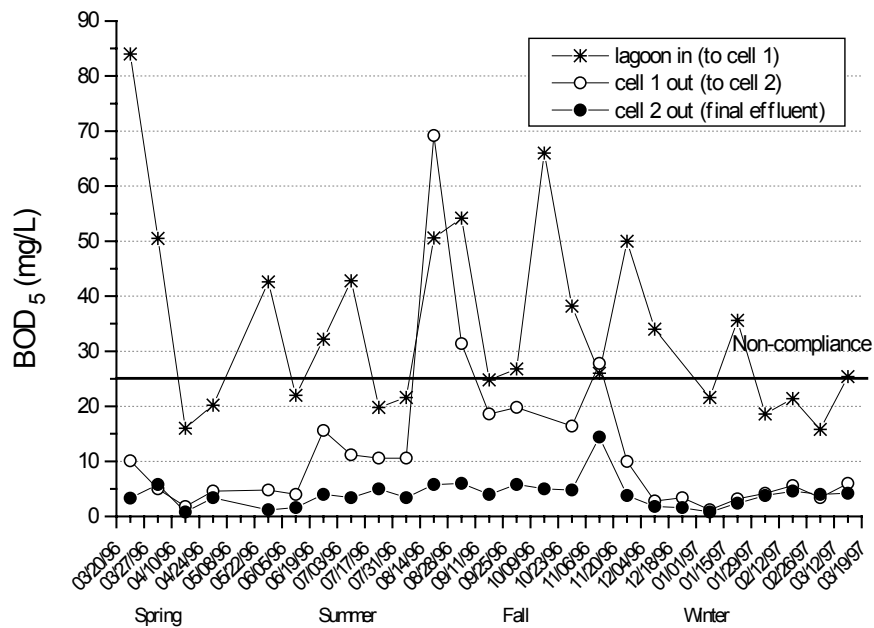
Table 7 presents the results of Student's t-tests performed on the data (at the 0.05% level of significance), these tests reveal some differences in cell pH. Cell effluent from the set containing plants was slightly lower than the lagoon influent pH (Tables 6 & 7). In the Queenston Shale set without plants, no statistically significant difference was found among the lagoon influent pH and cell 1 and 2 effluent pH. These same statistical tests also reveal that there is no difference in pH values among individual cells of the same set.



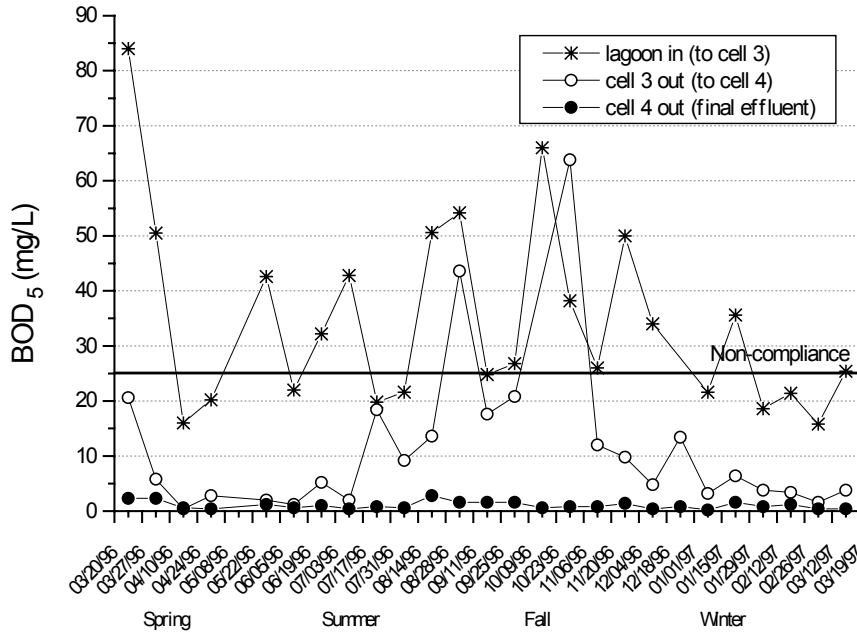
## Section 4.4 — Five Day Biochemical Oxygen Demand (BOD<sub>5</sub>)

Time series plots are presented in Figures 27 through 31 showing the influent and effluent BOD<sub>5</sub> for each cell in the experiment from March 20, 1996 to March 19, 1997. Effluent from the second cell in each set must meet MOE compliance levels before it can be discharged into the environment. The MOE non-compliance level for BOD<sub>5</sub> at Niagara-On-The-Lake, Ontario is 25 mg/L, and is shown on the plots as a solid line.

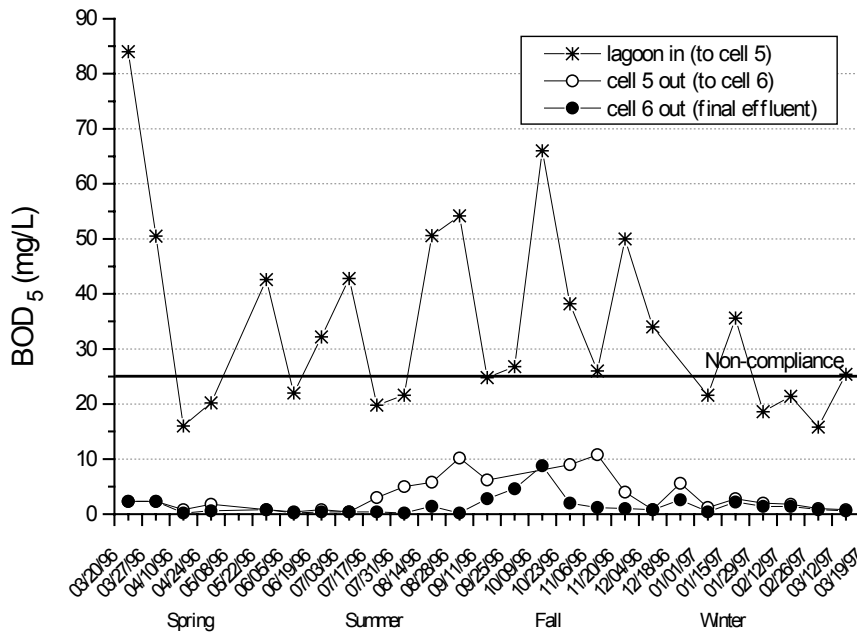
**Figure 27.** Time series plot of BOD<sub>5</sub> for lagoon influent and Queenston Shale cells 1 & 2 (without plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**Figure 28.** Time series plot of BOD<sub>5</sub> for lagoon influent and Queenston Shale cells 3 & 4 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.

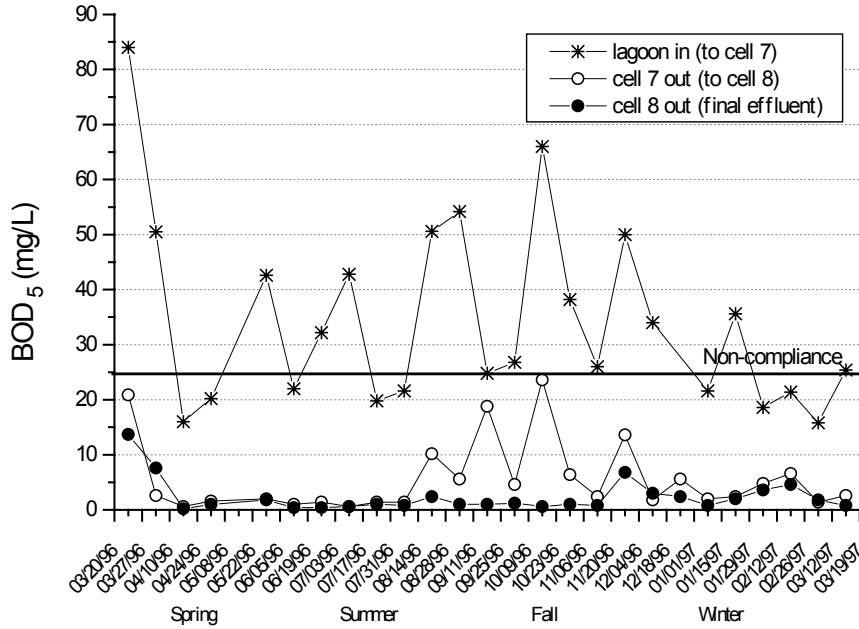


**Figure 29.** Time series plot of BOD<sub>5</sub> for lagoon influent and Fonthill Sand cells 5 & 6 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.

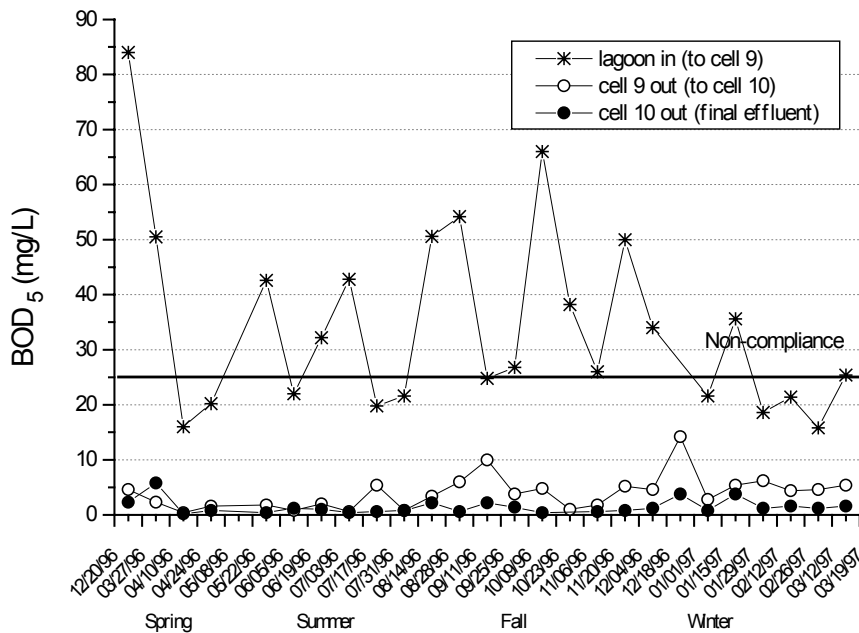


**Figure 30.** Time series plot of BOD<sub>5</sub> for lagoon influent and Niagara Shale cells 7 & 8 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.

plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**Figure 31.** Time series plot of BOD<sub>5</sub> for lagoon influent and Michigan Sand cells 9 & 10 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**BOD<sub>5</sub> concentration**

The above time series plots show that the lagoon influent BOD<sub>5</sub> was quite variable over time, ranging from 15.8 mg/L to 84 mg/L. These plots also show that the BOD<sub>5</sub> in the final effluent from all five series of cells was consistently below 25 mg/L, the MOE compliance level for discharge. Among the first cells in a set both Queenston Shale cells (Figures 27 & 28) show an increase in BOD<sub>5</sub> from about July 1996 to December 1996 and would not have met MOE discharge requirements. This same phenomenon is also seen in the first cell of the Niagara Shale set (Figure 30) and to a lesser extent in the first cells of the Fonthill and Michigan Sand sets (Figures 29 & 31) but the BOD<sub>5</sub> level of the effluent from these first cells at this time still met MOE compliance levels for discharge (Figures 29, 30 & 31).

The twelve month, mean BOD<sub>5</sub> for the effluent of each cell is shown in Table 8. Table 9 shows the results of Student's T-tests comparing these data. It is evident from Tables 8 & 9 that all of the cells lowered BOD<sub>5</sub> from the lagoon influent. It is also evident from Table 9 there was a significant difference in the effluent BOD<sub>5</sub> between cells of the same set, i.e., the effluent demands were higher in the effluent from the first cell in a set than in the effluent from the second cell in a set.

These same data can be used to assess the performance of a complete set as the data from the last cell of a set represents the performance of that set. The Queenston Shale set without plants had the highest mean effluent BOD<sub>5</sub> of 4.03 mg/L followed by the Michigan Sand set with an effluent BOD<sub>5</sub> of 2.36 mg/L. With the exception of the Queenston Shale set with plants which had a lower BOD<sub>5</sub> (1.05 mg/L) than the Niagara Shale set (2.36 mg/L), no difference was found among the rest of the sets.

**Table 8.** Mean BOD<sub>5</sub> in the lagoon influent and effluent from each cell calculated from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	Lagoon	Q Sh	N P	Q Sh	W P	F Sa	W P	N Sh	W P	M Sa	W P
	influent	cell 1	cell 2	cell 3	cell 4	cell 5	cell 6	cell 7	cell 8	cell 9	cell 10
concentrat -ion (mg/L)	34.43 (3.43)	12.05 (2.87)	4.03 (0.52)	11.57 (2.89)	1.05 (0.13)	3.32 (0.64)	1.54 (0.36)	5.61 (1.28)	2.36 (0.58)	4.00 (0.60)	1.48 (0.26)

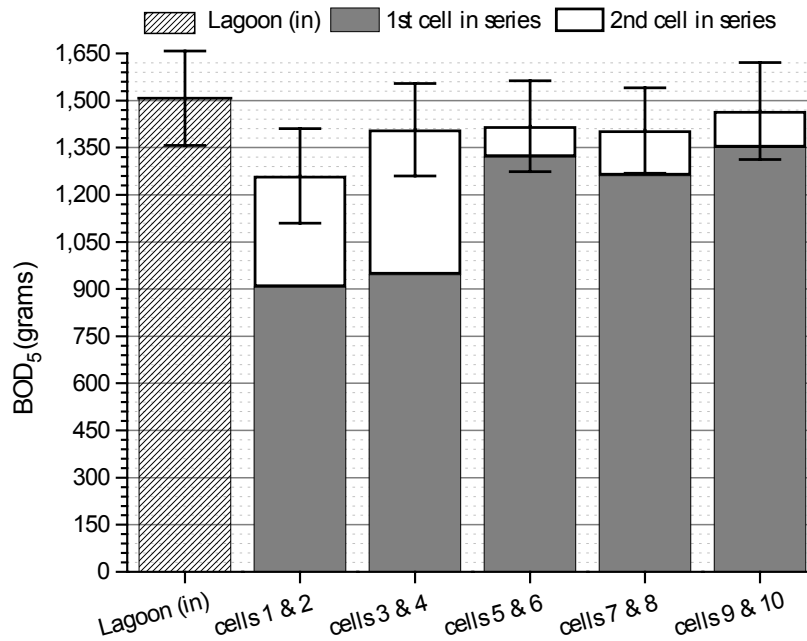
**Table 9** Results of Student's t-tests comparing the mean BOD<sub>5</sub> concentration in the effluent each cell from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	cell 1	cell 2	cell 3	cell 4	Cell 5	Cell 6	Cell 7	cell 8	cell 9	Cell 10
	Q Sh NP	Q Sh NP	Q Sh WP	Q Sh WP	F Sa WP	F Sa WP	N Sh WP	N Sh WP	M Sa WP	M Sa WP
Lagoon influent	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
cell 1 Q Sh NP		0.001	0.905	0.000						
cell 2 Q Sh NP			0.117	0.000						
cell 3 Q Sh WP				0.000	0.000	0.000	0.061	0.002	0.011	0.000
cell 4 Q Sh WP					0.144	0.089	0.013	0.093	0.000	0.917
Cell 5 F Sa WP						0.016	0.123	0.273	0.443	0.009
Cell 6 F Sa WP							0.003	0.237	0.000	0.889
Cell 7 N Sh WP								0.257	0.003	0.003
cell 8 N Sh WP									0.136	0.180
cell 9 M Sa WP										0.000

The BOD<sub>5</sub> data were also analyzed to show lowering of BOD<sub>5</sub> by each set of cells. These results are shown in Figure 32. Figure 32 shows that the first cell in a set removed more BOD<sub>5</sub> than the second cell of that set.

The total percent reduction by each set was then calculated and these data are presented in Table 10. Table 11 shows the results of Student's t-tests comparing the data presented in Table 10.

**Figure 32.** Reduction of BOD<sub>5</sub> in grams by each set of cells from 03/20/96 to 03/19/97. Standard Error bars are shown for the BOD<sub>5</sub> reduction by each set of cells.



The Quee

less efficient than all other sets (Table 10). The Queenston Shale set without plants significantly lowered BOD<sub>5</sub> over the Niagara Shale set (Table 11). The Queenston Shale set with plants significantly lowered BOD<sub>5</sub> over the Queenston Shale set without plants (Table 11).

**Table 10.** Mean % reduction in BOD<sub>5</sub> by each set of cells from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	set 1 Q Sh NP cells 1 & 2	set 2 Q Sh WP cells 3 & 4	set 3 F Sa WP cells 5 & 6	set 4 N Sh WP cells 7 & 8	set 5 M Sa WP cells 9 & 10
% reduction	85.93 (2.19)	96.81 (0.32)	95.68 (0.68)	93.46 (1.21)	95.57 (0.63)

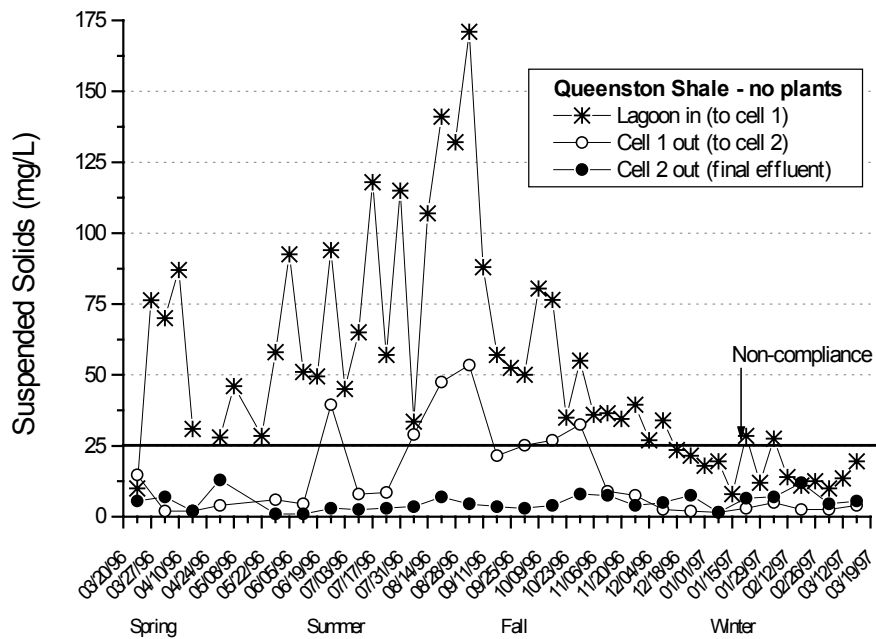
**Table 11.** Results of Student's t-tests comparing the mean % reduction of BOD<sub>5</sub> by each set of cells from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance

	Q Sh WP cells 3 & 4	F Sa WP cells 5 & 6	N Sh WP cells 7 & 8	M Sa WP cells 9 & 10
Q Sh NP cells 1 & 2	0.000			
Q Sh WP cells 3 & 4		0.213	0.093	0.081
F Sa WP cells 5 & 6			0.135	0.917
N Sh WP cells 7 & 8				0.130

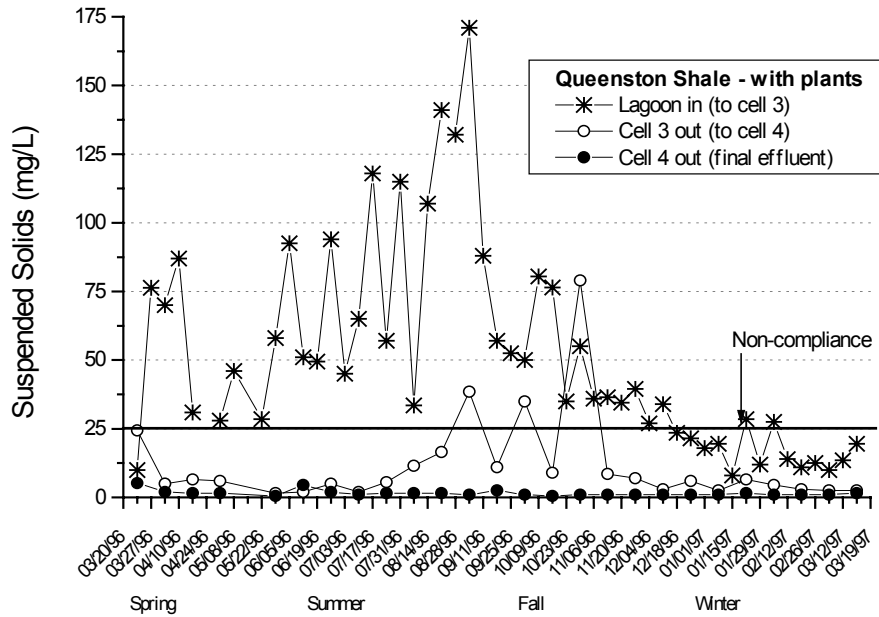
## Section 4.5 — suspended Solids

Time series plots are presented in Figures 33 through 37 showing the influent and effluent suspended solids concentration for each cell in the experiment from March 20, 1996 to March 19, 1997. Effluent from the second cell in each set must meet MOE compliance levels before it can be discharged into the environment. The MOE non-compliance level for suspended solids at Niagara-On-The-Lake, Ontario is 25 mg/L, and is shown on the plots as a solid line.

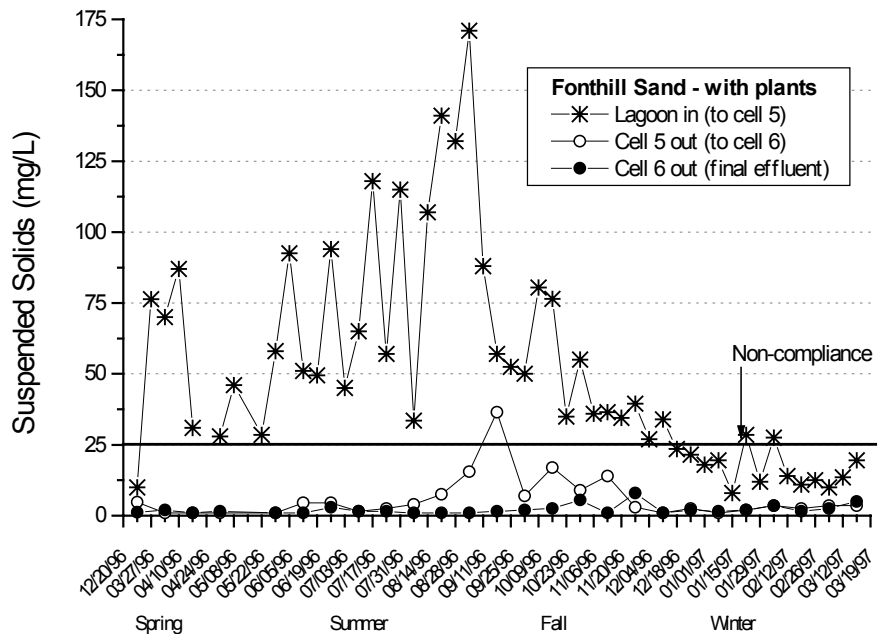
**Figure 33.** Time series plot of suspended solids for lagoon influent and Queenston Shale cells 1 & 2 (without plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**Figure 34.** Time series plot of suspended solids for lagoon influent and Queenston Shale cells 3 & 4 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.

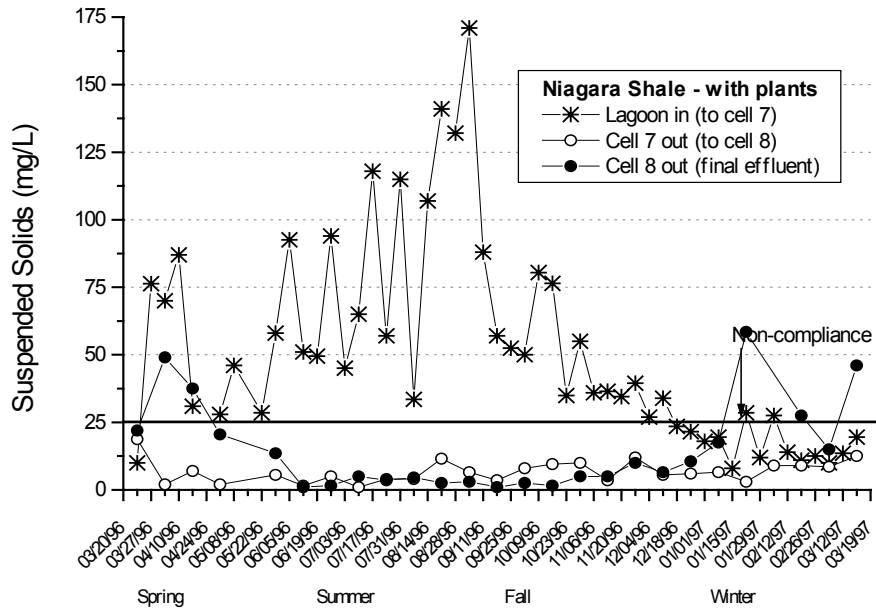


**Figure 35.** Time series plot of suspended solids for lagoon influent and Fonthill Sand cells 5 & 6 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.

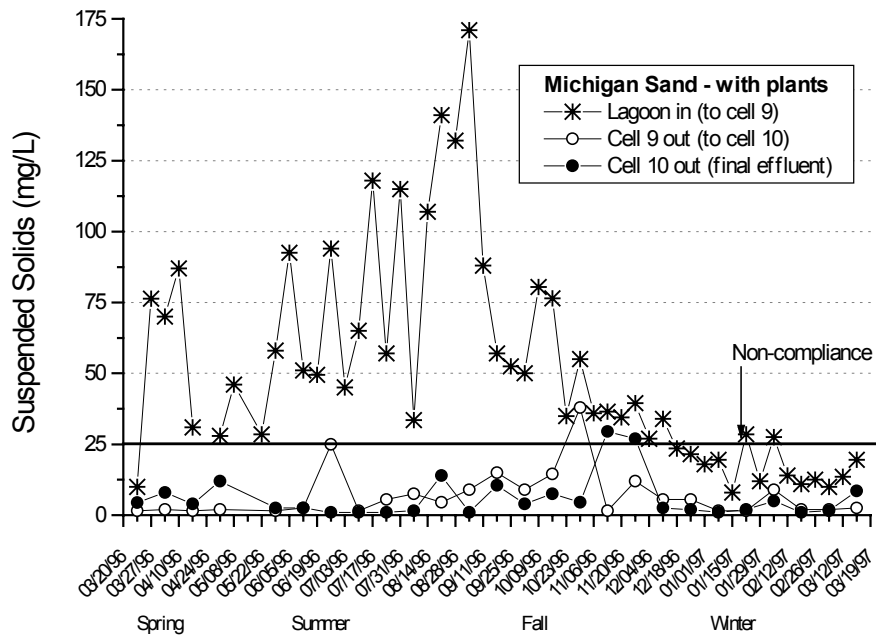


**Figure 36.** Time series plot of suspended solids for lagoon influent and Niagara Shale cells 7 & 8

(with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**Figure 37.** Time series plot of suspended solids for lagoon influent and Michigan Sand cells 9 & 10 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance level shown by solid line at 25 mg/L.



**Suspended Solids Concentration**

The time series plots (Figures 33 -37) show that the influent suspended solids concentration was quite variable over time, ranging from 171 mg/L to 8 mg/L.

Final effluent from both the Queenston Shale set (Figures 33 & 34) and the Fonthill sand set (Figure 35) was consistently below 25 mg/L and thus met MOE requirements for discharge during the time of the experiment. The Niagara Shale set (Figure 36) did not meet the MOE compliance objective during two distinct periods when effluent suspended solids concentration occasionally exceeded even the lagoon influent suspended solids concentration. The Michigan Sand set (figure 37) was out of compliance during two consecutive sampling events in November 1996.

The quality of individual cell effluent varied over time. For instance, effluent suspended solids concentration from the first cells in both of the Queenston Shale set increased from about July 1996 to November 1996 and would not have met MOE discharge requirements for most of this time. The first cell of the Fonthill sand set displayed a similar trend but only exceeded MOE discharge requirements once ( September 1996). The first cell of the Niagara Shale set met the MOE discharge requirements during the entire experimental period, whereas the second cell produced suspended solids.

The suspended solids concentration in the effluent of each cell is shown in Table 12. Table 13 shows the results of Student's T-tests comparing these data. It is evident from Table 12 that all of the cells removed suspended solids from the lagoon influent. It is also evident from Table 13 that with the exception of the Michigan Sand cells there was a significant difference in the effluent suspended solids concentration between cells of the same set, i.e., the effluent concentrations were higher in the effluent from the first cell in a set than in the effluent from the second cell of that set.



### Grams Of Suspended Solids Removed

The suspended solids data were also analyzed to show the total grams of suspended solids removed by each set of cells. These results are shown in Figure 38. The total set percent reduction was then calculated and the results presented in Table 14. Table 15 shows the results of Student's t-tests comparing the data presented in Table 14.

**Figure 38.** Grams of suspended solids in the lagoon influent and removed by each set of cells from 03/20/96 to 03/19/97. Standard Error bars are shown for the grams of suspended solids removed by each set of cells. The hatched section on the cells 7 & 8 column

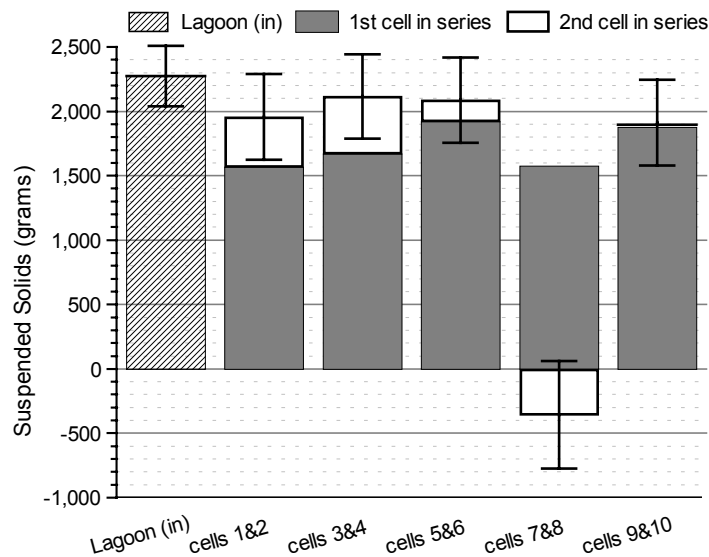


Figure 38

Cell 8 of the Niagara Shale set produced more suspended solids than it removed.

Tables 14

between them. The Queenston Shale set without plants removed approximately 81% of the total suspended solids applied to it. This was similar to the Michigan Sand set which removed approximately 83%. The Niagara Shale set performed poorly, removing only 36% of the suspended solids applied to it. The Queenston

Shale set with plants removed significantly more suspended solids than the Queenston Shale set without plants.

**Table 14.** Mean % reduction of suspended solids by each set of cells from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	set 1 Q Sh NP cells 1 & 2	set 2 Q Sh WP cells 3 & 4	set 3 F Sa WP cells 5 & 6	set 4 N Sh WP cells 7 & 8	set 5 M Sa WP cells 9 & 10
% reduction	80.64 (4.66)	94.24 (1.92)	92.83 (1.40)	36.10 (16.66)	82.77 (4.12)

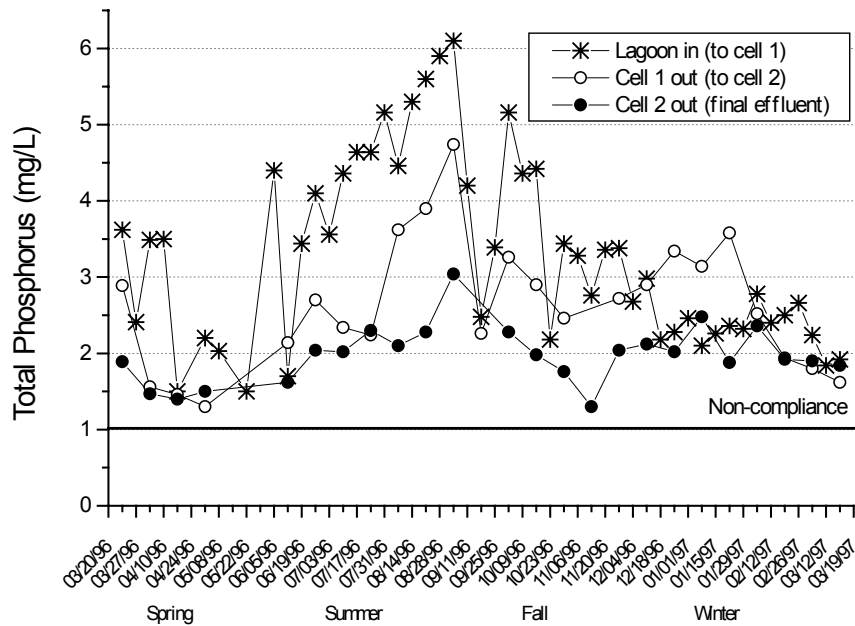
**Table 15.** Results of Student's t-tests comparing the mean % reduction of suspended solids by each set of cells from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	Q Sh WP cells 3 & 4	F Sa WP cells 5 & 6	N Sh WP cells 7 & 8	M Sa WP cells 9 & 10
Q Sh NP cells 1 & 2	0.009			
Q Sh WP cells 3 & 4		0.341	0.007	0.014
F Sa WP cells 5 & 6			0.001	0.025
N Sh WP cells 7 & 8				0.007

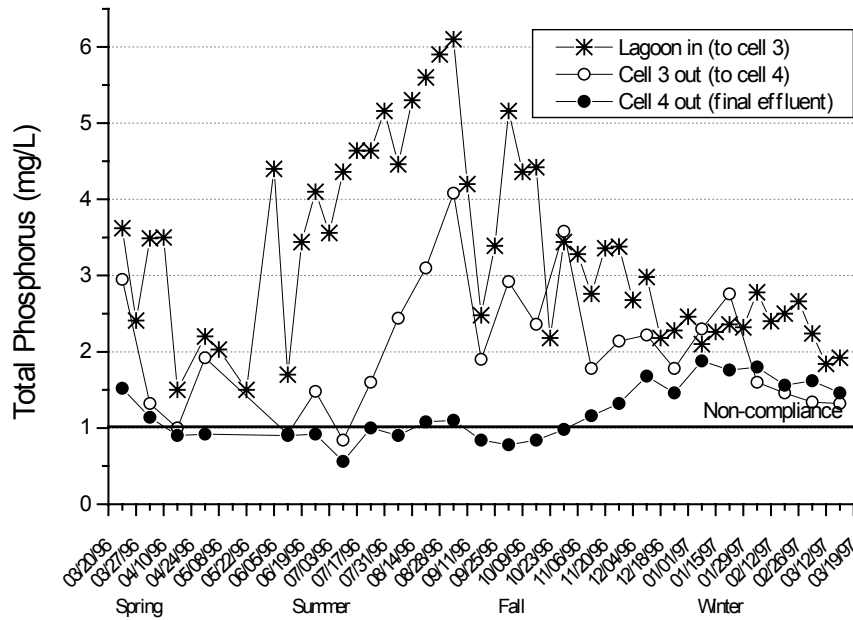
## Section 4.6 — Total Phosphorus

Time series plots are presented in Figures 39 through 43 showing the influent and effluent total phosphorus concentration for each cell in the experiment from March 20, 1996 to March 19, 1997. Effluent from the second cell in each set must meet MOE compliance levels before it can be discharged into the environment. The MOE non-compliance level for total phosphorus at Niagara-On-The-Lake, Ontario is 1 mg/L, and is shown on the plots as a solid line.

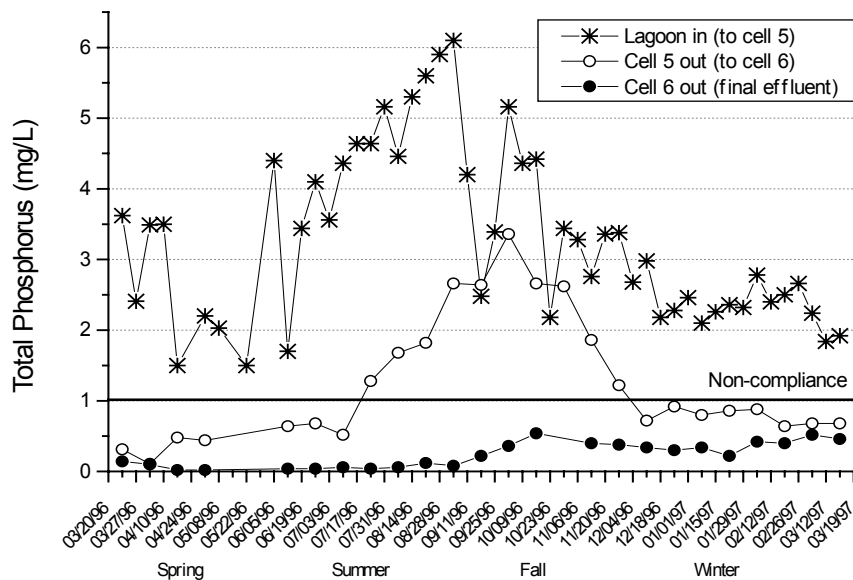
**Figure 39.** Time series plot of lagoon influent total phosphorus and Queenston Shale cells 1 & 2 (without plants) total phosphorus effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L. MOE non-compliance level shown by solid line at 1 mg/L.



**Figure 40.** Time series plot of lagoon influent total phosphorus and Queenston Shale cells 3 & 4 (with plants) total phosphorus effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L. MOE non-compliance level shown by solid line at 1 mg/L.

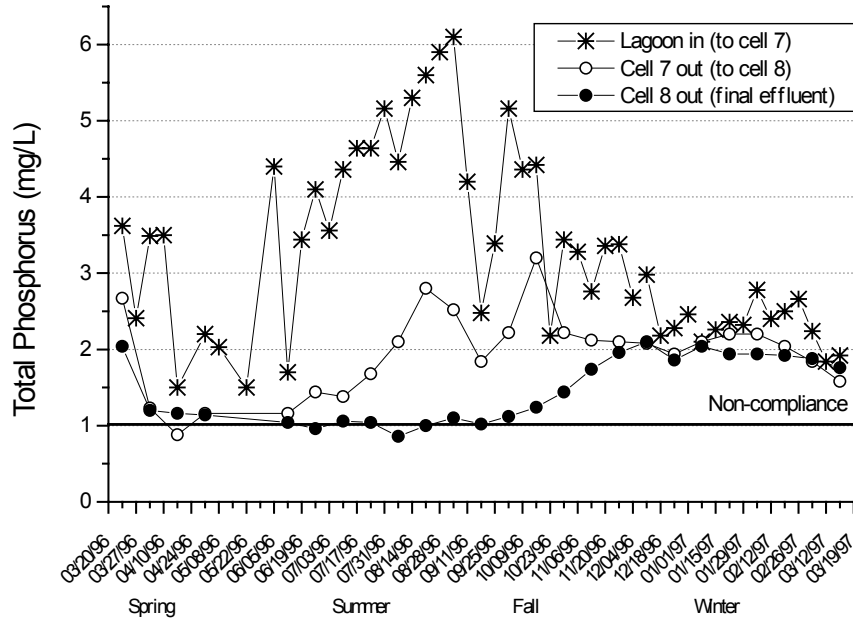


**Figure 41.** Time series plot of lagoon influent total phosphorus and Fonthill Sand cells 5 & 6 (with plants) total phosphorus effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L. MOE non-compliance level shown by dashed line at 1 mg/L.

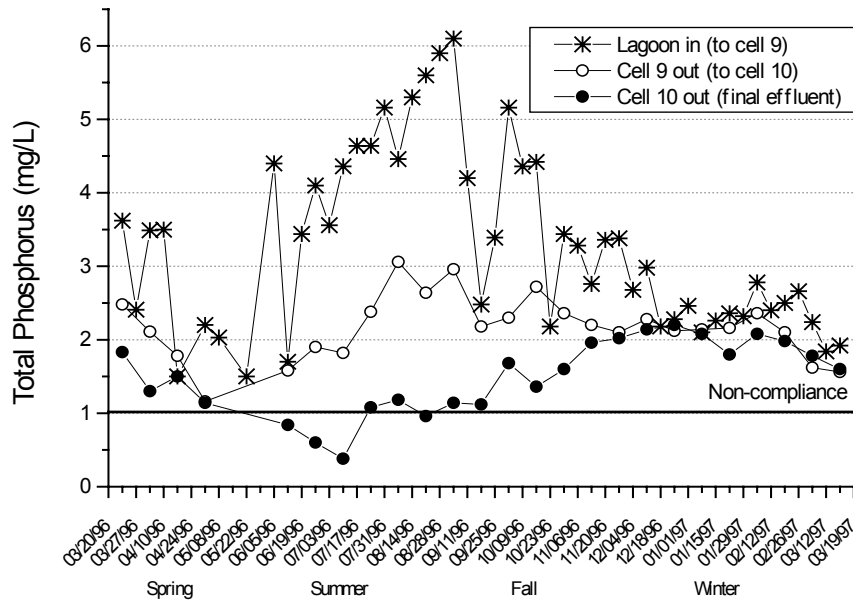


**Figure 42.** Time series plot of lagoon influent total phosphorus and Niagara Shale cells 7 & 8

(with plants) total phosphorus effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L. MOE non-compliance level shown by dashed line at 1 mg/L.



**Figure 43.** Time series plot of lagoon influent total phosphorus and Michigan Sand cells 9 & 10 (with plants) total phosphorus effluent from March 20, 1996 to March 19, 1997. All values reported in mg/L. MOE non-compliance level shown by dashed line at 1 mg/L.



Total Phosphorus Concentration

Figures 39 through 43 show that the lagoon input total phosphorus level varied from 1.5 to 6.1 mg/L. Levels were generally higher in summer and lower in winter. Cell effluent total phosphorus levels also varied considerably. Generally the effluent from the first cells in each set was highest from about July until November. During this time effluent from the first cells of all sets was above MOE discharge requirements.

Final effluent from the Queenston Shale set without plants (Figure 39) was out of compliance for the entire twelve months. The Queenston Shale set with plants (Figure 40) behaved somewhat differently. During spring, summer and fall (with the exception of 2 times each in the early spring and summer) final effluent from this set was within MOE requirements for discharge. However with the onset of winter this set no longer met the MOE discharge requirements.

Final effluent from the Fonthill Sand set (Figure 41) was well within MOE discharge requirements for the entire course of the experiment. However, effluent from the first cell in this set did not meet these requirements from about July to the end of November.

The Niagara Shale set (Figure 42) was continually out of compliance except for two occasions, once in early July and the second in mid August. Total phosphorus in the final effluent from this set was highest during the winter months. With the exception of one sampling period in April effluent from the first cell in set was out of compliance the entire experiment.

The Michigan Sand set (Figure 43) met MOE discharge requirements only through late June and early July. Final effluent from this set was highest in the Winter months. Effluent from the first cell in this set was out of compliance over the entire course of the experiment.

Table 16 shows the twelve month mean concentration of total phosphorus in the effluent of each cell and in the lagoon influent. The results of Student's t-

tests comparing these data are shown in Table 17. It is evident from Table 17 that all cells lowered total phosphorus. It is also evident that in all five sets the effluent total phosphorus concentration from the first cell in a set was higher and significantly different than the effluent total phosphorus concentration of the second cell of the same set.

**Table 16.** Mean total phosphorus concentration in the lagoon influent and effluent from each cell calculated for 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	Lagoon influent	Q Sh cell 1	N P cell 2	Q Sh cell 3	W P cell 4	F Sa cell 5	W P cell 6	N Sh cell 7	W P cell 8	M Sa cell 9	W P cell 10
concentration (mg/L)	3.28 (0.17)	2.64 (0.17)	1.98 (0.08)	2.04 (0.17)	1.20 (0.07)	1.25 (0.18)	0.23 (0.04)	1.95 (0.11)	1.46 (0.09)	2.16 (0.09)	1.49 (0.10)

**Table 17.** Results of Student's t-tests comparing the mean total phosphorus in the lagoon influent and cell effluent from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	cell 1 Q Sh NP	cell 2 Q Sh NP	cell 3 Q Sh WP	cell 4 Q Sh WP	Cell 5 F Sa WP	Cell 6 F Sa WP	Cell 7 N Sh WP	cell 8 N Sh WP	cell 9 M Sa WP	Cell 10 M Sa WP
<b>lagoon influent</b>	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
cell 1 Q Sh NP		0.000	0.011	0.000						
cell 2 Q Sh NP			0.817	0.000						
cell 3 Q Sh WP				0.000	0.000	0.000	0.836	0.232	0.663	0.001
cell 4 Q Sh WP					0.131	0.000	0.000	0.001	0.000	0.000
Cell 5 F Sa WP						0.000	0.000	0.044	0.000	0.282
Cell 6 F Sa WP							0.000	1.222	0.000	0.000
Cell 7 N Sh WP								0.007	0.819	0.000
cell 8 N Sh WP									0.001	0.208
cell 9 M Sa WP										0.000

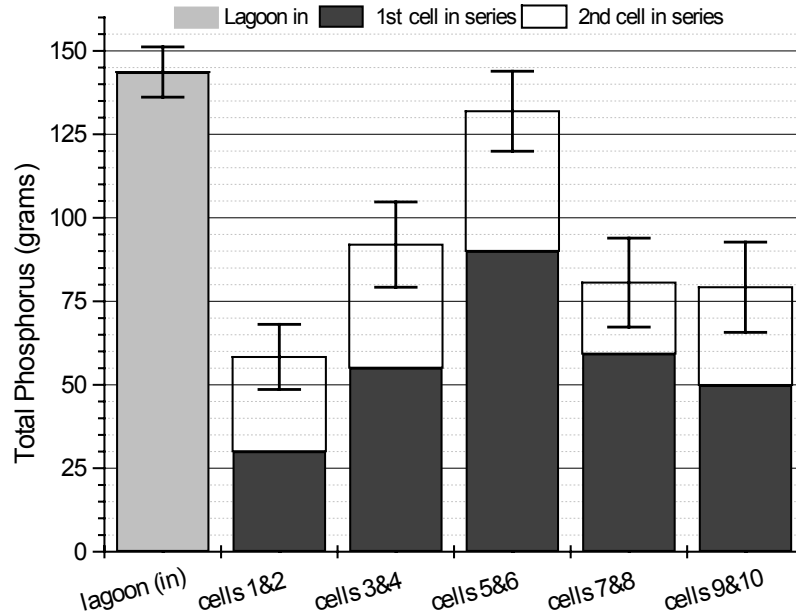
Tables 16 & 17 clearly show that the effluent from the first cell of the Fonthill Sand set had the lowest mean total phosphorus concentration and the first cell of the Queenston Shale set without plants had the highest. The total phosphorus concentration in the effluent from the first cells of the Queenston

Shale set with plants and the Niagara Shale and Michigan Sand set were statistically similar to each other.

These same data can be used to assess the performance of an entire set as the data from the last cell of that set represents the performance of the entire set. The Fonthill Sand set had the lowest mean effluent total phosphorus concentration of 0.23 mg/L followed by the Queenston Shale set at 1.20 mg/L. The Niagara Shale set and the Michigan Sand set had mean effluent concentrations of 1.46 and 1.49 mg/L respectively with no statistical difference between them. The Queenston Shale set without plants had the highest mean effluent total phosphorus concentration of 2.64 mg/L.

The data were also used to compare grams of total phosphorus removed by each set of cells (Figure 44). The percent reduction by each set was calculated and is shown in Table 18. Table 19 contains the results of Student's t-tests comparing these data. The data found in Tables 18 & 19 reveal that the Fonthill Sand set removed the most total phosphorus, 91.60%, followed by the Queenston Shale set with plants which removed 56.74%. The remaining sets removed between 34.09% and 47.45% total phosphorus and no significant difference exists among these sets. These data also show that the Queenston Shale set without plants removed significantly less total phosphorus than its counterpart containing plants.

**Figure 44.** Grams of total phosphorus removed by each set of cells from 03/20/96 to 03/19/97. Standard Error bars are shown for the grams of total phosphorus removed by each set of cells.



**Table 18.** Mean % reduction of total phosphorus by each set of cells from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	set 1 Q Sh NP cells 1 & 2	set 2 Q Sh WP cells 3 & 4	set 3 F Sa WP cells 5 & 6	set 4 N Sh WP cells 7 & 8	set 5 M Sa WP cells 9 & 10
% reduction	34.09 (4.51)	56.74 (4.52)	91.60 (1.48)	47.45 (5.24)	46.23 (5.63)

**Table 19.** Results of Student's t-tests comparing the mean % reduction of total phosphorus by each set of cells from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	Q Sh WP cells 3 & 4	F Sa WP cells 5 & 6	N Sh WP cells 7 & 8	M Sa WP cells 9 & 10
Q Sh NP cells 1 & 2	0.000			
Q Sh WP cells 3 & 4		0.000	0.039	0.263
F Sa WP cells 5 & 6			0.000	0.000
N Sh WP cells 7 & 8				0.093

**Section 4.7 — Total Nitrogen, Ammonia and Ammonium and Nitrate.**

Total nitrogen consists of total kjeldhal nitrogen, nitrate, nitrite, ammonia and ammonium. The MOE has no effluent discharge requirement for total nitrogen. However because ammonia ( $\text{NH}_3$ ) is toxic to aquatic life, the MOE has established discharge criteria for both ammonia and ammonium ( $\text{NH}_4^+$ ). These levels are dependent on season. The MOE Certificate of Approval for the Niagara-On-The-Lake sewage treatment plant defines summer season from May 1 to October 31 and winter season as November 1 to April 31. The ammonia and ammonium non-compliance level for summer is 10 mg/L and for winter 20 mg/L. Although not regulated by the MOE at Niagara-On-The-Lake some jurisdictions regulate nitrate concentrations in discharge from sewage treatment plants.

Time series plots of the lagoon influent and cell effluent total nitrogen, ammonia and ammonium and nitrate from March 20, 1996 to March 19, 1997 are presented in Figures 45 through 59. These figures show that total nitrogen and ammonia and ammonium concentrations in the lagoon influent fluctuate seasonally, with lagoon influent values being highest during the summer and early fall. No seasonal fluctuation of lagoon influent nitrate is evident.

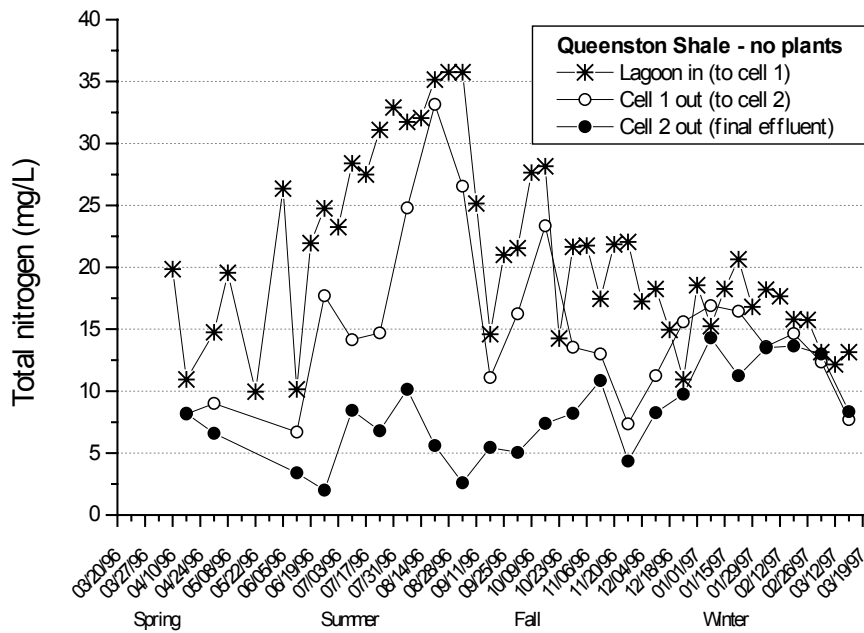
Figure 45 shows that in most instances effluent total nitrogen from the Queenston Shale set without plants was lower than influent total nitrogen. In all instances the data from the last cell of a set will be used to assess individual set performance because the data from this cell represent the performance of that set. Total nitrogen concentrations in the effluent from the first cell was higher than in the effluent from the second cell of a set (Figure 45).

Effluent ammonia and ammonium from the Queenston Shale set without plants was lower than lagoon influent ammonia and ammonium at all times and was within MOE compliance levels throughout the year (Figure 46). Effluent ammonia and ammonium from the first cell was out of compliance three times

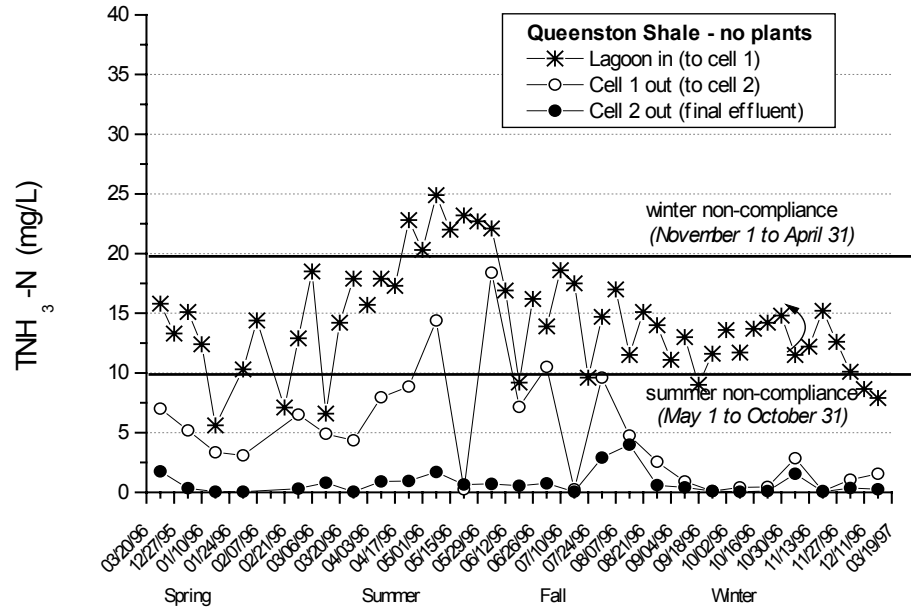
over the spring and summer.

The effluent nitrate concentrations from both cells in this set were higher than lagoon influent nitrate concentrations throughout the experiment (Figure 47).

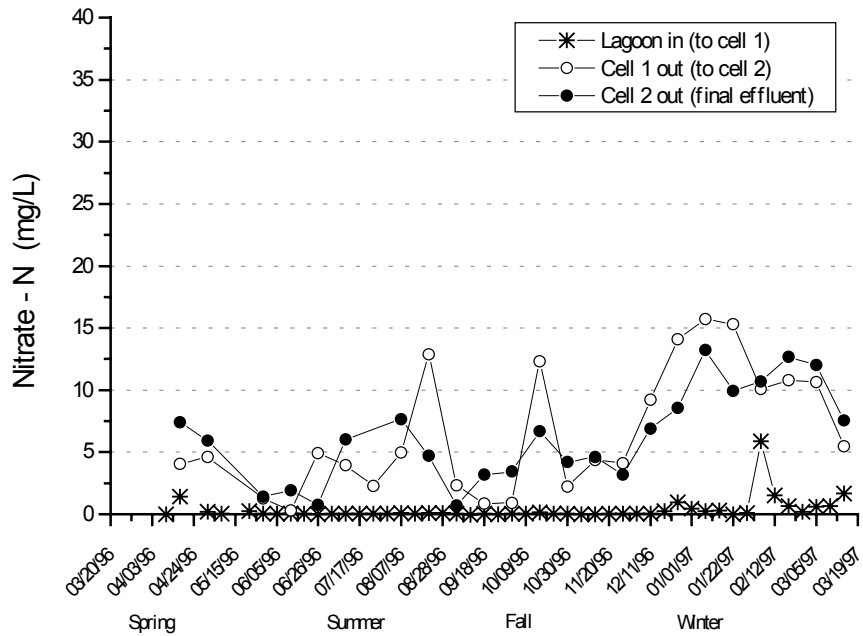
**Figure 45.** Time series plot of total nitrogen for lagoon influent and Queenston Shale cells 1 & 2 (without plants) effluent from March 20, 1996 to March 19, 1997. There is no MOE non-compliance level.



**Figure 46.** Time series plot of ammonia and ammonium for lagoon influent and Queenston Shale cells 1 & 2 (without plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.



**Figure 47.** Time series plot of nitrate for lagoon influent and Queenston Shale cells 1 & 2 (without plants) effluent from March 20, 1996 to March 19, 1997.

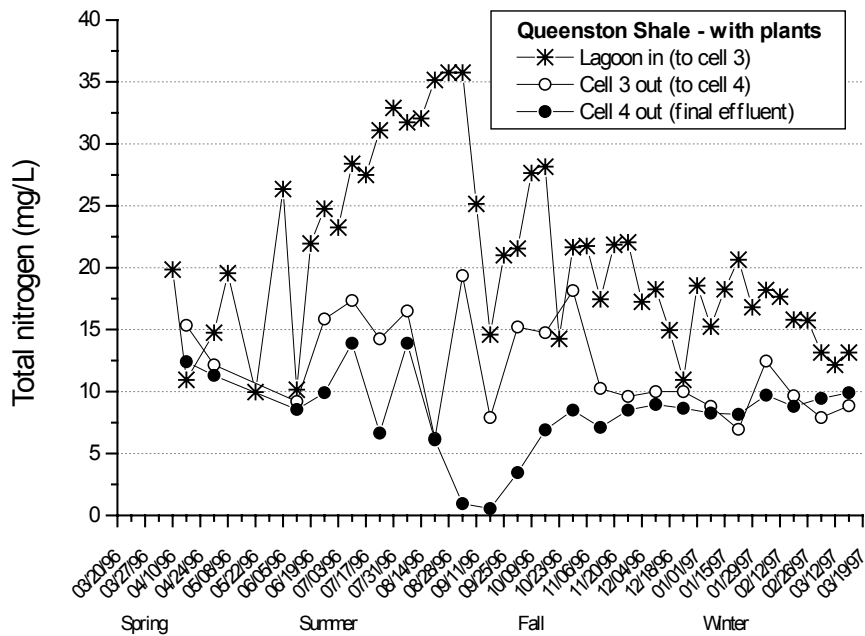


With one exception (on 04/17/96), effluent total nitrogen from the Queenston Shale set with plants (Figure 48) was lower than influent total nitrogen. Total nitrogen concentrations in the effluent from the first cell was higher than in the effluent from the second cell except for two times in the Winter.

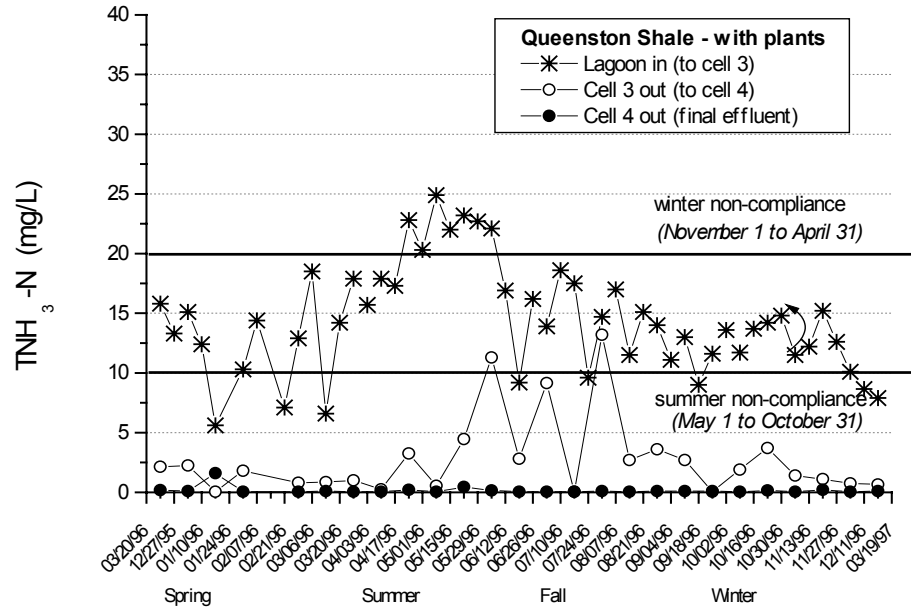
Effluent ammonia and ammonium from this set was lower than lagoon influent ammonia and ammonium at all times and was within MOE compliance throughout the year (Figure 49). Effluent ammonia and ammonium from the first cell was out of compliance twice during the Summer.

Cell effluent nitrate concentrations from both cells was higher than lagoon influent nitrate concentrations throughout the experiment (Figure 50).

**Figure 48.** Time series plot of total nitrogen for lagoon influent and Queenston Shale cells 3 & 4 (with plants) effluent from March 20, 1996 to March 19, 1997. There is no MOE non-compliance level.



**Figure 49.** Time series plot of ammonia and ammonium for lagoon influent and Queenston Shale cells 3 & 4 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.



**Figure 50.** Time series plot of nitrate for lagoon influent and Queenston Shale cells 3 & 4 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.

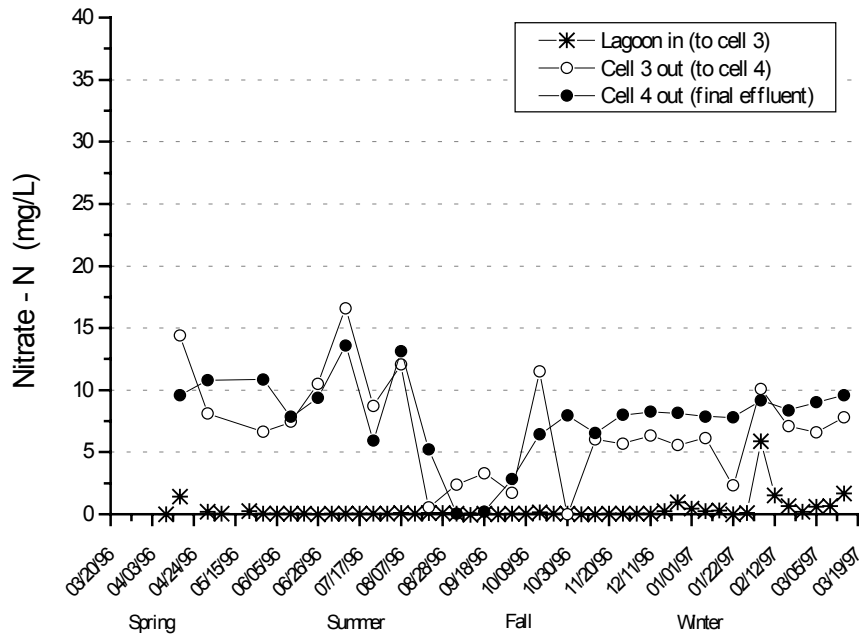
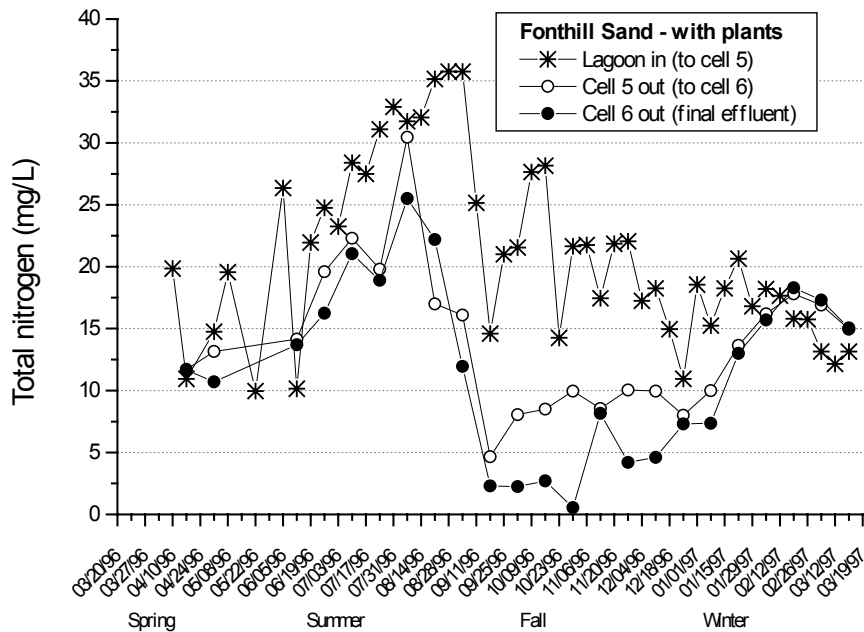
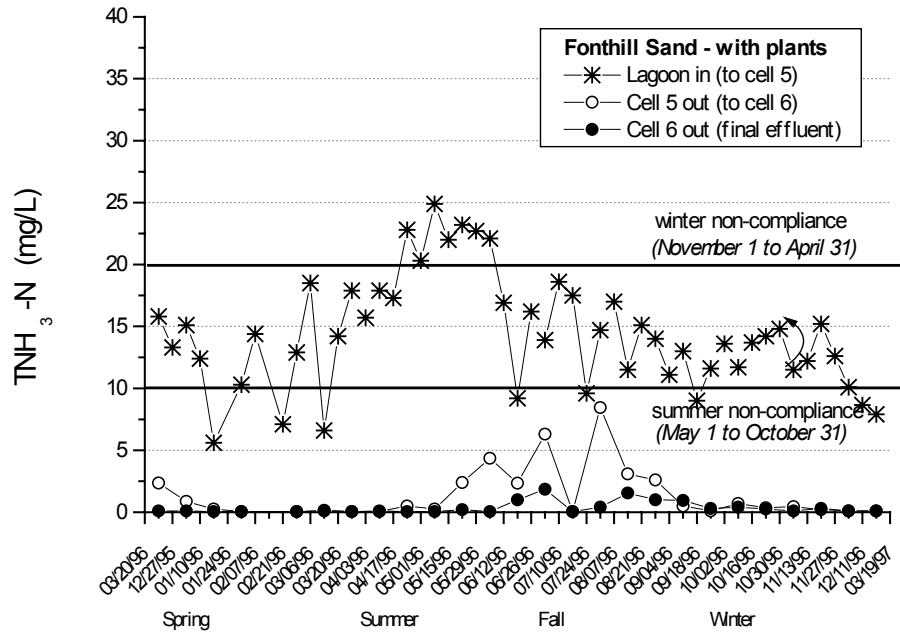


Figure 51 shows that effluent total nitrogen from the Fonthill Sand set was lower than influent total nitrogen with the exception of three sampling periods in February and March of 1997. Total nitrogen concentrations in the effluent from the first cell was higher than in the effluent from the second cell except for three times in the Summer. Effluent ammonia and ammonium from these cells was lower than lagoon influent ammonia and ammonium throughout the year and was within MOE compliance throughout the year (Figure 52). Effluent ammonia and ammonium from the first cell was in compliance throughout the year. With the exception of late September and early October, effluent nitrate concentrations from both cells were higher than lagoon influent nitrate concentrations (Figure 53).

**Figure 51.** Time series plot of total nitrogen for lagoon influent and Fonthill Sand cells 5 & 6 (with plants) effluent from March 20, 1996 to March 19, 1997. There is no MOE non-compliance level.



**Figure 52.** Time series plot of ammonia & ammonium for lagoon influent and Fonthill Sand cells 5 & 6 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.



**Figure 53.** Time series plot of nitrate for lagoon influent and Fonthill Sand cells 5 & 6 (with plants) effluent from March 20, 1996 to March 19, 1997.

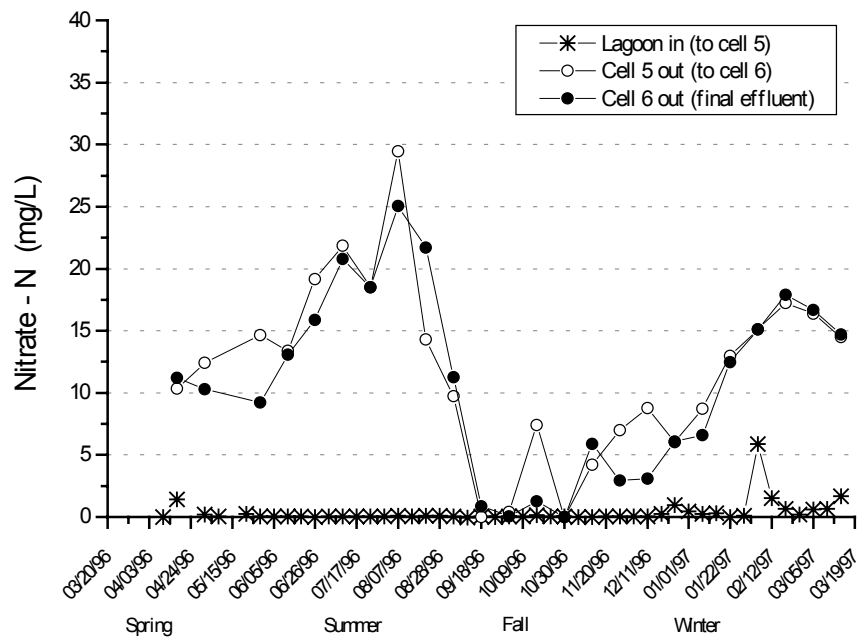
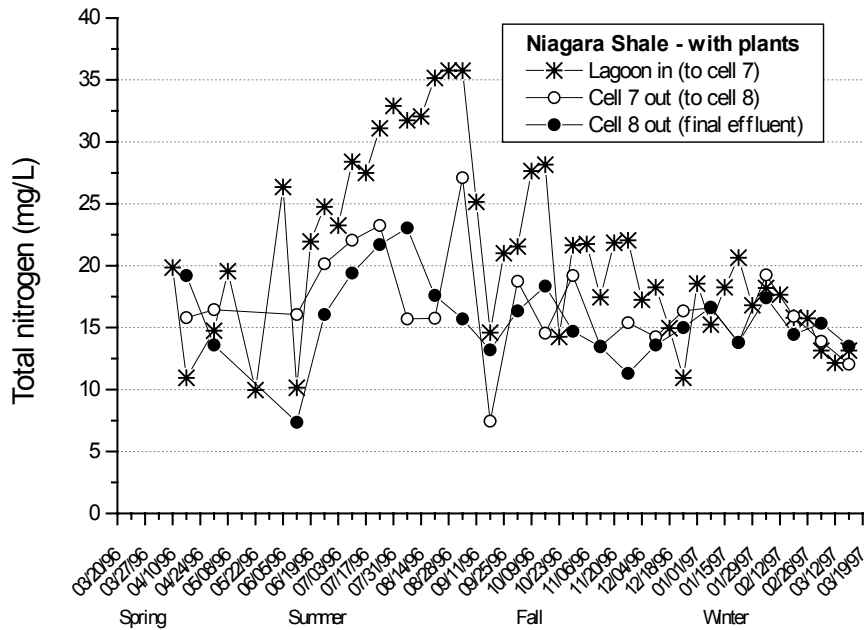
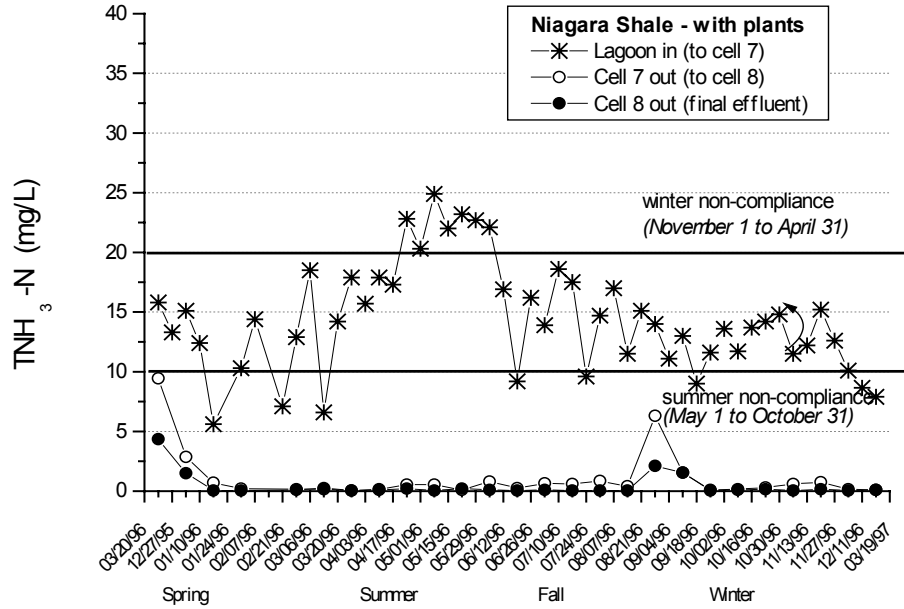


Figure 54 shows that the effluent total nitrogen from the Niagara Shale set was lower than influent total nitrogen most of the time. The exceptions to this occurred during the spring and winter when the concentration of total nitrogen in the effluent from the first and second cells was occasionally higher than influent total nitrogen. Effluent ammonia and ammonium from the Niagara Shale set was lower than lagoon influent ammonia and ammonium at all times and was within MOE compliance throughout the year (Figure 55). Effluent ammonia and ammonium from the first cell was in compliance throughout the year. Effluent nitrate concentrations from both cells was higher than lagoon influent nitrate concentrations throughout the experiment (Figure 56).

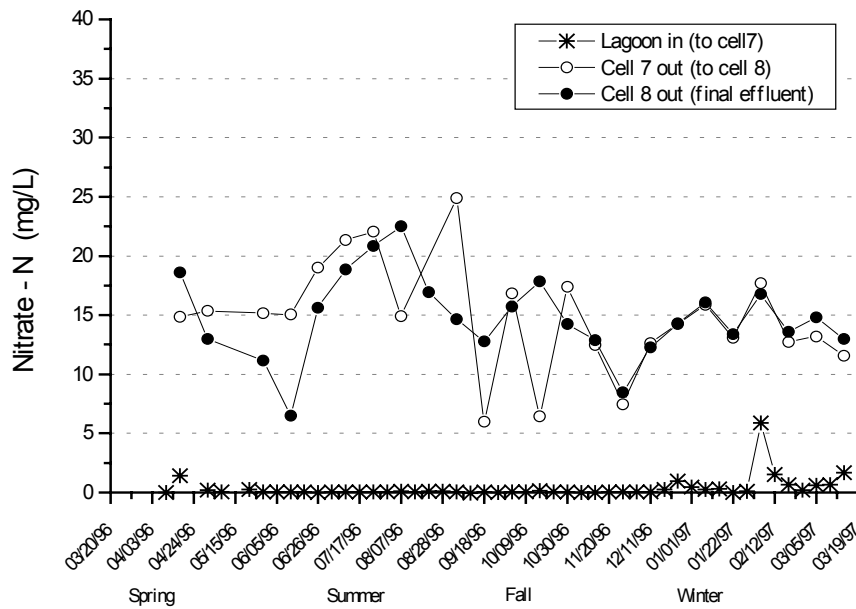
**Figure 54.** Time series plot of total nitrogen for lagoon influent and Niagara Shale cells 7 & 8 (with plants) effluent from March 20, 1996 to March 19, 1997. There is no MOE non-compliance level.



**Figure 55.** Time series plot of ammonia & ammonium for lagoon influent and Niagara Shale cells 7 & 8 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.

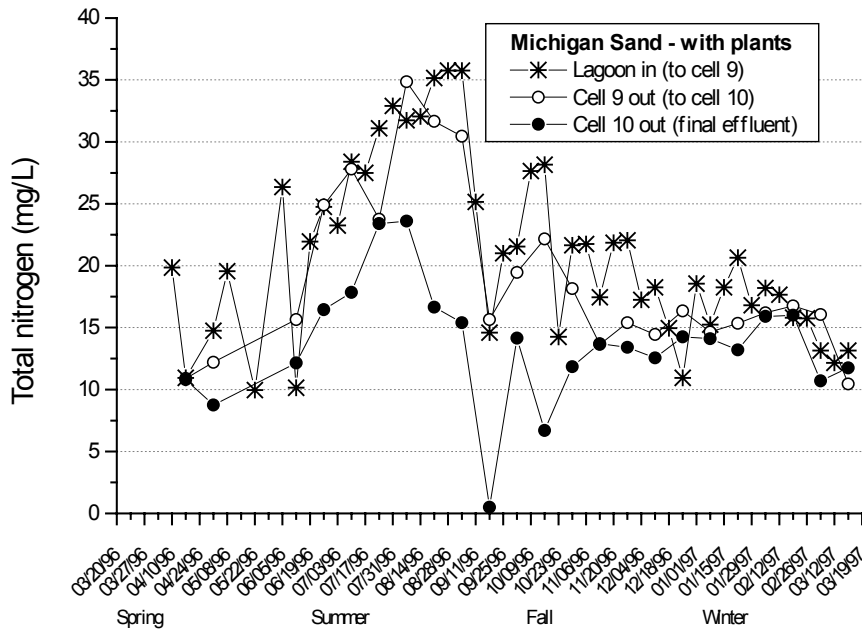


**Figure 56.** Time series plot of nitrate for lagoon influent and Niagara Shale cells 7 & 8 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.

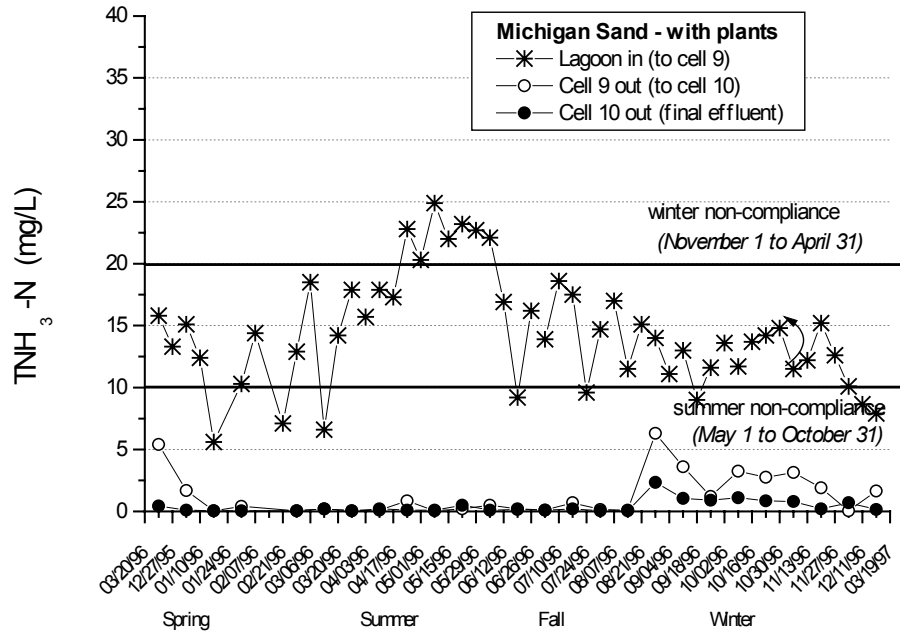


Effluent total nitrogen concentration from the first cell in the Michigan Sand set was higher than the lagoon influent total nitrogen concentration most of the time (Figure 57). Total nitrogen in the effluent from the second cell was lower than the lagoon influent concentration during Spring, Summer & Fall and higher during the Winter. Effluent ammonia and ammonium from this set was lower than lagoon influent ammonia and ammonium at all times and was within MOE compliance throughout the year (Figure 58). Effluent ammonia and ammonium from the first cell was in compliance throughout the year. Effluent nitrate concentrations from both cells was higher than lagoon influent nitrate concentrations (Figure 59).

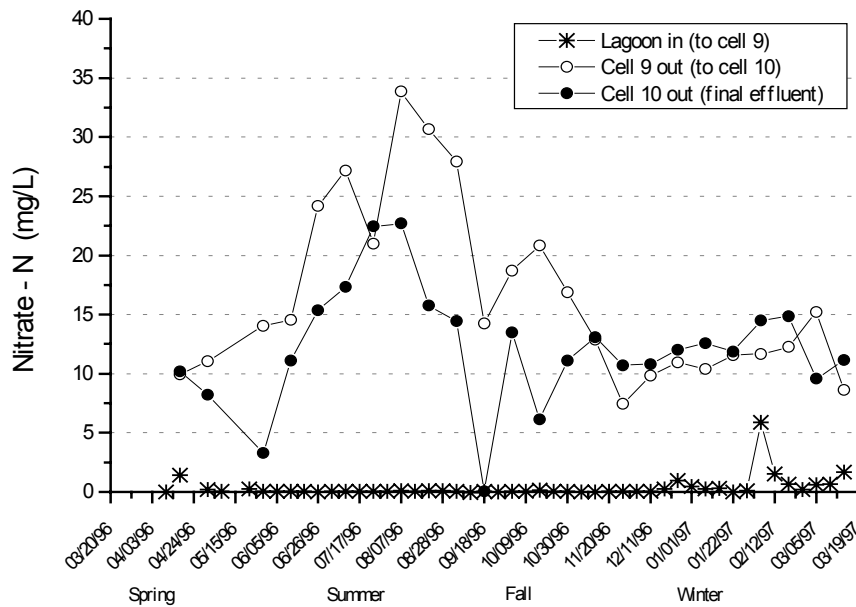
**Figure 57.** Time series plot of total nitrogen for lagoon influent and Michigan Sand cells 9 & 10 (with plants) effluent from March 20, 1996 to March 19, 1997. There is no MOE non-compliance level.



**Figure 58.** Time series plot of ammonia & ammonium for lagoon influent and Michigan Sand cells 9 & 10 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.



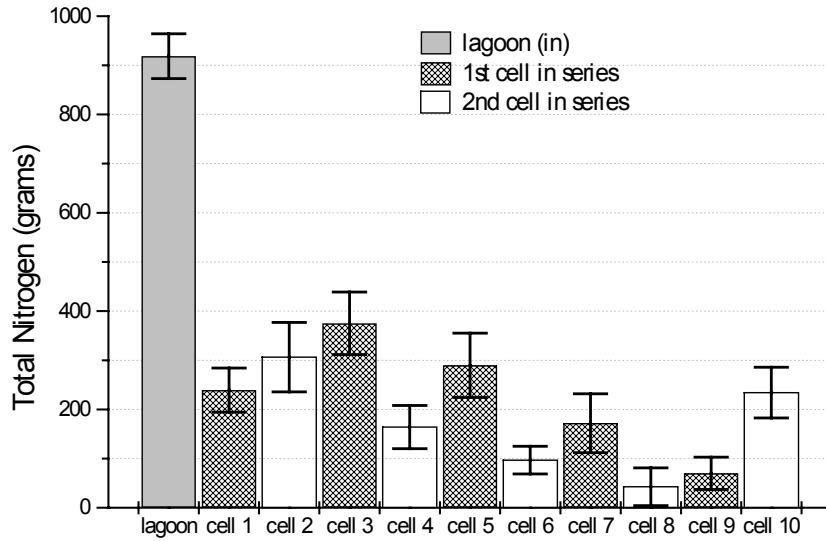
**Figure 59.** Time series plot of nitrate for lagoon influent and Michigan Sand cells 9 & 10 (with plants) effluent from March 20, 1996 to March 19, 1997. MOE non-compliance levels are shown on the graph by the solid lines.



The grams of total nitrogen removed by each cell along with the grams of total nitrogen in the lagoon influent was calculated and are presented in Figure 60. Table 20 shows the 12 month average total nitrogen concentration in the lagoon influent and individual cell effluent. A statistical comparison of the data presented in Table 20 can be found in Table 21.

Figure 60 and Tables 20 and 21 show that the first cells of the Queenston Shale set with plants, the Fonthill Sand set and the Niagara Shale set removed more total nitrogen than the second cells of those sets. The second cell of the Michigan Sand and Queenston Shale set without plants removed more total nitrogen than the first cells of those sets. The second cells of the Fonthill Sand and Niagara Shale sets and the first cell of the Michigan Sand set removed the least total nitrogen. A statistical comparison of the same cells within a set reveals no statistically significant difference among the two cells of the Fonthill Sand set or among the two cells of the Niagara Shale set. However differences were found among the cells of the remaining sets. For instance, the first cell of the Queenston Shale set with plants removed more total nitrogen than the second cell. However the opposite is true for the Queenston Shale set without plants, here the second cell removed more total nitrogen than the first.

**Figure 60.** Grams of total nitrogen removed by each cell from 03/20/96 to 03/19/97. Standard error bars are shown for the total grams of total nitrogen in the lagoon influent as well as for the total grams of total nitrogen removed by each individual cell.



**Table 20.** Mean total nitrogen concentration in the lagoon influent and effluent from each cell calculated from 03/20/96 to 03/19/97. Values in parenthesis represent the standard error.

	Lagoon influent	Q Sh cell 1	NP cell 2	Q Sh cell 3	W P cell 4	F Sa cell 5	W P cell 6	N Sh cell 7	W P cell 8	M Sa cell 9	W P cell 10
concentration (mg/L)	20.97 (1.04)	15.13 (1.37)	8.14 (0.74)	12.03 (0.81)	8.28 (0.70)	13.98 (1.21)	11.77 (1.48)	16.66 (0.84)	15.68 (0.71)	18.99 (1.42)	13.64 (1.01)

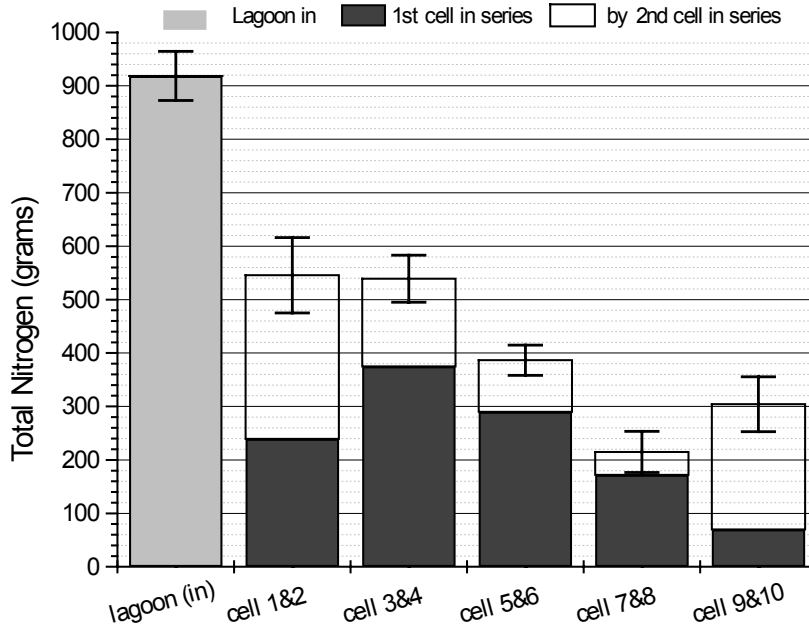
**Table 21.** Results of Student's t-tests comparing the mean total nitrogen concentration in the effluent each cell from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	cell 1 Q Sh NP	cell 2 Q Sh NP	cell 3 Q Sh WP	cell 4 Q Sh WP	Cell 5 F Sa WP	Cell 6 F Sa WP	Cell 7 N Sh WP	cell 8 N Sh WP	cell 9 M Sa WP	Cell 10 M Sa WP
Lagoon influent	0.001	0.000	0.000	0.000	0.000	0.000	0.009	0.001	0.273	0.000
cell 1 Q Sh NP		0.000	0.057	0.000						
cell 2 Q Sh NP			0.000	0.886						
cell 3 Q Sh WP				0.001	0.079	0.877	0.000	0.001	0.000	0.218
cell 4 Q Sh WP					0.000	0.038	0.000	0.000	0.000	0.000
Cell 5 F Sa WP						0.253	0.074	0.227	0.009	0.833
Cell 6 F Sa WP							0.006	0.021	0.001	0.300
Cell 7 N Sh WP								0.377	0.164	0.026
cell 8 N Sh WP									0.042	0.104
cell 9 M Sa WP										0.003

When examining total nitrogen removal by set (Figure 60 & table 22) it was found that both sets containing Queenston Shale removed 52 - 53 % of the total nitrogen, the Fonthill Sand set removed 37.11 %, the Michigan Sand set removed 28.32% and the Niagara Shale set removed 16.04% total nitrogen.

The results of Student's t-tests comparing the mean percent reduction of total nitrogen by each set of cells is shown in table 23. No statistical difference was found to exist among the Queenston Shale set without plants, the Queenston Shale set with plants and the Fonthill Sand set. No statistical difference was evident among the Fonthill Sand set and the Michigan Sand set. No statistical difference was found among the Niagara Shale set and the Michigan Sand set. The Fonthill Sand set did remove more total nitrogen than the Niagara Shale set.

**Figure 61.** Grams of total nitrogen removed by each set of cells from 03/20/96 to 03/19/97. Standard Error bars are shown for the grams of total nitrogen removed by each set of cells.



**Table 22.** Mean % reduction of total nitrogen by each set of cells from 03/20/96 to 03/19/97. Values in parenthesis are standard errors.

	set 1 Q Sh NP cells 1 & 2	set 2 Q Sh WP cells 3 & 4	set 3 F Sa WP cells 5 & 6	set 4 N Sh WP cells 7 & 8	set 5 M Sa WP cells 9 & 10
% reduction	53.03 (5.89)	52.66 (5.69)	37.11 (8.26)	16.04 (6.21)	28.32 (5.90)

**Table 23.** Results of Student's t-tests comparing the mean % reduction of total nitrogen by each set of cells from 03/20/96 to 03/19/97. All tests conducted at the 0.05% level of significance.

	Q Sh WP cells 3 & 4	F Sa WP cells 5 & 6	N Sh WP cells 7 & 8	M Sa WP cells 9 & 10
Q Sh NP cells 1 & 2	0.131			
Q Sh WP cells 3 & 4		0.104	0.019	0.037
F Sa WP cells 5 & 6			0.026	0.087
N Sh WP cells 7 & 8				0.153

## Section 5.0 — Discussion

In spite of the different physical and chemical characteristics of the four media types the sets containing them met MOE discharge guidelines throughout the year for ammonium, BOD<sub>5</sub> and suspended solids. Only the set containing Fonthill Sand medium met MOE discharge requirements for phosphorus removal throughout the year. The hydraulic conductivity through all of the media was sufficient to ensure adequate water flow through the system while maintaining satisfactory reduction of nutrients, suspended solids and organic matter.

The similarity in temperature between the lagoon influent water and effluent water from any of the cells, regardless of medium type, indicates that the wetland cell water temperature is dependent on the temperature of the influent water. This is meaningful when assessing the ability of the wetland system to operate through the winter. If the pre-treatment unit ( e.g. septic tank or lagoon) can operate without freezing through winter a properly designed and operated VF wetland treatment system can also be made to operate under these conditions.

Reduction of BOD<sub>5</sub> in this experiment was not influenced by Winter time conditions. This is not consistent with the results obtained from the FWS system in Listowel, Ontario. Here decreased BOD<sub>5</sub> reduction occurred as winter conditions developed. This phenomenon was attributed to the formation of an ice layer on the water surface that inhibited oxygenation of the wetland. Although an ice layer did form on the cell surface in this experiment, the pulse loading and vertical flow arrangement actively draws air into the system, continually oxygenating the system. This was confirmed by Lemon *et al.* (1997) and Rozema *et al.* (1996) who showed that the pulse loading and downward flow arrangement peculiar to the VF system is the predominant oxygenation mechanism of cell water. This arrangement works as follows: a pulse of water is applied to the top of the cell and allowed to percolate down through the cell. This downward

percolation physically draws air into the cell. As the next pulse of water percolates down through the cell it is oxygenated as it comes into contact with the air that was drawn down into the cell from the previous water pulse.

Lowering of  $BOD_5$  is both a physical process by way of settling and filtration of organic particles and a biochemical process through decomposition and mineralization of organic and inorganic compounds (Reed, 1995). The VF wetland allows both of these processes to occur. Particles are filtered out of the water as the water passes vertically down through the medium and the dissolved compounds undergo decomposition and mineralization by bacteria existing in the wetland cell.

These removal mechanisms explain why little difference exists among the sets when comparing the influence of medium type on lowering  $BOD_5$ . The results of this experiment are consistent with what has been reported by other researchers. For instance Reed (1995) and Watson et al. (1989) consider settling (in FWS wetlands) and filtration (in SF) wetlands to be the dominant  $BOD_5$  reduction mechanism. Steiner and Freeman (1989) found that medium type had little influence in biological degradation of organics but they did conclude that smaller root-bed particles provide greater attachment surfaces for micro-organisms that aid in the decomposition and mineralization of organic matter. This conclusion was also reached by Brix and Schierup (1989) who concluded that the chemical composition of a root-bed medium is not an important consideration when selecting a medium to reduce  $BOD_5$ .

Media type did impact removal of suspended solids. Removal of suspended solids is facilitated by much of the same processes as that of  $BOD_5$  removal, i.e., physical filtration of particles and subsequent biological decomposition and mineralization of compounds within the wetland medium (Reed, 1995). With the exception of the Niagara Shale set a two cell system was

required to consistently meet MOE discharge requirements. A reasonable explanation for the differences found in suspended solids removal among the different media may be made by examining the hydraulic conductivity of each medium. The medium with the lowest hydraulic conductivity (Fonthill Sand and Queenston Shale) removed the most suspended solids. The Niagara Shale medium, with the highest hydraulic conductivity, was the least efficient of all the media. This explanation is consistent with that reported by Faulkner and Richardson (1989) who theorized that rapid water movement through a wetland treatment cell caused the water to create and travel through channels (a process known as short circuiting) in the root bed medium resulting in a decrease in treatment efficiency. Reed (1995) supports this when he reports that short-circuiting within a SF wetland resulted in decreased suspended solids removal. The 'production' of suspended solids by the second cell of the Niagara Shale set may be due to disturbance and re-suspension of silt sized particles caused by this rapid water movement through the cell. Care should be taken when selecting a root-bed medium for removal of suspended solids and BOD<sub>5</sub>. The hydraulic conductivity and particle size has to provide adequate flow and attachment sites for bacteria while ensuring that flow is slow enough to prevent re-suspension of suspended solids.

The lagoon influent nitrogen was primarily in the form of ammonium and thus had to be nitrified (an aerobic process) to nitrates and denitrified (an anaerobic process) to nitrogen gases to be entirely removed from the wetland system. The removal of total nitrogen by a wetland system is complicated and cannot be easily explained by a single removal mechanism. In this experiment it is likely that a number of mechanisms accounted for nitrogen removal. Plants obviously removed nitrogen, at least during the growing season, in all of the planted sets. Medium physical and chemical characteristics however also

accounted for a large proportion of total nitrogen removal. The final effluent from each set easily met MOE requirements for ammonium discharge throughout the year, regardless of media type or the presence or absence of plants.

Levels of nitrate in the cell effluents were highly variable within and among the different sets. No obvious pattern could be detected. Relatively high levels of nitrate in the cell effluents indicate that the biological conversion of nitrogen is limited by insufficient anaerobic sites within the cell. Ammonia and ammonium is being nitrified to nitrate but the nitrate is not being denitrified to nitrogen gas. Additional total nitrogen reduction may have been accomplished in this experiment if more anaerobic sites were made available for denitrification of nitrates. This could be accomplished by either increasing the height of the water table in the first cell thus leading to more saturated conditions or changing the flow direction within a cell from downward to upward i.e. flooding the bottom and draining from the top, or adding an additional cell to a set (Lemon *et al.*, 1997).

A number of explanations can be given to account for the differences in removal of total nitrogen among the sets containing different media. The removal of the ammonium ion from solution can occur by ion exchange onto negatively charged soil particles of the root-bed medium, biological conversion to nitrogen gas via nitrification-denitrification reactions and plant uptake (Lemon *et al.*, 1997; Mitsch & Gosselink, 1986; Steiner & Freeman, 1989). Lemon *et al.* (1997) hypothesized that the Queenston Shale medium acts as an 'ammonium sponge' because of its relatively high cation exchange capacity. In times of warm temperature the processes of nitrification-denitrification and plant uptake are the dominant nitrogen removal mechanisms. As temperatures begin to cool at the onset of Winter biological activity slows and excess ammonium is then adsorbed onto the negatively charged clay particles of the shale. As Winter progresses the exchange sites gradually become saturated with ammonium. As temperatures

begin to rise in Spring with a simultaneous increase in biological activity, the ammonium is stripped off its exchange sites and is either nitrified or taken up by plants renewing the exchange sites for next Winter. The Cation Exchange Capacity of the Queenston Shale was highest, measured at 8.7 cmol + /kg followed by the Niagara Shale at 8.1 cmol + /kg and the Fonthill Sand at 4.7 cmol + /kg (table 2). The Michigan Sand did not have an appreciable Cation Exchange Capacity. The similarity in Cation Exchange Capacity of the two shale media and the distinct differences in removal efficiency of these shales (the Queenston Shale removed 52.66 % as opposed to the Niagara Shale's 16.04 % removal of total nitrogen demonstrates that Cation Exchange Capacity alone cannot be used to account for nitrogen removal by these media. The same may be said for the Michigan Sand medium. Although this medium does not have an appreciable Cation Exchange Capacity and does not possess ammonium adsorption capabilities. The set did remove the same amount of nitrogen as the Niagara Shale.

Because the medium Cation Exchange Capacity cannot solely be used to explain nitrogen removal in this experiment, other mechanisms such as biological conversion of nitrogen into gas which results in its removal from the system, must be at work. An examination of the medium surface area and subsequent microbial attachment sites located in the media illustrates an additional nitrogen removal pathway. The set containing media with the largest surface areas removed the most total nitrogen. The Queenston Shale surface area is 93 m<sup>2</sup>/gm and the Fonthill Sands is 53 m<sup>2</sup>/gm compared to the Niagara Shale's 18 m<sup>2</sup>/gm and the Michigan Sand's 3 m<sup>2</sup>/gm (table 2). Steiner and Freeman (1989) concluded that the medium provides surface area for micro-organism attachment and Webber (1990) estimated that anywhere from 10 to 100 million bacteria per gram of soil is possible depending on surface area. Burgoon *et al.* (1991) state

that the population of bacteria supported within a given volume of medium may generally be assumed to be proportional to the surface area of that medium. They found that removal of nitrogen in some instances was higher and less variable in media with the highest surface area.

The Langmuir Adsorption Isotherms for ammonium (Figure 10) may be used with limited success to predict removal of nitrogen when selecting a root-bed medium. For instance these isotherms correctly predicted that the Queenston Shale would adsorb the most ammonium to be followed by the Fonthill Sand and then the Niagara Shale. However the Michigan Sand did not conform to the Langmuir Adsorption Isotherm but still removed nitrogen; evidence that plant uptake and/or biological nitrification- denitrification is an important nitrogen removal mechanism.

When comparing the removal of phosphorus by sets containing different medium types it was found that the Fonthill Sand set removed significantly more phosphorus than the remaining media. Only this set consistently met MOE discharge criteria. It is unclear from the data obtained in this experiment as to the precise factors contributing to the large differences found in the phosphorus removal capability of the Fonthill Sand when compared to the other root-bed medium. The Fonthill Sand medium had a much lower concentration of  $\text{CaCO}_3$  than the Niagara Shale medium yet it removed approximately 45% more phosphorus than the Niagara Shale. The Fonthill Sand had almost double the surface area of the Queenston Shale medium but removed 35% more phosphorus than the Queenston Shale. A textural analysis revealed that the Fonthill Sand was virtually identical in texture to the Michigan Sand but removed approximately 45% more phosphorus than the Michigan Sand.

An additional disparity exists when comparing the results of this experiment to what was predicted to occur via the Langmuir Adsorption Isotherm

for phosphorus. The Michigan Sand medium had a low phosphorus adsorption capacity of only 1.71 mg P / kg of medium as predicted by the Langmuir adsorption isotherm yet it removed the same amount of phosphorus as the Queenston and Niagara Shale which had much higher predicted phosphorus adsorption capacities of 21.03 and 25.43 mg P / kg of medium respectively (Figure 9). The Fonthill Sand adsorption capacity was predicted to be only 16.97 mg P / kg of medium yet this set removed between 35 - 45 % more phosphorus than any other medium used in this experiment.

It is possible that the Fonthill Sand possesses other physical and chemical attributes that were not measured in this experiment. It may also be possible that the interaction of the plants with a specific medium greatly modified its ability to sequester phosphorus. It has been shown by Lemon *et al.* (1997) that plants do increase the redox status of the root-bed in a VF wetland cell. Of course the plants themselves also sequestered phosphorus, at least during the growing season.

The presence of plants did influence total phosphorus removal. With the onset of Fall and plant senescence the phosphorus content of final effluent from all the sets, except the Fonthill Sand, dramatically increased, emphasizing the important role plants play in phosphorus removal. The Queenston Shale set without plants removed approximately 34 % of the total phosphorus, the identical set with plants removed approximately 22 % more. The varying importance of plants and wetland medium in the removal of phosphorus is reported on in the literature as well. Reddy and DeBusk (1987) found that plants can account for 12 to 73 % of total phosphorus removal, depending on factors such as the plant "growth rate .... and physio-chemical characteristics of sediment and water." Plants accounted for 67.3 % of the total phosphorus removal in a SF wetland when a root-bed medium with a low affinity for phosphorus adsorption was used

(Breen, 1990). In a bench scale experiment Burgoon *et al.* (1991) found that removal of phosphorus was dependent on both the plant species and medium type. They found that unplanted pots and pots containing Bulrush (*Scirpus sp.*) and Reed grass (*Phragmites sp.*) removed less phosphorus than pots containing cattails (*Typha sp.*) and Arrowhead (*Sagittaria sp.*). Phosphorus removal is dependent on both plants and wetland medium and the importance of one over the other seems to depend on the individual characteristics of the plants and wetland medium.

Plant uptake and storage of phosphorus can be used to account for the difference in levels of phosphorus removal between planted and unplanted sets. Throughout most of the plant growing season the phosphorus effluent concentration from the planted set was < 1mg/L while effluent from the unplanted set was > 1 mg/L. When the plants began to die off in the fall the phosphorus concentration from the planted set began to exceed 1 mg/L and approached values similar to the unplanted set.

Although the Michigan Sand medium had the lowest phosphorus adsorption capacity, the lowest CaCO<sub>3</sub> content and the lowest surface area, it removed similar amounts of phosphorus as both the Queenston Shale and Niagara Shale. This indicates that luxury uptake of phosphorus by plants is probably occurring. Plants were identified by Lantzke *et al.* (1999) as a long term removal mechanism in a VF wetland in Australia.

The role that plants play in temperature regulation of the wetland cells was not made clear in this experiment because the set without plants was insulated with a straw layer during winter. However, in a thermal analysis of a sub-surface vertical flow wetland system at Niagara -On-The- Lake, Ontario Smith *et al.* (1997) found that plants provided an insulating thatch layer. These authors constructed a thermal model and predicted that if a 20 cm thatch layer was

present on the top of the cell, the wetland system could operate at temperatures as low as  $-50^{\circ}\text{C}$ .

No difference exists in the cell effluent dissolved oxygen concentration between the unplanted and planted sets. Based on  $\text{BOD}_5$  reduction data, approximately 10 % more oxygen was made available in the planted set indicating that the plants did supply oxygen ( recall that  $\text{BOD}_5$  = biochemical oxygen demand or the oxygen consumed during the oxidation of organic and inorganic matter). The contribution of oxygen by plants to a wetland treatment system is consistent with what other authors report. In a FWS wetland Gersburg *et al.* (1989) calculated that aquatic plants added 7.2 grams of oxygen per square meter of wetland surface area and also that the supply of oxygen to these wetlands by plants was in excess of what was needed to satisfy aerobic respiration. Watson *et al.* (1989) attribute increasingly efficient  $\text{BOD}_5$  reduction in a SF wetland to the presence of an expanded aerobic root-zone made possible as aquatic plant roots penetrate deeper into the root-bed of constructed wetlands.

The planted set removed approximately 14% more suspended solids than the unplanted set. These results do not agree with what was found by Tanner *et al.* (1995) who reported no difference in suspended solids removal between planted and unplanted cells of SF wetland treatment system. The difference may be explained, in part, by the organic component of suspended solids. The planted set did reduce more  $\text{BOD}_5$  and thus will remove more organic suspended solids than the unplanted set. Among the planted wetland sets the Queenston Shale, Fonthill Sand and Michigan Sand sets were as efficient as or more efficient than a number of wetland treatment systems reported on in the literature. Gearhart V(1989) found that the FWS wetland located in Arcata, California could remove approximately 85% of the suspended solids. They attributed this removal

primarily to the settling out of solids. The FWS at Listowel, Ontario had a mean suspended solids reduction of 58% over the first three years and 91 % in years four and five. The suspended solids removal of the SF wetlands in Norway averaged 73% for an eighteen month period.

No difference was found to exist in the amount of total nitrogen removed between the unplanted and planted sets. This is not consistent with what is routinely reported on in the literature. For instance, Conway and Murtha (1989) found that the supply of oxygen to a wetland by plants is the most critical factor in nitrogen removal. In a study comparing vegetated to unvegetated FWS wetlands these authors found the vegetated wetlands to be 80% more efficient in the removal of ammonia and ammonium nitrogen. Tanner *et al.* (1995) propose that plant roots may modify redox potentials by oxygenation of the medium thus enhancing microbial decomposition and nitrification. Reed and Brown (1995) propose that the plant root/rhizome structure provides a substratum for microbial activity and nitrification and Gersburg *et al.* (1984) state that "the most successful procedure for removal of nitrogen from wastewater is sequential nitrification-denitrification...".

In this experiment the VF wetlands were routinely supplied with oxygen via the flooding and draining mechanism used, ensuring that oxygen supply was not limiting nitrogen the nitrogen cycle. In fact, the nitrate content of the treated water indicates that nitrogen cycling was limited by the number of anaerobic sites in the wetland cells. In spite of this, results from some of the sets in this experiment are better than or equal to those obtained from other systems. The FWS system at Listowel, Ontario removed 35 % of the total nitrogen over the first three years and 43 % in years four and five (Kadlec, 1995). The SF systems in Norway removed between 48 and 59 % of total nitrogen (Jenssen *et al.*, 1992). Brix and Schierup (1989) reviewed twenty-five SF systems in Denmark and found that

total nitrogen removal efficiencies ranged between 25 to 50%. Because no difference exists among the planted and unplanted sets, nitrogen removal via nitrification-denitrification is likely the dominant removal mechanism in the Queenston Shale cells.

The results of this experiment show that the VF wetland has the potential for wide use in warm climates as well as in cold climates. These results also indicate that the VF system can be used successfully over a wide geographical area, not being limited by the availability of a specific medium type. The data in this experiment were compared to MOE discharge guidelines to surface waters, which are much more rigorous than those for subsurface discharge. In many instances in Ontario, treated water is discharged subsurface into a tile bed system. In these instances phosphorus concentration is not an issue. In the Province of Ontario, the Ministry of Housing regulates systems using subsurface discharge systems with total flows below 10,000 L/day. The performance of these systems is only assessed using water quality data based on BOD<sub>5</sub> and suspended solids concentration. According to this Ministry's guidelines, tertiary treatment of domestic sewage occurs when the suspended solids concentration is below 10 mg/L and BOD<sub>5</sub> is below 15 mg/L (O.B.C., 1992). At these levels the water is considered suitable for discharge. No guidelines exist for nitrogen or phosphorus discharge. In the Province of Ontario the VF system containing a medium selected only on the basis of adequate hydraulic conductivity, can be used successfully to treat domestic sewage to meet Ministry of Housing standards for tertiary treatment. Discharge of treated water to surface waters (regulated by the MOE) would require attention to phosphorus removal. In these instances care should be taken to select a medium that will remove phosphorus. If possible the selection of this medium should be based past experience or on the results of small pilot projects set up similar to this experiment.

## Section 6.0 — general conclusions:

### dissolved oxygen

- Plants did contribute oxygen to the wetland.

### BOD<sub>5</sub> reduction

- The planted set was approximately 10 % more efficient than the unplanted set.
- Medium type had no influence on BOD<sub>5</sub> reduction.
- All of the sets consistently met MOE requirements for BOD<sub>5</sub> reduction throughout the year. The 12 month average BOD<sub>5</sub> concentration from all sets with plants was below 2.36 mg/L.
- The Queenston Shale sets required two cells to meet MOE discharge requirements, all of the other sets met these requirements with one cell.

### suspended solids reduction

- The planted set was approximately 14% more efficient than the unplanted set.
- The Queenston Shale and Fonthill Sand media was more efficient than the Niagara Shale and Michigan Sand media.
- The 12 month average suspended solids concentration from the Queenston Shale and Fonthill Sand media was below 2.16 mg/L.
- The Queenston Shale, Fonthill Sand and Michigan Sand sets required two cells to meet MOE discharge requirements; the Niagara Shale set met these requirements with only one cell.
- Media with a low hydraulic conductivity is more efficient in the removal of suspended solids.

### total phosphorus reduction

- The planted set was approximately 22% more efficient than the unplanted set.
- Regardless of medium type, removal of total phosphorus was better during the plant growing season.
- The Fonthill Sand set was by far the most efficient, removing 91.60% of the

total phosphorus added to it with an average 12 month effluent concentration of 0.23 mg/L.

- The Fonthill Sand set was the only set to consistently meet MOE discharge requirements year-round.
- Total phosphorus removal was best when the medium contained CaCO<sub>3</sub> and had a larger surface area.
- The Langmuir Adsorption Isotherm or medium physical and chemical characteristics cannot adequately predict phosphorus removal ability.

#### nitrogen reduction

- The presence of plants made no difference in overall reduction of total nitrogen but plants did affect how nitrogen was cycled through the system.
- Removal of total nitrogen was best in the Queenston Shale and Fonthill Sand sets.
- All of the media consistently and easily met MOE discharge requirements with respect to ammonium concentration year-round.
- A large medium surface area (with accompanying large populations of bacteria), a high medium CEC and the ability to adsorb ammonia as predicted by the Langmuir adsorption isotherm likely contribute to significant nitrogen removal.

#### general

- Overall, the sets containing Queenston Shale and Fonthill Sand media performed the best.
- Plants increased treatment efficiency for every parameter except total nitrogen.
- The VF wetland treatment system investigated here was generally superior to the FWS and SF systems in other cold climate situations.
- Vertical flow wetland sewage treatment systems can be designed and built to consistently and easily meet MOE discharge requirements throughout the year for BOD<sub>5</sub>, suspended solids, total phosphorus and ammonium.

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