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The Role of a Temperate Constructed Wetland in Nutrient Mitigation

Le rôle de la construction d'un marais tempéré dans l'atténuation des impacts sur les nutriants

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ABSTRACT

Constructed wetlands are designed as treatment facilities for pollutants in non-point source runoff. This study assessed the effectiveness of a newly constructed 4.2 hectare wetland in Eastern Ontario as a sink for nutrients within an urban fringe watershed. During the initial two years after construction, the wetland was a sink for total phosphorus (TP) during the spring (21 to 53%), summer (48 to 57%) and early fall (21%). However, it was a large source of TP during late fall (-1869%). During the spring, summer and early fall the wetland was a minor sink (19 to 28%) for total nitrogen (TN), but it was an effective mechanism for reduction of TN in late fall (94%). Anoxic conditions in November of 1997 in one of the wetland's two distinct cells, Cell 2, were likely the cause of both a major release of TP from the sediments and a major reduction of TN within the water column with the release of N to the atmosphere through denitrification. These major changes were only found in this second wetland cell. Extensive submerged aquatic plant growth in Cell 1 may have been the cause of delayed late fall anoxia. An analysis of the nutrient retention capabilities of plant litter indicated that sediments of lower organic content may be more effective at P retention and although TN concentrations increase as the organic content of the litter increases, mixed and cattail litter show little variation at different stations and from month-to-month over the study period. Naturally accumulating mixed litter was found to be equally effective as deadfall cattail litter at TP and TN retention during the spring and summer, but significantly better at TN retention during the fall. An off-site source of leaf litter was significantly less effective than mixed or cattail litter at phosphorus retention during the growing season. The very different reactions of P and N to late fall anoxia in 1997 imply the need to focus design and management of constructed wetlands on the nutrient of major concern within the watershed.

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RÉSUMÉ

Les marais artificiels sont conçus comme installations de traitement pour les polluants dans une source non ponctuelle d'écoulement. Cette étude évalue l'efficacité d'un marais de 4,2 hectares récemment construit dans l'Est de l'Ontario comme puits d'éléments nutritifs dans un bassin en milieu périurbain. Au cours des deux années suivant sa construction, le marais a constitué un bassin de phosphore total (PT) au cours du printemps (21 % à 5 %), l'été (48 % à 57 %) et le début de l'automne (21 %). Cependant il constituait une source importante de PT au cours de l'arrière-automne (-1 869 %). Au cours du printemps, de l'été et du début de l'automne, le marais constituait un bassin mineur (19 % à 28 %) d'azote total (AT), mais il constituait aussi un mécanisme efficace pour la diminution de AT au cours de l'arrière-automne (94 %). En novembre 1997, les conditions anoxiques de l'une des deux cellules distinctes du marais, soit la cellule 2, ont provoqué une libération importante de PT provenant des sédiments et une diminution importante de l'AT dans la colonne d'eau avec la libération d'azote (N) dans l'atmosphère par le biais de la dénitrification. Ces changements importants ne se produisaient que dans la deuxième cellule du marais. La croissance considérable des plantes aquatiques submergées dans la cellule 1 peut avoir causé le retard de l'anoxie en arrière-automne. Une analyse des capacités de rétention des nutriants de la litière végétale indiquait que des sédiments de plus faible contenu organique peuvent être plus efficace dans la rétention du P; bien que les concentrations de AT augmentent à mesure qu'augmente le contenu organique de la litière, la litière mixte ou du roseau des étangs démontre peu de variation à des stations différentes, et de mois en mois, au cours de la période de l'étude. On a découvert que la litière mixte qui s'accumule naturellement était aussi efficace que la litière enchevêtrée de roseau des étangs pour la rétention de PT et de AT au cours du printemps et de l'été, mais significativement meilleure pour la rétention de AT au cours de l'automne. Une source hors terrain de litière de feuilles était significativement moins efficace que la litière mixte ou de roseau pour la rétention du phosphore au cours de la saison de croissance. Les réactions très différentes de P et de N à l'anoxie d'arrière-automne en 1997 laissent supposer le besoin de se concentrer sur la conception et la gestion de marais construits sur le nutriant le plus important du bassin.

Chapter 1 - INTRODUCTION

Constructed wetlands are becoming a popular choice for municipalities and industries as a way of meeting increasingly stringent surface water discharge regulations (Pries, 1996). While constructed wetlands may be designed to mitigate both water quantity and quality impacts, their primary goal is usually to improve or maintain the quality of water flowing to downstream areas by acting as a sink for upstream contaminant and nutrient loading. In addition to the relatively low cost and ease of operation of such treatment systems, as compared to conventional water treatment facilities, constructed wetlands are a compelling choice for politicians and administrators, because they are also perceived as providing significant recreational and environmental benefits to the community (Arnell and Eastlick, 1996).

This paper assesses the effectiveness of a newly constructed temperate wetland as a sink for phosphorus and nitrogen, the two most important nutrients from a water quality perspective. Phosphorus plays a major role in the eutrophication of aquatic systems (Ward and Elliot, 1995) and watersheds subject to extensive human impacts can become nitrogen-saturated, with resulting conversion of ammonium to nitrate releasing hydrogen ions that contribute to acidified soil and water (Vitousek et al., 1997). This research attempts to provide some answers to the following questions: whether we can rely on constructed wetlands to function as sinks for nutrient loads directed toward them; whether constructed wetlands are equally effective as sinks for phosphorus and nitrogen; and what role accumulating plant litter plays in nutrient retention. The answers to these questions are important to optimizing the design and management of constructed wetlands in a manner that will improve their effectiveness in nutrient mitigation.

To provide some context for this research, the following presents a brief account of why phosphorus and nitrogen are considered important water pollutants, a summary of the main pathways by which these nutrients are thought to cycle through wetland ecosystems, and what research studies in other aquatic systems can lead us to expect concerning the nutrient retention capacity of constructed wetlands. The role of plant litter in nutrient mitigation is also reviewed. The accumulation of litter is considered by some researchers to be the most significant contribution of plants to pollutant mitigation in wetlands (Faulkner and Richardson, 1989; Davis, 1991).

1.1 Natural and Constructed Wetlands

Environment Canada (1987) defines a wetland as "*...land that is saturated with water long enough to promote wetland or aquatic processes, as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment.*"

Wetlands are classified according to a range of distinctive hydrological and vegetative characteristics. Although these classifications differ somewhat from country-to-country, the Canadian Wetland Classification System (Environment Canada, 1987) sets out five main types, defined by Mitsch and Gosselink (1993) as follows.

1. Bogs: peat-accumulating wetlands with no significant inflows or outflows.
2. Fens: peat-accumulating wetlands receiving some drainage from surrounding mineral soils.
3. Marshes: frequently or continually inundated wetlands characterized by emergent herbaceous vegetation adapted to saturated mineral soil.
4. Swamps: wetlands dominated by trees or shrubs (includes reedswamps).
5. Shallow Waters: usually contain water less than 2 metres deep.

It is generally accepted that natural wetlands should not be used for pollutant mitigation (Schueler, 1992), primarily because of the largely unknown risks posed to wetland biota by the widespread use of wetlands for pollutant mitigation and the associated contaminant accumulation. If this is the case, the desirable habitat created by these wetlands may lead to toxicity (Willard and Hiller, 1990; Ohlendorf, 1996; Wren et al., 1996). Moreover, in many parts of Canada, natural wetlands are scarce. According to the Government of Canada (1991), since 1800 approximately one seventh of Canada's total natural wetland base has been drained or lost to other functions. This includes the loss of 80-98% of wetlands immediately within or adjacent to most Canadian urban centres, areas where pollutant mitigation is most needed. Other countries have similar or worse records of natural wetland degradation and loss.

As an alternative, the possibility arises of creating artificial wetlands whose placement and "natural" functioning are specifically designed for water quality mitigation.

Mitsch and Gosselink (1993) classify constructed wetlands designed for water quality mitigation on the basis of the type or source of water being treated, namely:

a) Wastewater;

b) Mine drainage;

c) Stormwater and nonpoint source runoff.

Beyond these three categories, there are three generally recognized methods for the design of constructed wetland treatment systems (Pries, 1994):

1. Free-water, surface flow wetlands, which typically simulate natural cattail marshes;
2. Subsurface flow wetlands, where water is directed through a vegetated gravel or soil bed;

3. Floating aquatic plant system, usually dominated by highly productive species such as water hyacinth or duckweed.

A 1994 telephone survey by Pries (1996) identified 67 constructed wetlands either built or under development across Canada, in all provinces and territories, with treatment applied to municipal and industrial wastewater, and farm, agricultural and municipal stormwater runoff. All three constructed wetland types were represented in the survey. The survey did not include the identification of acid mine treatment wetlands.

As the science and engineering of such artificial wetlands evolves, many involved in their design and management are looking to maximize the limits of their effectiveness for pollutant mitigation. Several researchers (Schueler, 1992; and Mitsch, 1993) have tried to determine if constructed wetlands are more effective in pollutant mitigation than natural wetlands. While on the surface constructed wetlands appear to be significantly more effective than natural wetlands, comparisons have been between large natural wetlands of long standing occurrence and much smaller artificial wetlands of recent construction. Therefore, both Schueler (1992) and Mitsch (1993) suggest that assessments would seem to indicate that differences in the size, age, loading rates, and complexity, among other factors, make an equitable comparison of this type highly problematic.

1.2 Nutrients as Pollutants

1.2.1 Phosphorus

The major environmental concern associated with increased concentrations of phosphorus in water is eutrophication, the accelerated growth of phytoplankton and suspended and rooted plants (Ward and Elliot, 1995). Increased algal growth decreases water clarity, blocking out light penetration into the water column. The die-off of extensive plant material provides the substrate for aerobic microbial activity, leading to oxygen depletion. Decreased oxygen levels then decrease habitat quality for other aquatic plant and wildlife species, resulting in reduced biodiversity in wetlands (Mackenzie et al., 1998). Phosphorus is considered a particular problem since it is typically the growth limiting nutrient in aquatic systems. Because phosphorus is often the limiting nutrient in aquatic systems, increased phosphorus loading generally results in eutrophication (Loucks et al., 1977; Riemer, 1984).

The main source of phosphorus in aquatic systems is the runoff of phosphate-laden fertilizers and detergents from agricultural and urban areas. Clearing of forested areas for land development and cultivation for agricultural purposes increases soil erosion, which results in increased phosphorus loading in the runoff sediment (Ward and Elliot, 1995; Mackenzie et al., 1998).

According to Pries (1996), water quality monitoring by the Ontario Ministry of Environment and Energy is most concerned with phosphorus removal efficiency and subsequent release by wetlands in the spring and fall. The Ministry considers seasonal discharge to be more acceptable than continuous, year-round discharge.

1.2.2 Nitrogen

According to Vitousek et al. (1997), Ward and Elliot (1995), and Horne and Goldman (1994), nitrogen, in the form of nitrates, is a major water pollutant, particularly where human inputs are an important factor. Vitousek et al. (1997) report that during the past century, increasing production of nitrogen fertilizers, the combustion of fossil fuels and the cultivation of legume crops have doubled the rate of annual nitrogen availability. Previously nitrogen was a major limiting factor controlling many biological processes, requiring nitrogen-fixing organisms or lightning to form inorganic annomium (NH_4) and nitrate (NO_3) that is biologically available.

Vitousek et al. (1997) also report that ecosystems subject to high rates of nitrogen deposition can become nitrogen-saturated as its retention by soils, plants and microbes is limited. Excess nitrogen is then transported into water bodies, groundwater and the atmosphere. Water-soluble, negatively charged nitrates carry with them positively charged minerals which results in a depletion of soil fertility. When large amounts of nitrogen are deposited in low nitrogen systems, species which had evolved to survive within nitrogen limited environments are typically replaced by species which are adapted for the enriched nitrogen conditions. Through denitrification, nitrogen saturated soils also exhibit enhanced emissions of nitrous oxides. Concerns related to the effects of acid rain have focused on reducing sulfur dioxide emissions, with almost no attention being given to nitrogen oxides. This oversight is becoming more important with the recognition that many watersheds are approaching nitrogen saturation, with resulting acidified soils and water. In northern climates, nitric acid can accumulate in the snow, resulting in an acid pulse with the first flush of spring meltwater.

Although nitrate is typically not toxic in concentrations below 10 mg liter^{-1}, at high levels it can be toxic to infants (Vitousek et al., 1997; Horne and Goldman, 1994, and Ward and Elliot, 1995). Young children, with their low digestive activity, reduce nitrate to nitrite, which then binds with hemoglobin in the blood to restrict the transfer of oxygen from the lungs. Ammonia can be highly toxic to plants and animals, especially at elevated pH levels when toxic ammonium hydroxide forms (Horne and Goldman, 1994). However, in most aquatic systems ammonia levels are generally below 0.1 mg liter^{-1}, so that detrimental impacts associated with naturally occurring ammonia in non-point source runoff are uncommon. Toxic effects are mostly due to pollution associated with sewage outfalls emitting ammonia at levels in the range of 10 to 30 mg liter^{-1}. After a few days in an aquatic environment these point sources become diluted or transformed to non-toxic levels or forms.

1.3 Nutrient Cycling

1.3.1 Phosphorus

Phosphorus is a naturally occurring element in soil, with >99% occurring in the solid phase, composed of organic P, Fe, Al, and Ca phosphates. Less than 25% of the solid phase is in dynamic equilibrium with the solution phase through sorption-desorption, precipitation-dissolution, or immobilization-mineralization reactions (Ward and Elliot, 1995).

The phosphorus cycle has no atmospheric component. Therefore, permanent removal of phosphorus from a wetland system can only be achieved by such mechanisms as physical harvesting and removal of plants, or by allowing a flush-out of phosphorus from the water column between growing seasons (Sloey et al., 1978).

Boström et al. (1988) outlines six major transfer mechanisms for the deposition of phosphorus in sediments:

1. sedimentation of detrital phosphorus minerals originating from within the watershed;
2. adsorption or precipitation of phosphorus with inorganic compounds,
    a) coprecipitation with iron and manganese ,
    b) adsorption to clays,
    c) association with carbonates;
3. sedimentation of phosphorus with allochtonous organic matter (metal mediation may play a significant role);
4. sedimentation of phosphorus with autochtonous organic matter;
5. direct uptake by assimilation of phosphorus from water column by periphyton and other surficial sediment biota;
6. direct adsorption of water dissolved phosphorus onto sediment particles.

Mitsch and Gosselink (1993) state that the iron and aluminium content of a wetland soil exerts significant influences on the ability of that wetland to retain nutrients such as phosphorus. Welch and Lindell (1992) report that the most significant process determining when and to what extent lake sediments will act as a source for phosphorus is iron redox, whether the lake is stratified, unstratified, oxic or anoxic. Low oxygen converts oxidized ferric iron (Fe^{3+}) to ferrous iron (Fe^{2+}), which changes insoluble ferric phosphate precipitates to soluble ferrous iron and phosphate (Horne and Goldman, 1994). Welch and Lindell (1992) report that temperature can be an important driving force in the release of phosphorus, especially from organic sediments. However, Kamp-Nielsen (1974) found that the role of temperature is related to the stimulation of bacterial activity, which in turn creates anoxic conditions and the reduction of iron.

Nichols (1983) found that chemical adsorption dominated at equilibrium concentrations of phosphorus up to about 1 mg/L. Above this threshold, physical adsorption predominated. Physical adsorbed phosphorus is not held as tightly by soil as is chemically adsorbed phosphorus. Physical and chemical adsorption is a rapid process, most of which occurs in a minute, with some slower reactions continuing to remove phosphorus from solution for periods of from several days to several months.

1.3.2 Nitrogen

Nitrogen occurs in a gaseous cycle, readily changing form in the system and moving between the soil-water-air compartments (Mitsch and Gosselink, 1993; Horne and Goldman, 1994; and Ward and Elliot, 1995). The nitrogen cycle in wetlands is dominated by the sequential processes of mineralization, nitrification and denitrification. Mitsch and Gosselink (1993) describe the major pathways for nitrogen as follows:

- nitrogen fixation - molecular nitrogen (N_2) gas is transformed to organic nitrogen by the activity of aerobic and anaerobic bacteria in the presence of the enzyme nitrogenase;
- mineralization or ammonification - under both aerobic and anaerobic conditions organically combined nitrogen is biologically transformed to ammonium nitrogen (NH_4^+) during organic matter decomposition;
- nitrification - under aerobic conditions, bacteria oxidize ammonium (NH_4^+) to nitrate (NO_3^-);
- denitrification - usually under anaerobic conditions nitrate (NO_3^-) is first transformed by microorganisms to nitrite (NO_2^-), then to gaseous nitrous oxide (N_2O) and molecular nitrogen (N_2) gas.

Another potentially important pathway for gaseous loss of nitrogen from some wetlands is ammonia volatilization (Mitsch and Gosselink, 1993). This process, in which ammonium (NH_4^+) is transformed to ammonia gas (NH_3), is only significant when the pH of surface sediments exceed 7, with nearly 40% converted to NH_3 at pH 9 (Hassett and Banwart, 1992). Losses by ammonia volatilization also increase with temperature and are therefore more likely to occur in the summer (Murphy and Brownlee, 1981).

Lee et al. (1975), Hammer and Knight (1994), Van Oostrom and Russell (1994), and Reddy and Burgoon (1996) all state that denitrification is the major mechanism for nitrogen loss from wetlands, with N_2 the dominant product. Denitrification usually occurs under anaerobic conditions, when facultative anaerobic bacteria use nitrate in place of free oxygen as the terminal electron acceptor in respiration products. Lee et al. (1975) and Knowles and Lean (1987) found that significant denitrification only occurs in the presence of low or zero oxygen concentrations. Hammer and Knight (1994) suggested that surface flow wetlands may be limited in their denitrification potential due to a lack of anaerobic conditions, especially in younger systems with low internal organic matter storages or during winter when oxygen solubility is higher. In such cases, high nitrate outflow concentrations may be observed in young surface flow wetlands or during winter months. Kadlec (1994) found that denitrification appears to be more efficient in wetlands with lower hydraulic loading rates, which is equivalent to increased retention time.

Hammer and Knight (1994) found that nitrification rates appear to be inhibited when water temperatures fall below 8 to 12°C, with a rapid decline to near zero at temperatures below 6°C, whereas ammonification and denitrification did not appear to be significantly reduced at the lowest temperatures typically encountered in North American treatment wetlands. However, Knowles and

Lean (1987) found evidence of significant nitrification during the winter months under the ice of a temperate lake.

1.4 Wetlands as Nutrient Sinks

Mitsch (1989) defines a chemical sink as a system which retains more nutrients or sediments than it releases over a given period of time, and a chemical source as one which exports more than it takes in. A chemical transformer is defined as a system where the net amount of a chemical entering and leaving is the same, but its form is changed.

Much of the literature on nutrient retention in aquatic systems comes from lake studies. Where wetlands have been studied, most of the research has been carried out on natural wetland systems. Those studies which examine constructed wetlands mainly focus on the treatment of sewage effluent, with very few covering treatment of non-point source runoff from urban areas, even though according to Baker (1993), 65 to 75 percent of the pollutants entering lakes and rivers in the United States can be attributed to non-point source runoff. In selecting relevant background studies, the focus was placed on freshwater wetlands, with additional emphasis on temperate wetlands and cattail marshes, the latter which represents the dominate constructed wetland type.

Table 1 provides a summary of relevant individual research studies. This summary appears to indicate that constructed wetland systems are a more effective mechanism for the mitigation of nutrient loads than natural aquatic systems. However, as previous stated, since most of the constructed wetland research studies represent recently constructed systems, while the natural systems have generally been in operation for long periods of time, this really only provides an indication that recently constructed wetlands have good nutrient mitigation potential in the early years following their completion. It does not tell us about the long-term potential of constructed wetlands for nutrient mitigation.

1.4.1 Natural Aquatic Systems

The following provides additional information on some of the research studies included in Table 1.

- In a one year study (except during ice covered conditions during January and February) of the Horicon and Waunakee marshes in Wisconsin, Lee et al. (1975) found that during the summer the concentration of nitrate and ammonia in Horicon Marsh were generally less than 0.1 mg/L of nitrogen. The low level was thought to be associated with uptake by aquatic plants. During the fall and early winter the concentrations were much higher, approaching a total of approximately 1 mg/L. Organic nitrogen values in the range of 0.5 to 20 mg/L, with the highest concentrations found in early fall of 1968, showed the importance of particulate matter as a mode of transport of nitrogen from the marsh.

Table 1: Summary of Relevant Aquatic System Studies

Project	Reference	Wetland Type	Location	Ice Cover	Total Phosphorus Retention					Total Nitrogen Reduction				
					Spring	Summer	Fall	Winter	Annual	Spring	Summer	Fall	Winter	Annual
Natural Aquatic Systems:														
Horicon & Waunakee Marshes	Lee et al. (1975)	agricultural runoff	Wisconsin (43°N, 89°W)	Yes (2 mo.)	release *	retention	release *		**steady state**		retention	reduction	reduction	
Lake Wingra	Loucks et al. (1977)	open space/ urban runoff	Wisconsin (43°N, 89°W)	not reported	8% retention †	18% retention †	16% retention †	1% †	**8% retention †**					
Apalachicola River Wetland	Elder (1985)		Florida (31°N, 85°W)	No	steady state	steady state	steady state	steady state	**10% retention**					**2% retention**
Lake St. George	Knowles and Lean (1987)		Ontario (44°N, 79°W)	Yes									reduction **	
Central Ontario Beaver Pond	Devito et al. (1993)	shallow pond	Ontario (45°N, 79°W)	Yes	-14% release †	45% retention †	-4% release †	-34% release †	**-11% release †**	-23% reduction †	71% retention †	23% retention †	-8% reduction †	**-5% reduction †**
Constructed Wetland Systems:														
Brillion Marsh Experimental Wetlands	Spangler et al. (1977)	surface flow in natural, sewage	Wisconsin (43°N, 89°W)	not reported	retention	retention	release	release	**25-44% retention ‡**					
Cobalt Artificial Cattail Marsh	Miller (1989)	surface flow, sewage	Ontario (47°N, 80°W)	Yes (5 mo.)	71% retention ‡	56-69% retention ‡	60% retention ‡	-6% release ‡	**46% retention ‡**					
Des Plaines River Wetlands Demonstration Project	Kadlec and Hey (1994)	surface flow, agricultural/urban runoff	Illinois (42°N, 89°W)	not reported	23-99% retention ‡	47-92% retention ‡	47-88% retention ‡	21-68% retention ‡	**69-82% retention ‡**					**-66-91% reduction **‡**
SWAMP Niagara-on-the-Lake Pilot Project	Lemon et al. (1996)	subsurface, sewage	Ontario (43°N, 79°W)	Yes					**poor retention**	<-50% reduction ‡	>65% retention ‡	>65% retention ‡	<-50% reduction ‡	**>50% reduction ‡**

LEGEND:

steady state inflow concentrations approximate outflow concentrations

* orthophosphate flushed from system

** nitrate lost from system by denitrification

† results based on nutrient mass balance calculation

‡ results based on calculation of change in inflow and outflow nutrient concentrations - (for Miller (1989) retention was calculated based on provided inflow and outflow data)

- For Lake Wingra, a shallow (mean depth of 2.4 m), eutrophic lake with the littoral areas representing 31% of the total lake surface area of 140 hectares, (Loucks et al., 1977) found that data from low or no runoff periods indicated significantly higher concentrations of phosphorus and nitrogen in the littoral than in open zones. Based on seasonal nutrient budget calculations, they also determined that there is potential for retention of 83% of the incoming phosphorus in summer. However, due to small summer inflows, summer retention is well below this potential (18%) for the study period. Statistical analysis by of spring and summer turnover times (time to fill or empty a particular nutrient reservoir) of dissolved inorganic nitrogen and phosphorus pools were 10 days and 0.5 days respectively, while turnover times of total nitrogen and phosphorus were 5 to 9 years, and 20 to 28 years, respectively (Loucks et al., 1977). (Loucks et al., 1977) determined that the very long turnover times of total nitrogen and phosphorus relative to those of the individual compartment pools of these nutrients indicate a high degree of recycling of these elements within the system. They also found that preliminary analysis of nitrogen and phosphorus dynamics indicated some support for the hypothesis that phosphorus is limiting in the spring and summer periods.

- In a two year, year round study of a Florida sub-tropical river wetland system Elder (1985) found that the concentrations of total phosphorus and total nitrogen were relatively stable in the system, regardless of season or flow conditions. However, he also found that the nutrient load demonstrated changes in its form, with nitrate and dissolved phosphorus showing a tendency to decrease in summer and increase in winter. As a result, he suggested that it was more appropriate to depict the wetland as a transformer, rather than either a sink or source. Nutrient transformation therefore affect the water chemistry more than the nutrient loads, with the forms of nutrients exported from the wetland thought to be of equal or greater importance than reductions in the total amount of nutrients.

- Through analyses of water samples collected over a 9 year period through ice cover at Lake St. George, Knowles and Lean (1987) found a pattern of progressively decreasing concentrations of ammonium and oxygen, and a simultaneous increase in nitrate concentrations to a maximum of 0.1-0.2 mg/L. They concluded that there was potential for reduction of nitrate to gaseous products through denitrification whenever the oxygen in the water column dropped below 0.3 mg O^2.liter^{-1}. They determined that this condition was found in anoxic water which developed under the ice during winter stratification.

- In a shallow temperate beaver pond in central Ontario, Devito and Dillion (1993) determined that biotic assimilation was thought to be controlling phosphorus and nitrogen retention during the growing season, with anaerobic conditions during winter ice cover associated with increased levels of P and nitrogen in surface water during the winter and spring.

Three composite studies were also found which assessed a range of natural wetlands located in North America and Europe. These studies seem to support the idea that the age of a wetland plays an important role in its effectiveness in nutrient mitigation.

- At twenty different freshwater wetland sites, including bogs, fens, and swamps, Richardson (1985) showed high initial removal rates of phosphorus followed by large exports of phosphorus within a few years. It also showed that wetland types with predominately mineral soils and high amorphous aluminium content, such as swamps, are better phosphorus sinks than peatlands.

- As part of an analyses of data from ten natural wetlands receiving wastewater, Richardson et al. (1985) determined that at low loading rates, wetlands have the capacity to remove a high percentage of the phosphorus applied, but this capacity can decrease substantially within a few years, depending on the wetland type. Generally, as the loading rate is increased, the efficiency of phosphorus removal also declined rapidly. The nitrogen removal efficiency of the wetlands was similar to phosphorus at low loading rates and rapidly declined in efficiency as loading rates increased. It was suggested that this might be due to limits on the rate of denitrification, nitrification, oxygen availability, or ammonia or nitrate diffusion.

- Nichols (1983) examined nine natural North American wetlands receiving wastewater effluent for nutrient retention. He found that during the first 1 or 2 years of application of wastewater, a higher phosphorus retention efficiency can be expected, with 90 to 95% retention at low loading rates and up to 70% retention at high loading rates. This efficiency will not be maintained, but will decrease yearly with continued application of wastewater or fertilizer phosphorus. It is known that if sufficient phosphorus is added, the phosphorus adsorption capacity of a soil can be saturated. Wetland soils can also release some of the phosphorus that it has previously absorbed if the phosphorus concentration in the water in contact with the soil is reduced.

1.4.2 Constructed Wetlands

The following is some additional information related to the four individual research studies included in Table 1.

- In assessing two experimental artificial marsh designs located within Brillion Marsh in Wisconsin, Spangler et al. (1977) found predictable seasonal patterns in phosphorus retention, with accumulation during the growing season and flushing from the wetland after the onset of freezing weather. Phosphorus removal ranged from 31-44% for tertiary sewage treatment and 25% for secondary treatment. Accumulation in the artificial marshes seems to be associated with high rates of biological activity, with discharge occurring sometime after cold weather kills the flora;

- In a continuous one year study of an artificial cattail marsh, in Cobalt, which was being used for sewage treatment, Miller (1989) found that the phosphorus concentrations leaving the marsh during the growing season were low. However, during anoxic periods under winter ice cover (from the 1st week of November to the 2nd week of April) much of the retained phosphorus was lost;

- The Des Plaines River Wetlands Demonstration Project, Illinois is a federally funded project designed to provide some comprehensive, long-term answers to the effectiveness of large-scale constructed wetlands. A three year study was carried out on four completed wetlands representing several marsh designs, located within a watershed receiving 80% agricultural and 20% urban NPS runoff. Kadlec and Hey (1994) reported that the average immobilization of approximately three-quarters of incoming phosphorus from constructed wetlands is attributed to its use in the development of live and dead plant tissue during the growing season, and in particular the summer. They thought that some of this retention may be temporary during the early development years of a newly constructed wetland;

- Using a series of pilot scale wetlands designed as subsurface vertical flow systems with pulse-flooding of tertiary sewage, Lemon et al. (1996) reported poor total phosphorus removal, while the annual reduction of total nitrogen was >50%.

One composite study was also found which assessed a range of constructed wetlands located in North America and Europe.

- Using performance data from fifteen small constructed wetlands being used for municipal wastewater treatment in North America and Europe, Watson et al., (1989) found that phosphorus and nitrogen reduction was variable. Phosphorus removal efficiencies ranged from 0 to 90%, with the highest in subsurface flow systems with substrate material high in clay, iron, aluminium and calcium. Nitrogen removal was similar in surface and subsurface flow systems, with removals up to 79%.

Nichols (1983) found that at low loading rates, wetlands have the capacity to remove much of the phosphorus applied and to continue to do so for many years. However, as the loading rate is increased, the efficiency of phosphorus removal declines rapidly. The amount of phosphorus retained by wetlands under natural conditions are low. Annual phosphorus accumulation in undisturbed organic soils is about 0.1 to 0.2 g/m^2.

According to Nichols (1983), Richardson (1985), and Van Oostrom and Russell (1994) no reduction in nitrogen removal efficiency is expected to occur at a given loading rate with continued application of wastewater. As long as the supply of nitrate is maintained, denitrification should continue at the same rate, unless the organic carbon supply in the wetland becomes exhausted. However, Hammer and Knight (1994) state that total nitrogen removal efficiency may be reduced if any of the limiting factors (aerated zone, anaerobic zone, alkalinity, organic matter, hydraulic and solids residence time) are not available. According to Nichols (1983) nitrogen removal efficiencies are lower if calculated on the basis of total nitrogen, since the major mechanism for removing nitrogen from wetlands seems to be denitrification.

From a water quality point of view, it appears difficult to make generalizations regarding phosphorus retention in constructed wetlands since the literature (Boström et al., 1988; Richardson et al., 1985; Richardson, 1985) indicates high variability among sites and wetland types, depending to a large degree on the chemical characteristics of the wetland substrate, differing wetland morphometry and hydrological conditions, and different historical phosphorus loading rates.

1.4.3 The Importance of Anoxic Conditions

The effectiveness of cold climate wetlands has been shown to be limited by transient anoxic conditions which develop under winter ice cover (Lemon et al., 1996). Nürnberg (1984) defines anoxic conditions as those in which the oxygen concentration in the hypolimnion are <0.5 mg.liter^{-1} for at least 2 weeks, but no longer than 7 months. According to Horne and Goldman (1994) anoxic conditions can also be defined as having a redox potential (Eh) of <100 mV. Redox potential is used to quantify the tendency of the soil to oxidize or reduce substances (Mitsch and Gosselink, 1993).

This is important for phosphorus retention in that according to Mitsch and Gosselink (1993) iron is transformed from ferric (Fe^{3+}) to ferrous (Fe^{2+}) form at about 120 mV, resulting in the release of soluble ferrous iron and phosphate to the water column. Based on an analysis of Great Lakes data compiled by others, Mortimer (1971) concluded that if oxygen concentrations do not fall below 1 or 2 mg O_2/liter in aquatic systems, sediments will not contribute phosphorus to the overlying water. This was also shown by Knowles and Lean (1987) in their work through winter ice cover at Lake St. George. This is also relevant for nitrogen mitigation, in that according to Chamie and Richardson (1978) <-200 mV is considered anaerobic and supports denitrification, the major mechanism for nitrogen transformation out of wetlands.

1.4.4 Relative Retention of Nutrients

According to Schindler (1977) the mean annual concentration of total nitrogen and total phosphorus in lakes subjected to fertilizer additions indicated that the nitrogen content of a lake increases when phosphorus input is increased, even when little or no nitrogen is added with fertilizer. This is because nitrogen can be biologically fixed from atmospheric sources, while phosphorus has no similar external compensation mechanism. The typical N/P ratio at which most aquatic organism use these nutrients is 16:1 (Horne and Goldman, 1994). An excess of nitrogen would result in phosphorus being stoichiometrically limited (proportions in which the elements react with one another). Loucks et al. (1977) found some evidence that phosphorus was limiting in Lake Wingra during the spring and summer periods. The reported N/P ratio for Lake Wingra was 9:1. Schinder (1977) also determined that low N/P ratios in the range of 5:1, favour the growth of nitrogen-fixing blue-green algae which tend to be those that raise concerns from a water quality perspective, while higher N/P ratios, <14:1, are more conducive to the growth of less objectionable forms of algae.

1.4.5 Plant Litter and Nutrient Retention

Most pollutant retention or transformation in constructed wetlands is thought to be carried out by the sediment-litter compartment with the aid of a large population of attached micro-organisms. According to Faulkner and Richardson (1989) in natural wetlands this compartment is considered a major pool for phosphorus, accounting for >95% of this element in the system, with <5% contained in the living plant pool, and little in the water column. Guntenspergen et al. (1989) considered nutrients tied up in litter and eventually sediments, a semi-permanent storage compartment. Davis (1991) stated that organic matter accumulation, plus soil adsorption, appear to be the two major processes controlling long-term nutrient immobilization in wetlands.

Organic matter in wetland soils generally varies between 15 and 75 percent. When a soil has less than 20% to 35% organic matter (on a dry weight basis), it is considered a mineral soil (Mitsch and Gosselink, 1993). Clay-sized particles and organic matter are considered the most chemically reactive materials in soil (Ward and Elliot, 1995). According to Mitsch and Gosselink (1993), organic soils have a greater cation-exchange capacity (sum of exchangeable cation [positive ions] that a soil can hold) than mineral soils which have a cation-exchange capacity that is dominated by the major cations (Ca^{++}, Mg^{++}, K^{+}, and Na^{+}). As organic content increases, both the percentage and amount of exchangeable hydrogen ions increases.

1.4.5.1 Phosphorus

Nichols (1983), Richardson et al. (1985), and Mitsch and Gosselink (1993) found that phosphorus fixation by an organic soil was related to the soil's Fe, Al and Ca content. They concluded that in general organic material has almost no capacity to fix phosphorus, with organic soils low in Al, Fe and Ca having very low phosphorus fixing capacities. Nichols (1983) reported that the amount of phosphorus retained by wetlands under natural conditions are typically about 0.1 to 0.2 $g/m^2.y$. Data collected by Richardson (1985) at 20 different freshwater wetland sites showed that wetland types with predominately mineral soils and high amorphous aluminium content, such as swamps, are better phosphorus sinks than peatlands. According to Puriveth (1980) an increase in the population of microbes inhabiting the plant litter tends to be a major cause of phosphorus increase.

1.4.5.2 Nitrogen

According to Riemer (1984) in sediments with a high organic matter content, high levels of organic nitrogen may be recorded. He also reported that most of the inorganic nitrogen in a typical aquatic system is found in the water, with only a small percentage found in the sediments. The large reservoir of organic nitrogen in the sediments is not in rapid equilibrium with the water and is not readily available for plant uptake. It is only the small amount of inorganic nitrogen in the sediments which acts as an exchangeable reservoir with the water column.

Nitrate is an anion, repelled by the net negative charge in most soils, and therefore is not capable of forming a strong chemical bond with mineral soils. Nitrate is highly mobile, and rapidly assimilated by vegetation. Ward and Elliot (1995) state that there is little change in nitrogen stored in soil organic matter. According to Horne and Goldman (1994), while nitrate moves freely through soils, ammonium and phosphate tend to be adsorbed by clay particles or organic matter.

Nichols (1983) suggested that the disappearance of nitrogen from acid organic soils may be due as much to the chemical breakdown of NO_2^- as to microbial denitrification and that the rate of denitrification is related to the availability of organic matter that can furnish energy for the growth of denitrifying bacteria and serve as an H donor for the denitrification process. He reported, that in mineral soils, denitrification is usually limited by available carbon, while in organic wetland soils, sufficient organic matter is usually available so that rapid rates of denitrification are typically obtained. He also reported that the rate of denitrification is independent of nitrate concentrations over a fairly wide range, with the rate of denitrification remaining constant as nitrate concentrations in the soil decline. When the soils were overlain by a layer of water containing nitrate, the diffusion of nitrate from the water to the soil becomes the limiting factor as the nitrate originally present in the soil was depleted, causing the rate of denitrification to become dependent on the nitrate concentration. Sloey et al. (1978) and Watson et al., (1989) reported that denitrification occurs mainly in the soil and not in the overlying water, even if the water is anaerobic. However, Knowles and Lean (1987) showed that the water column has the potential for denitrification whenever oxygen levels fall below 0.3 mg/L.

Davis (1991) reported that the main effect of water enrichment on nutrient concentrations in aquatic plants was to increase translocation or leaching from dying macrophyte plant tissue, rather than to increase the retention in dead leaf material. Where surface water nutrient concentrations were higher, phosphorus and nitrogen retention rates in 2 year old detritus were higher, but only up to a finite capacity.

1.4.6 Different Types of Litter and Nutrient Retention

While a number of studies have focused on variations in the nutrient content among different live plant species, the nutrient retention capability of different plant parts (e.g. stalks, leaves, flowers, etc.) of growing, dying or decomposing aquatic plants (Gessner, 2000; Kadlec, 1989; Chamie, 1978), or the decomposition rates of plant litter dominated by different species (Kadlec, 1989; Chamie, 1978), very few have focused on whether plant litter dominated by different species displayed variable nutrient retention capabilities.

Several studies that did, include a 2 year litter bag experiment by Verhoeven and Arts (1992) in two Dutch fens, one dominated by *Carex diandra*, the other by *Carex acutiformis*. They found that both total phosphorus and total nitrogen remained approximately constant over the period for *C. acutiformis*, while *C. diandra* only remained constant during the first year, then dropped to 15% of its initial value for total phosphorus and 23% for total nitrogen. They thought that the residence time of nutrients in litter depends on the rate of decomposition. Since the rate of decomposition for *C. diandra* was found to be significantly faster than *C. acutiformis* this might help explain the nutrient retention differences in the two species. A 2 year investigation of macrophyte growth, death and decomposition in the Florida Everglades, found that even though *Typha domingensis* accumulated higher nutrient concentrations in its tissue during growth than *Cladium jamaicense*, the concentration in dead leaf tissue, after mortality were similar for both species (Davis, 1991). In a 2 year study of four wetland communities in the Great Dismal Swamp in southeastern Virginia (37°N, 65°W) dominated by different tree species, Day (1982) found that seasonal nutrient dynamics in decaying litter were similar among all litter types, with phosphorus and nitrogen generally accumulating or remaining the same. In a small, shallow eutrophic lake with a 1.25 ha. fringe reedswamp in Norwich, dominated by *Phragmites communis* and *Typha angustifolia*, Mason and Bryant (1975) found that during the growing season, the nitrogen content of *Phragmites* was higher than in *Typha*, while the phosphorus content of both species was similar. Upon death of both species there was a sharp decline in nutrient content, and after the first month there was no significant trend of increase or decrease for nutrients, irrespective of species.

1.4.7 Relationship Between Nutrient Concentrations in Litter and Water Column.

As part of a study of macrophyte growth, death and decomposition in the Florida Everglades, Davis (1991) found that the amount of phosphorus that leaf litter accumulated during 2 years of decomposition was related to the mean surface water total phosphorus concentrations during the period. He also found that litter nitrogen uptake was not clearly related to surface water nitrogen concentrations.

Work by Puriveth (1980) in a Wisconsin marsh revealed that ammonium and soluble reactive phosphorus in the water column showed significant intercorrelation with nitrogen and phosphorus in plant litter.

The relationship between nutrients in the litter and water column may be influenced by the typical residence time of each nutrient in an aquatic system. According to Welch and Lindell (1992) the residence time for an incoming quantity of nitrogen is longer than that for phosphorus, due to the strong retention capacity of inorganic metal complexes and organic particulate matter.

1.5 Hypotheses and Predictions

1.5.1 Constructed Wetlands as a Nutrient Sink

1.5.1.1 Phosphorus

Hypothesis: Biotic assimilation will cause constructed wetlands in temperate areas to be a sink for phosphorus during the growing season, while anoxic conditions resulting from increased respiration by decomposing bacteria and/or the cut off of atmospheric oxygen by ice cover formation will cause the dissolution of iron phosphate precipitates from the wetland substrate. Constructed wetlands in temperate areas will thus become a phosphorus source during late fall.

Prediction: Inflow concentration of total phosphorus in the water will be higher than outflow concentrations during the spring and summer, and lower in late fall.

1.5.1.2 Nitrogen

Hypothesis: Biotic assimilation will cause constructed wetlands in temperate areas to be sinks for nitrogen during the growing season, while anoxic conditions in late fall will cause increased denitrification. Export of gaseous nitrogen from the water column to the atmosphere will also result in reduced total nitrogen concentrations in the water.

Prediction: Inflow concentration of total nitrogen in the water will be higher than outflow concentrations during all seasons, with the largest reduction in the outflow in late fall.

1.5.1.3 Relative Retention of Nutrients

Hypothesis: Constructed wetlands retain phosphorus better than nitrogen because phosphorus is more limiting than nitrogen.

Prediction: The reduction of water phosphorus concentrations from inflow to outflow during the growing season (spring and summer) will be greater than the reduction in nitrogen.

1.5.2 Plant Litter and Nutrient Retention

1.5.2.1 Phosphorus

Hypothesis: Phosphorus retention in newly constructed wetlands derives primarily from sediment adsorption to clay particles and precipitation with metals such as iron. Adsorption and precipitation will increase as the mineral content of wetland sediments increases.

Alternate Hypothesis: Phosphorus retention in wetland sediments is primarily influenced by microbial assimilation. Microbial assimilation will increase as the organic content of wetland sediments increases.

Prediction: If sediment adsorption and precipitation are the dominant mechanisms for phosphorus retention in constructed wetlands, litter phosphorus concentrations will decrease as the organic content of wetland sediments increases.

Alternate Prediction: If microbial assimilation is the dominant mechanism for phosphorus retention in constructed wetlands, litter phosphorus concentrations should increase as the organic content of wetland sediments increases.

1.6 Other Questions

Several aspects of the study are not concerned with testing particular hypotheses about wetland function, but rather with answering questions of direct relevance to the design and management of constructed treatment wetlands.

1.6.1 Plant Litter and Nitrogen Retention

Since nitrogen does not tend to build-up in wetland sediments, it is very difficult to test the effectiveness of plant litter for nitrogen retention. Therefore an analysis of litter nitrogen will be undertaken primarily to provide a more complete picture of nitrogen cycling within the wetland. Several important aspects to consider are: that nitrogen is a primary organic compound of plant tissue and therefore as the organic content of the sediments increases, so will the nitrogen content; that organic sources of nitrogen are converted to dissolved inorganic nitrogen through mineralization/ammonification, which is the process of decay by bacteria; since the rate of decomposition increases with increasing temperature, litter nitrogen concentrations will likely decrease as water temperatures increase; and inorganic nitrogen is the form which can be readily used by aquatic plants, which is likely to be most available during the height of the growing season, when water temperatures are highest.

1.6.2 Different Types of Litter and Nutrient Retention

The importance of plant litter in nutrient retention is widely recognized. However, from a construction cost perspective, whether different types of litter vary in their ability to retain nutrients is an important question. Previous research suggests that even though different plant species, or different plant parts may retain different amounts of nutrients during the growing season, once the initial leaching phase of dying plant material has been completed, the resulting plant litter is equally effective at nutrient retention, irrespective of its species composition. An assessment is therefore made as to whether three different types of litter (mixed, cattail, leaf) show any significant differences in their ability to retain total phosphorus and nitrogen.

1.6.3 Relationship Between Nutrient Concentrations in Litter and Water Column.

1.6.3.1 Phosphorus

Phosphorus is strongly attracted to mineral substrates, which can become saturated by phosphorus adsorption. If attachment sites on the substrate become available, phosphorus from the water can be expected to exploit any excess adsorption capacity, creating a dynamic relationship between phosphorus in the substrate and phosphorus in the water column. Therefore, a negative relationship may be found between total litter phosphorus and water phosphorus concentrations. Although the substrate compartment in this two year old constructed wetland is not likely to be saturated, it is important to track the possible progress of phosphorus saturation and water - sediment dynamics.

1.6.3.2 Nitrogen

Since the large reservoir of organic nitrogen in the sediments is not in rapid equilibrium with the water, only the small amount of inorganic nitrogen in the sediments acts as an exchangeable reservoir with the water column. Therefore, a relationship between total litter and water nitrogen concentrations is not likely to be found.

1.6.4 Iron as an Indicator of Phosphorus Release from Sediments

Iron-phosphorus compounds in sediments are a significant source of phosphorus to the water column under anaerobic conditions. As such, it was the intention of this research project to investigate this relationship in an effort to better inform the cycling of phosphorus in the Monahan Drain constructed wetland. However, budget limitations only permitted laboratory analyses of water total iron concentrations. The lack of data for litter total iron concentrations severely limits the ability to analyze this important relationship. Nevertheless, statistical analysis of the relationship between water iron and phosphorus concentrations will be undertaken to provide some preliminary insight into this potentially important pathway.

Chapter 2 – MATERIALS AND METHODS

2.1 Study Site

The Monahan Constructed Wetland is a stormwater management facility constructed by the City of Kanata (45°N, 75°W). The facility was designed to control the quality and quantity of stormwater runoff resulting from a new growth area in the municipality. The stormwater management facility is a free water surface flow constructed wetland consisting of two cells in series with a surface area of approximately 4.2 hectares (Fig. 1). The treatment facility also includes five sediment forebays and three storage cells. The total drainage area of the Monahan Drain upstream of the constructed wetland is approximately 637 hectares (J.L. Richards & Associates, 1993).

Phase 1 of the Monahan Drain Constructed Wetland was initiated in August 1994. Planting of a broad range of aquatic and upland species was completed in September 1996. The facility was considered to be in full operation in mid-May of 1997, with the installation of permanent and portable sampling equipment for compliance monitoring. By the summer of 1998, the littoral zone of the wetland was dominated by *Typha latifolia*, with *Scirpus* species and *Sagittaria latifolia* present, but much less dominant. In deeper zones, mainly in Cell 1, *Potamogeton pectinatus* and other submerged macrophyte species dominated. Upland areas surrounding the wetland were almost entirely composed of tall grasses and meadow species.

Pre-construction, water quality results (Gore & Storrie Limited, 1992) of the Monahan Drain agricultural ditch, derived from sampling data collected by the Rideau Valley Conservation Authority in 1991, indicates that values as high as 1.50 mg/L TKN and 0.2 mg/L TP were recorded.

Based on post-construction contour plans of the constructed wetland, the maximum water depth within the wetland is 2 metres. Approximately 18% of the area of Cell 1 and 4% of Cell 2 are deeper than 1.5 metres, while approximately 24% of the area of Cell 1 and 29% of Cell 2 are less than 0.5 metres deep (littoral). During the two year study period, Cell 1 displayed more extensive growth of submerged aquatic plants, while Cell 2 was subject to more turbulence from wave action.

Fig. 2 and 3 summarize mean monthly local temperature and rainfall conditions during the study period, as recorded by Environment Canada at the Macdonald-Cartier Airport weather station, Ottawa. This provides some indication of the different climatic conditions to which the wetland was exposed over the two year period. 1997 experienced higher mean spring precipitation than 1998, while 1998 received much greater mean summer rainfall than 1997. Mean monthly temperatures were generally higher in 1998 than 1997, except for June and July.

2.2 Field Procedures

Water and litter samples were collected at approximately monthly intervals over a two year period. In 1997 samples were collected from mid-May to mid-November, while in 1998 samples were collected from mid-April to mid-October.

Due in part to the young age of the wetland, several different types of litter were included in the study. Litter types generally falls into two categories; naturally occurring litter resulting from the growth and senescence of plants in and around the study wetland, and an off-site source of leaf litter imported and installed within the wetland.

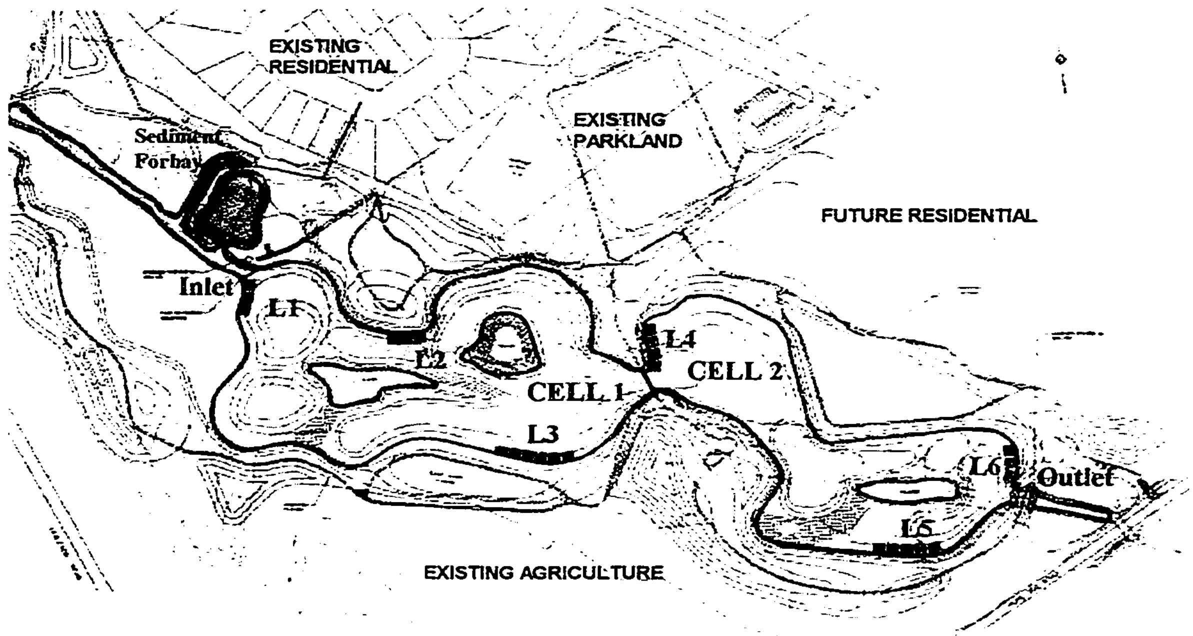

Fig. 1: Study Site - Monahan Drain Constructed Wetland, May 1997 to October 1998.

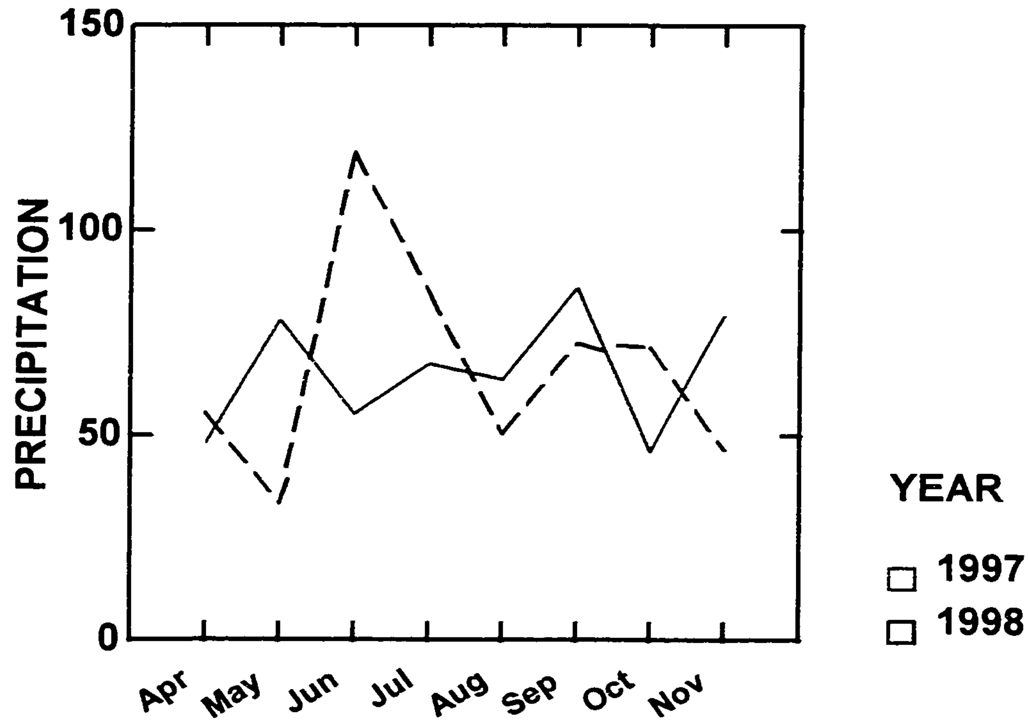

Fig. 2: Mean monthly precipitation (mm)

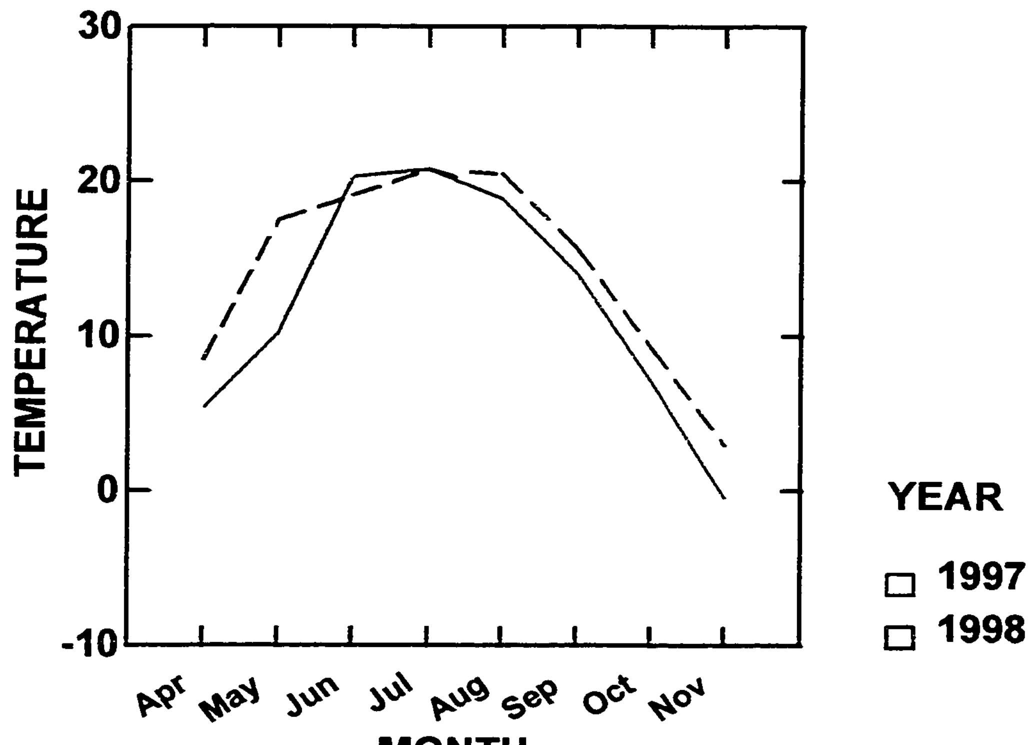

Fig. 3: Mean monthly air temperatures (°C)

2.2.1 Natural Litter

Six sampling stations (inlet, outlet, and four locations in between) were selected in the littoral zone around the outer perimeter of the two wetland cells (Fig. 1). At each station several litter samples (3 replicates in 1997 and 2 replicates in 1998) of approximately 50 g were collected and placed immediately in sealable polyethylene freezer bags.

In 1997, because the build-up of natural litter was just beginning, half of the total replicates taken consisted of pieces of deadfall cattail litter plucked from beneath the water surface of the wetland. The remaining litter samples consisted of mixed decomposing litter scooped from the bottom of the wetland with a net, fabricated from a double layer of 1.5 mm fiberglass mesh. Some effort was made to sift out excessive mineral substrate material from the collected litter samples.

In 1998 the natural accumulation of litter was found to provide adequate amounts of organic matter for sampling purposes. Therefore in the second year, all of the litter samples consisted of mixed decomposing litter collected from the bottom of the wetland.

All natural litter samples where taken from littoral areas within approximately 2 metres of the shoreline, and at a maximum depth of approximately 0.5 metres.

2.2.2 Imported Litter

Two stations were established with litter bags filled with an off-site source of fallen hardwood leaf litter, composed predominantly of red maple (*Acer rubrum*) and largetooth aspen (*Populus grandidentata*) leaves. The litter bags were placed in two large nylon mesh bags which were then anchored in two open water locations near the wetland inlet in Cell 1.

Litter bags, 15 x 15 cm in size, were fabricated from 1.5 mm fiberglass mesh and contained approximately 50 g of leaf litter per bag. The total number of litter bags allowed for a minimum of 5 replicate litter bags to be collected from each station at each sampling interval over the two year period. However, after several months in 1997, the litter bags were lost due to severance of the anchor lines. Whether severing of the lines was due to human or animal actions is unknown, but the bags could not be retrieved. As a result, five replicate leaf litter samples were collected from each of the two open water stations in June, July and August (at one station only). The May sample represents a baseline condition for the off-site source of litter.

2.2.3 Water and Temperature

Immediately prior to the collection of each litter sample, a grab water sample was also collected from just below the water surface at each of the stations. At the open water stations where five leaf litter samples were collected, only three water samples were collected at each sampling interval.

The water temperature at the time of collection of each water sample was also recorded.

2.3 Laboratory procedures

2.3.1 Water Preparation and Analyses

Water samples were collected for nutrients and metals in separate, single use 250 ml PET bottles as supplied by Systems Plus (New Hamburg, Ontario). Immediately following field collection, water samples coded for metal analyses were preserved in a solution of approximately 0.3 to 0.5% nitric acid. All water samples were then refrigerated.

Collected water samples were analyzed for concentrations of total Kjeldahl nitrogen (TKN), total phosphorus (TP), total iron (Fe) and total calcium (Ca). Analyses of TKN and TP consisted of digestion using an Hg catalyst, followed by automated colorimetric analysis using a Skalar autoanalyser (phenate/hypochlorite for TKN and molybdate/ascorbic acid for TP) based on MOE (1983). Ca was analysed by inductively coupled plasma (ICP) - Varian Liberty 110. For Fe analyses, nitric acid evaporation was followed by ICP using an ultrasonic nebuliser for sample introduction.

Detection limits for TP, TKN, Fe, and Ca in water samples were 0.005, 0.01, 0.0002, and 0.01mg/L, respectively.

2.3.2 Litter Preparation and Analyses

Immediately following field collection, all litter samples were frozen until they could be prepared for analysis. Preparation consisted of each litter sample being drained of excess water by placing it on a mesh sieve. Animal organisms visible with the naked eye were then picked out of or scraped off of each litter sample. Each sample was then placed in a paper bag and dried in an oven at 80°C for a minimum of 4 days. The dried samples were then homogenized to a fine consistency by pulverizing with a mortar and pedestal and/or chopping with an electric grinder. The homogenized samples were then sifted through a 2 millimeter sieve to remove large stones and an 250 micron sieve to remove fine clay sediments.

Collected litter samples were analyzed for concentrations of total Kjeldahl nitrogen (TKN) and total phosphorus (TP). Litter samples were analysed using wet digestion with sulfuric acid on a hot plate followed by a hydrogen peroxide (30%) digestion for 5 minutes. This step was repeated until the solution became clear. Digests were analyzed colorimetrically for TP and TKN as for water samples, above.

Detection limits for TP and TKN in litter samples were both 0.01 (% by weight).

A loss on ignition test was also conducted on litter samples to determine the percent organic matter for each prepared sample. This was determined from the ash content resulting from combustion of each sample at 550°C for a minimum of 4 hours in a muffle furnace.

2.4 Data Analyses/Statistics

While nutrient retention in natural aquatic systems is often determined by mass balance calculations (Loucks et al., 1977; Devito et al., 1993), nutrient retention in constructed wetland systems is commonly calculated as the change in concentration between the inflow and outflow (Spangler et al., 1977; Kadlec and Hey, 1994; and Lemon et al., 1996). A positive value indicates nutrient retention, a negative value nutrient release or reduction. Ideally the outflow concentration should account for the time required for the inflowing water to move the length of the wetland (retention time). In practical application, frequent sampling at the inflow and outflow are usually averaged in making this calculation.

In assessing whether the Monahan Drain constructed wetland acts as a sink for nutrients, data collected from locations near the inflow (station L1) and outflow (station L6) were used. On each day of sampling there was typically a 6 to 9 hour time interval between sample collection at these two stations, with sampling occurring at approximately monthly intervals. While a more frequent sampling interval would have been more desirable, budgetary constraints and the intent to capture a full range of seasonal variations, resulted in a more widely spread time interval. However, a monthly sampling scale was used in most of the relevant aquatic studies (Lee et al., 1975; Puriveth, 1980; Elder, 1985; Miller, 1989 and Davis, 1991), with some studies using a seasonal scale (Spangler et al., 1977; and Kadlec and Hey, 1994).

Since nutrient retention in cold climate wetlands appears to be strongly influenced by season, analyses of the data has been broken down into three seasons: spring represented by mid-April and mid-May; summer by mid-June, mid-July, mid-August and mid-September; and fall by mid-October and mid-November.

Chapter 3 - RESULTS

3.1 Constructed Wetlands as a Nutrient Sink

3.1.1 Phosphorus

Total phosphorus concentrations in the water column ranged from 0.026 to 0.673 mg/L in 1997 and from 0.020 to 0.099 mg/L in 1998.

Using data for all sampling stations, a two-way non-parametric factorial ANOVA with log10 total water phosphorus concentrations as the dependent variable, and season and year as factors shows a significant effect for year (H_{year}=20.943, df=1, p=0.000), and significant interaction between season and year ($H_{season*year}$=10.920, df=2, p= 0.004). Fig. 4 shows average monthly total phosphorus concentrations in the water column pooled over all sampling stations in the wetland. It shows that total water phosphorus concentrations were higher in 1997 as compared to 1998.

To assess nutrient retention, a 3-way Model III non-parametric ANOVA was carried out with log10 total water phosphorous concentrations as the dependent variable and station (L1 or L6), year (1997 or 1998) and season (spring, summer, fall) as the factors. This analysis indicates a significant 3-way interaction between station, year and season ($H_{station*year*season}$=9.557, df=2, p=0.008).

As a result, a comparison was made of L1 and L6 concentrations within a given (season, year) class. These results (Table 2) show that the wetland was a significant sink for phosphorus in the spring and summer of 1997, and a significant source in late fall of 1997. By contrast, in 1998, there was no significant difference in water phosphorus concentrations at L1 and L6 in the spring, but there was evidence for a sink during both summer and early fall. It is worth noting that the estimate of retention for spring and fall 1998 are almost identical. The lack of statistical significance in the former is due to the increased variability among replicates and the corresponding reduction in statistical power.

More detailed examination of spatial variation in water phosphorus concentrations within the wetland shows that the elevated values at station L6 in late fall of 1997 also occur at the other two sampling stations (L4 and L5) in Cell 2 (Fig. 5). Thus, all stations in Cell 2 in 1997 showed dramatically elevated water phosphorus concentrations. In contrast, no such pattern was observed in 1998. The very high negative retention value calculated for fall 1997 (Table 2) to a large degree reflects this dramatic difference between the two cells.

3.1.2 Nitrogen

Total nitrogen concentrations in the water column ranged from 0.04 to 1.91 mg/L in 1997 and from 0.14 to 1.11 mg/L in 1998.

Using data for all sampling stations, a two-way non-parametric factorial ANOVA with log10 total water nitrogen concentrations as the dependent variable, and season and year as factors shows a significant effect for season (H_{season}=24.253, df=2, p= 0.000), and significant interaction between season and year ($H_{season*year}$=36.770, df=2, p= 0.000). Fig. 6 shows average monthly total nitrogen concentrations in the water column pooled over all sampling stations in the wetland. It shows that total water nitrogen concentrations were higher in 1997 than 1998, except for mid-November 1997, when they were much lower.

An analysis similar to that described above for water phosphorus, comparing L1 and L6 water nitrogen concentrations within a given (season, year) class shows the wetland was a significant sink in the spring and late fall of 1997, and the summer and early fall of 1998 (Table 3). There was no significant difference in water nitrogen concentrations at L1 and L6 in the summer of 1997 and the spring of 1998.

More detailed examination of spatial variation in water nitrogen concentrations within the wetland shows that the depressed values at station L6 in late fall of 1997 also occur at the other two sampling stations (L4 and L5) located within Cell 2 (Fig. 7). In this case, all stations in Cell 2 showed dramatically reduced water nitrogen concentrations. In contrast, no such pattern was observed in 1998. Once again, the very high positive reduction value calculated for fall 1997 (Table 3) is reflective of this dramatic difference between the two cells.

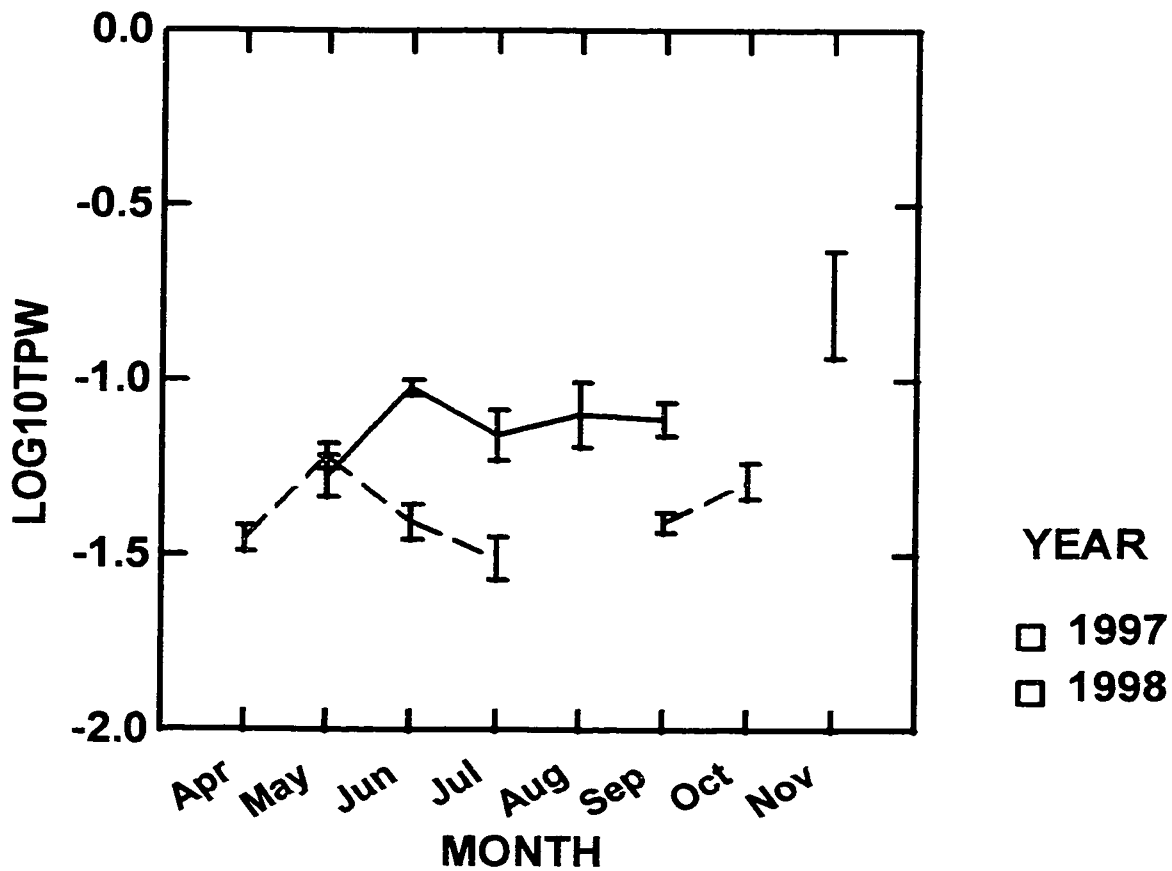

Fig. 4: Average monthly log10 total phosphorus concentrations, mg/L (+/-se) in water for 1997 (n=108) and 1998 (n=72), pooled over all sampling stations in the wetland.

Parameter	1997		
	Spring	Summer	Fall
Station L1, (N)	0.087±.011 (3)	0.164±.064 (12)	0.032±.001 (3)
Station L6, (N)	0.041±.003 (3)	0.071±.009 (12)	0.630±.074 (3)
Mann-Whitney U, (p)	9 (0.025)	105 (0.029)	9 (0.025)
% Retention (*R*)	52.9	56.7	-1868.8

Table 2a

Parameter	1998		
	Spring	Summer	Fall
Station L1, (N)	0.061±.019 (4)	0.058±.007 (6)	0.052±.007 (2)
Station L6, (N)	0.048±.010 (4)	0.030±.002 (6)	0.041±.002 (2)
Mann-Whitney U, (p)	8.5 (0.443)	36 (0.002)	4 (0.061)
% Retention (*R*)	21.3	48.3	21.2

Table 2b

Mean ± 1 standard error (sample size) total phosphorus concentrations (mg/L) in water near inflow (station L1) and outflow (station L6) during 1997 (Table 2a) and 1998 (Table 2b), and the Mann-Whitney U and probability (p) associated with a test of the one-tailed null hypothesis that the wetland is a sink during the spring and summer, and a source during the fall. Percent retention (*R*) is calculated $R=[(\bar{c}_I-\bar{c}_O)/\bar{c}_I]\times 100$, where $\bar{c}_I,\bar{c}_O$ are the average L1 and L6 concentrations respectively. A positive value indicates a sink, a negative value a source.

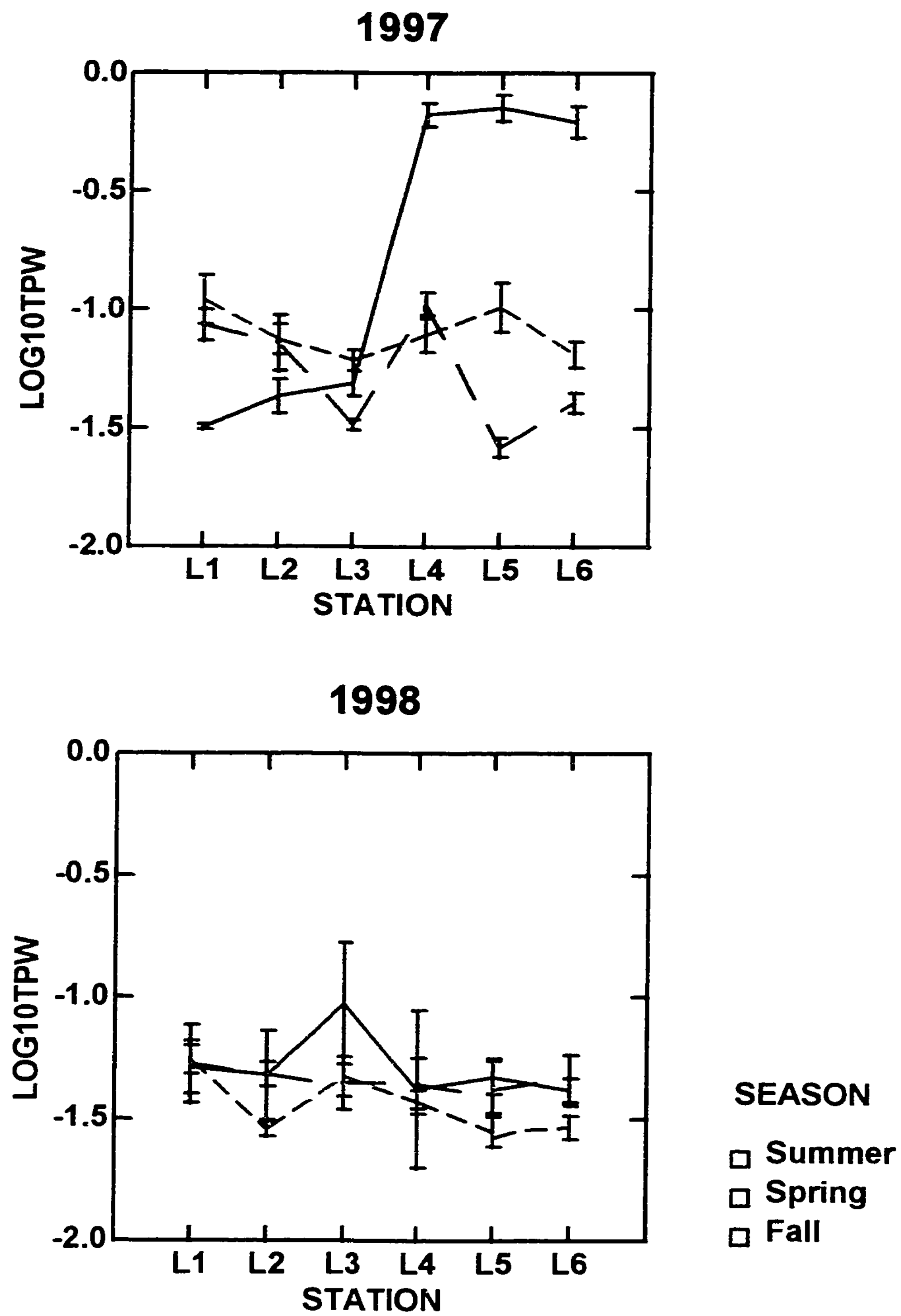

Fig. 5: Average seasonal log10 total phosphorus concentrations, mg/L (+/-se) in water at each sampling station in 1997 and 1998.

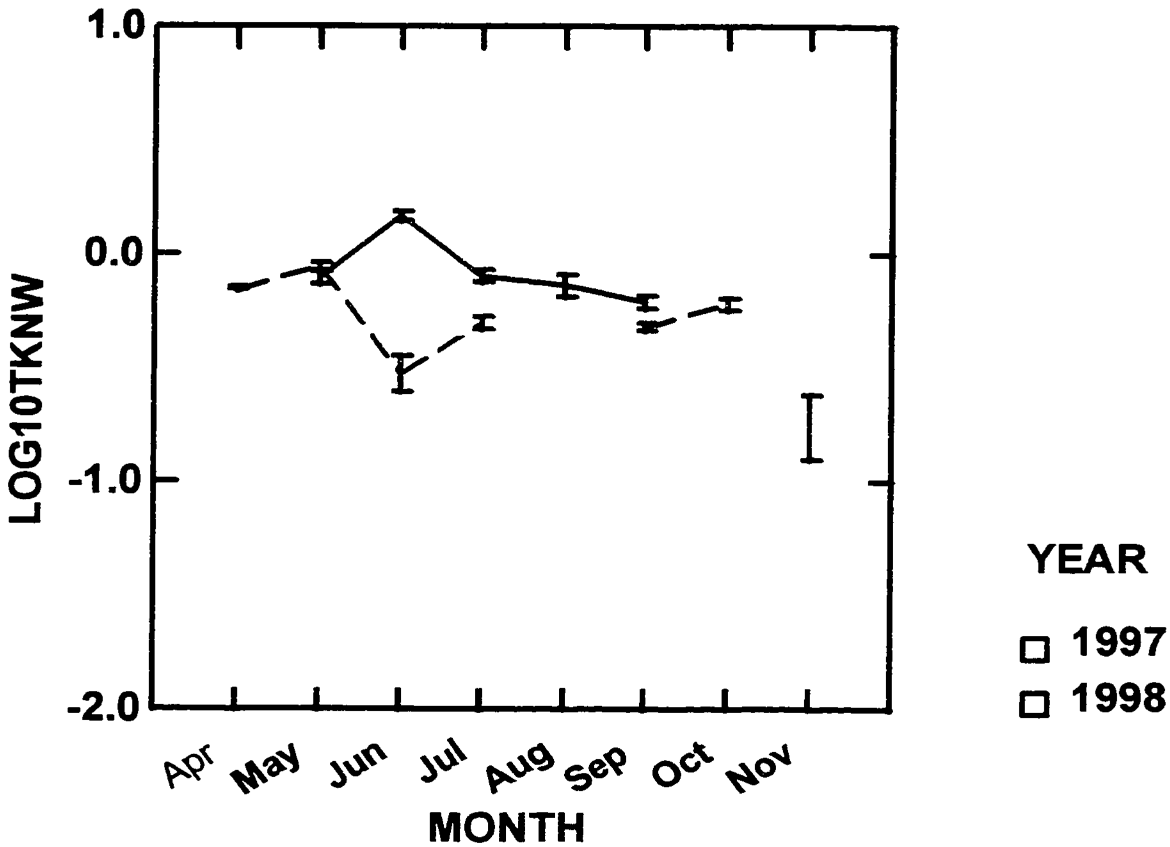

Fig. 6: Average monthly log10 total nitrogen concentrations, mg/L (+/-se) in water for 1997 (N=108) and 1998 (n=72), pooled over all sampling stations within the wetland.

Parameter	1997		
	Spring	**Summer**	**Fall**
Station L1, (N)	0.913±.059 (3)	0.822±.085 (12)	0.567±.032 (3)
Station L6, (N)	0.663±.024 (3)	1.006±.155 (12)	0.037±.005 (3)
Mann-Whitney U, (p)	9 (0.025)	64.5 (0.333)	9 (0.025)
% Retention (*R*)	27.4	-22.4	93.5

Table 3a

Parameter	1998		
	Spring	**Summer**	**Fall**
Station L1, (N)	0.897±.139 (4)	0.563±.067 (6)	0.695±.075 (2)
Station L6, (N)	0.727±.020 (4)	0.433±.077 (6)	0.500±.020 (2)
Mann-Whitney U, (p)	10 (0.281)	28 (0.055)	4 (0.061)
% Retention (*R*)	19.0	23.1	28.1

Table 3b

Mean ± 1 standard error (sample size) total nitrogen concentrations (mg/L) in water near inflow (station L1) and outflow (station L6) during 1997 (Table 3a) and 1998 (Table 3b), and Mann-Whitney U and probability (p) associated with a test of the one-tailed null hypothesis that the wetland is a sink during the spring and summer, and a major sink during the fall. Percent reduction (*R*) is calculated $R = [(\bar{c}_I - \bar{c}_O)/\bar{c}_I] \times 100$, where $\bar{c}_I, \bar{c}_O$ are the average L1 and L6 concentrations respectively. A positive value indicates a reduction, while a negative value indicates an increase.

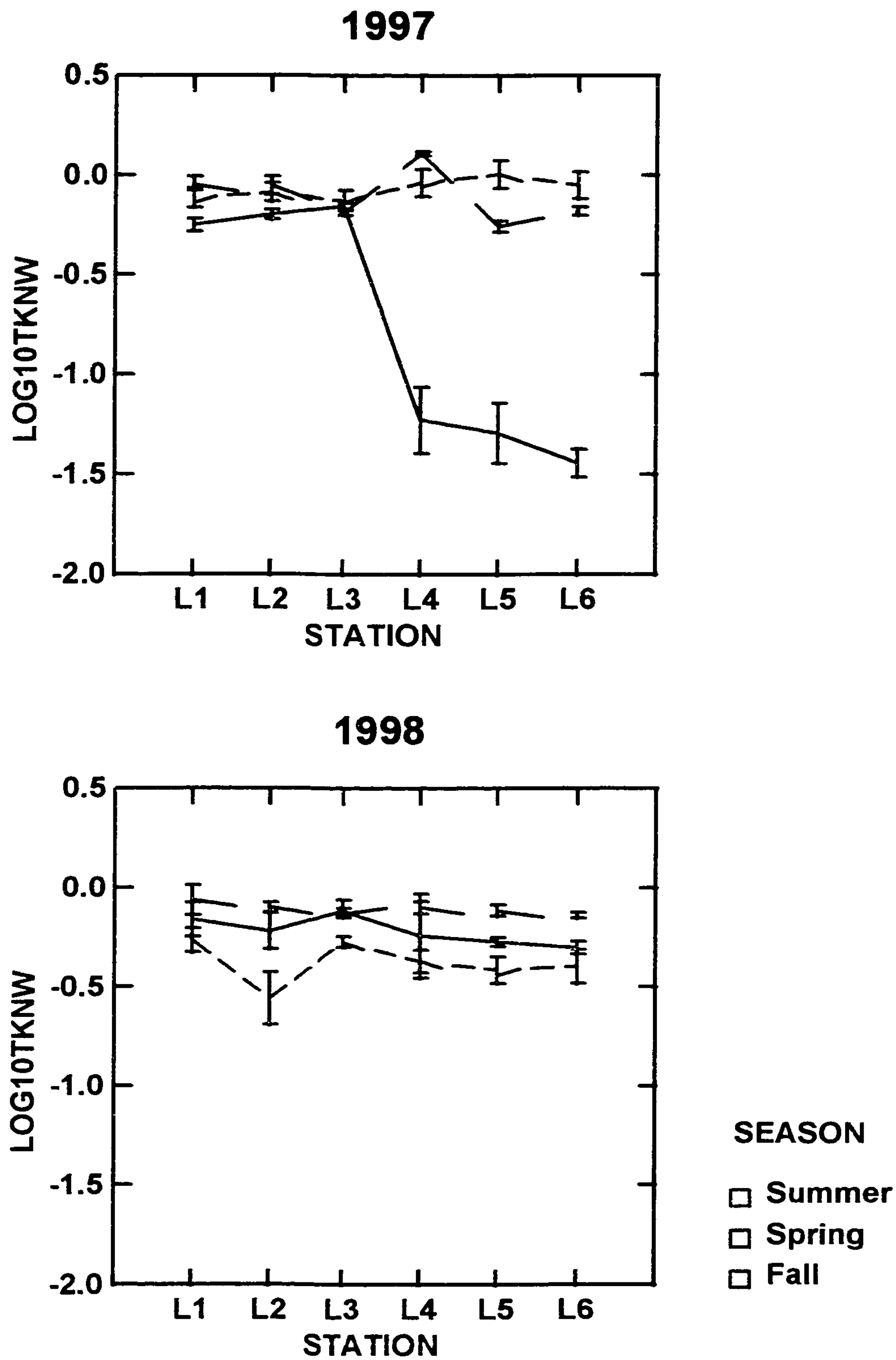

Fig. 7: Average seasonal log10 total nitrogen concentrations, mg/L (+/-se) in water at each sampling station in 1997 and 1998.

3.2 Relative Retention of Nutrients

Fig. 8 shows the average log10 water nutrient (total nitrogen and total phosphorus) concentrations at the inflow station (L1) and outflow station (L6) during the growing season (spring and summer) of each sampling year.

A two-way non-parametric factorial ANOVA with log10 total water concentrations of phosphorus and nitrogen as the dependent variables, and station (L1 or L6) and nutrient (phosphorus or nitrogen) as the factors shows no significant interaction between station and nutrients during the spring and summer of both 1997 ($H_{station*nutrients}$=0.996, df=1, p=0.318), and 1998 ($H_{station*nutrients}$=0.188, df=1, p= 0.665). Hence, there is no statistical evidence for greater phosphorus than nitrogen retention (as indicated by the reduction in water nutrient concentrations between L1 and L6). However, the corresponding plots (Fig. 8) indicate a considerably larger effect size for phosphorus in 1997 (as evidenced by the difference between the L1-L6 slopes) whose detection is precluded by the low statistical power associated with the small sample size.

Table 4 provides seasonal N to P ratios in the water column over the two year study period. These results show that the N to P ratio decreases from spring to fall of each year, and increases from 1997 to 1998.

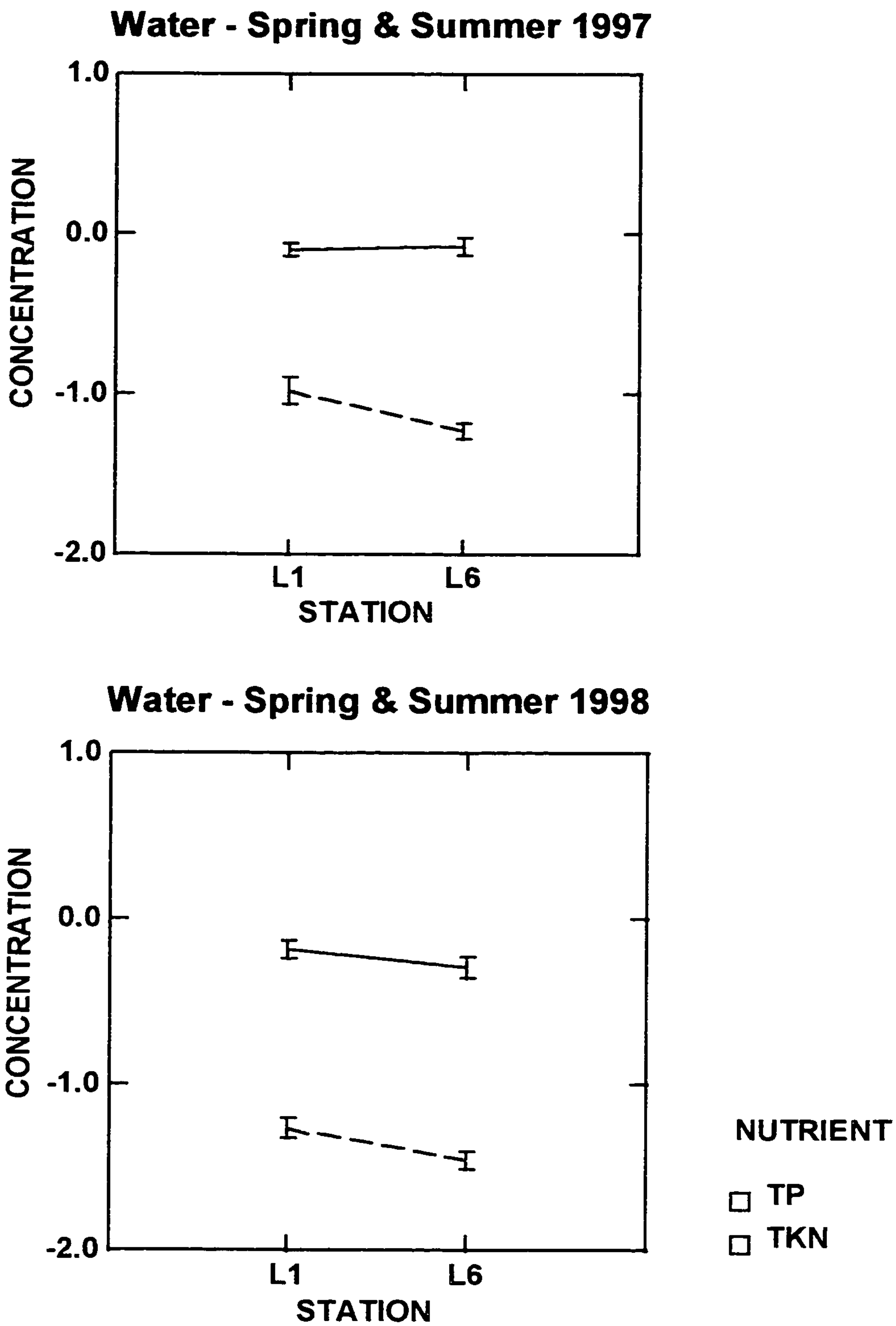

Fig. 8: Average log10 nutrient concentrations, mg/L (+/-se) in water at stations L1 and L6, during the spring and summer seasons of 1997 and 1998.

Year	Season	N/P Ratio
1997	Spring	13:1
1997	Summer	9:1
1997	Fall	1:1
1998	Spring	16:1
1998	Summer	11.5:1
1998	Fall	11:1

Table 4: Water N to P ratios based on mean seasonal and yearly concentrations, mg/L.

3.3 Plant Litter and Nutrient Retention

3.3.1 Phosphorus

Total phosphorus concentrations in litter ranged from 0.11 to 0.23 percent dry weight in 1997 and from 0.01 to 0.22 percent dry weight in 1998.

Using data for all sampling stations, a two-way non-parametric factorial ANOVA with log10 mixed and cattail litter phosphorus concentrations as the dependent variable, and season and year as factors shows a significant effect for both season (H_{season}=15.786, df=2, p= 0.000) and year (H_{year}=21.937, df=1, p= 0.000), and weak significant interaction between season and year ($H_{season*year}$=7.039, df=2, p= 0.030). Fig. 9 shows average monthly total phosphorus concentrations in mixed and cattail litter, pooled over all sampling stations in the wetland. A similar pattern results when only mixed litter is considered. Fig. 9 shows that total litter phosphorus concentrations were higher in 1997 than 1998, and lower in the summer, especially during 1998.

An analysis (Table 5) for total litter phosphorus, comparing L1 and L6 concentrations within a given (season, year) class, shows consistently smaller concentrations at the outflow station (L6) than at the inflow station (L1), although this difference is only statistically significant during the spring of each year.

As discussed previously, because the build up of natural litter was just beginning in this newly constructed wetland, half of the total samples collected in 1997 consisted of pieces of deadfall cattail litter plucked from beneath the water surface of the wetland. The remaining litter samples consisted of mixed decomposing litter scooped from the bottom sediments. In 1998, the natural accumulation of litter was found to provide adequate amounts of organic matter for sampling purposes. Therefore in the second year, all of the litter samples consisted of mixed decomposing litter.

Fig. 10 shows the relationship between log10 mixed litter phosphorus concentrations and percent organic matter over the two year study period.

Linear regressions (Table 6) of log10 mixed litter phosphorus concentrations and transformed percent organic matter (by weight) within a given year and season show only a significant positive relationship during the summer of 1998. For the summer of 1998, these results are consistent with the alternate prediction that microbial assimilation is an important mechanism for phosphorus retention in wetland sediments.

Fig. 11 shows the relationship between log10 cattail litter phosphorus concentrations and percent organic matter during the first year of the study, which was the only year this type of litter was collected.

Linear regressions (Table 7) of total cattail litter phosphorus concentrations and transformed percent organic matter (by weight) within a given season shows a significant negative relationship during the summer of 1997. For the summer of 1997 these results are consistent with the main prediction, that sediment adsorption and precipitation are the dominant mechanisms for phosphorus retention in cattail litter.

Linear regressions of total leaf litter phosphorus concentrations and transformed percent organic matter (by weight) do not show a significant relationship for either of the two seasons (spring and summer) for which samples of this type of litter were collected in 1997.

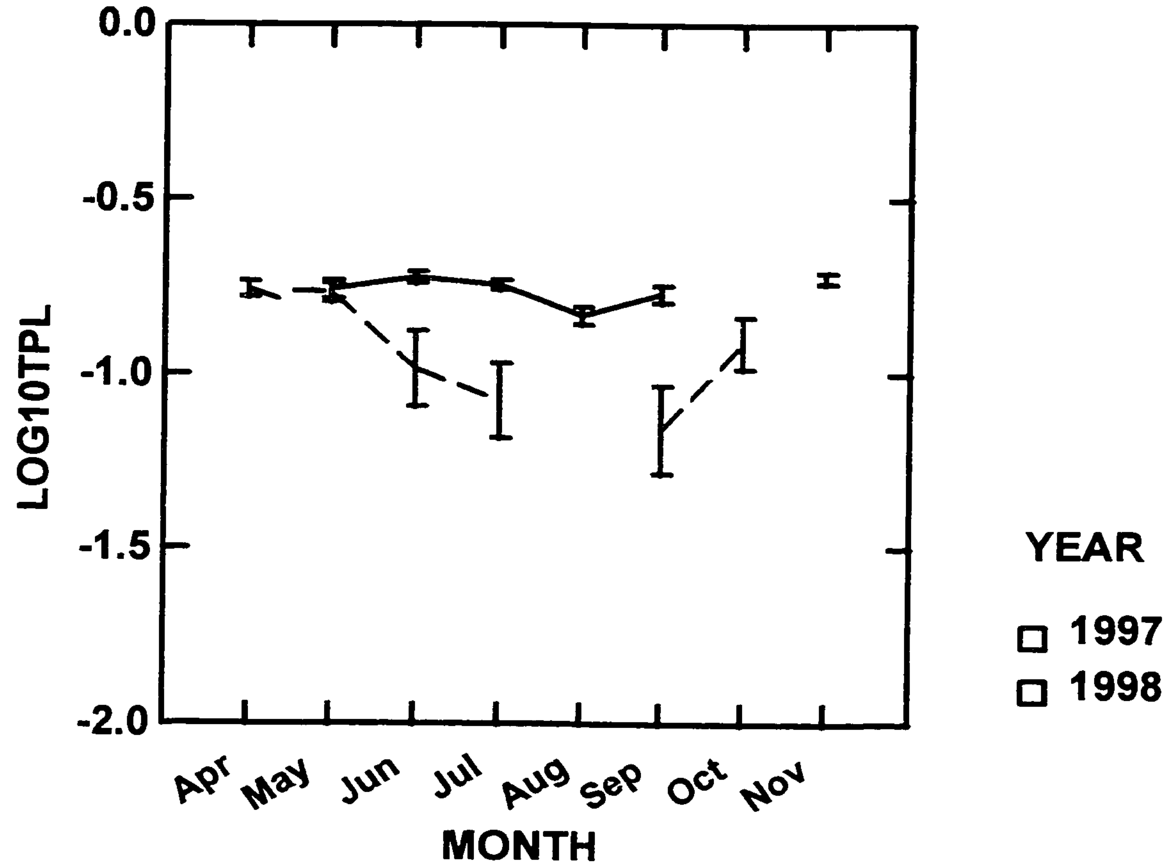

Fig. 9: Average monthly log10 total phosphorus concentrations, % dry weight (+/- se) in mixed and cattail litter for 1997 (n=108) and 1998 (n=72), pooled over all sampling stations within the wetland.

Parameter	1997		
	Spring	Summer	Fall
Station L1, (N)	0.220±.000 (3)	0.188±.012 (12)	0.200±.021 (3)
Station L6, (N)	0.160±.040 (3)	0.177±.011 (12)	0.170±.015 (3)
Mann-Whitney U, (p)	9 (0.034)	79 (0.684)	6.5 (0.376)
% Retention (*R*)	27.3	5.9	15.0

Table 5a

Parameter	1998		
	Spring	Summer	Fall
Station L1, (N)	0.203±.010 (4)	0.115±.032 (6)	0.175±.005 (2)
Station L6, (N)	0.160±.004 (4)	0.082±.027 (6)	0.080±.010 (2)
Mann-Whitney U, (p)	16 (0.019)	23 (0.413)	4 (0.121)
% Retention (*R*)	21.2	28.7	54.3

Table 5b

Mean ± 1 standard error (sample size) total phosphorus concentrations (% dry weight) in mixed and cattail litter near inflow (station L1) and outflow (station L6) during 1997 (Table 5a) and 1998 (Table 5b), and Mann-Whitney U and probability (p) associated with a test of the two-tailed null hypothesis that average concentrations are the same at the inflow and outflow stations. Percent change (*R*) is calculated $R=[(\overline{c}_I-\overline{c}_O)/\overline{c}_I]\times 100$, where $\overline{c}_I, \overline{c}_O$ are the average L1 and L6 concentrations respectively. A positive value indicates a reduction, while a negative value indicates an increase.

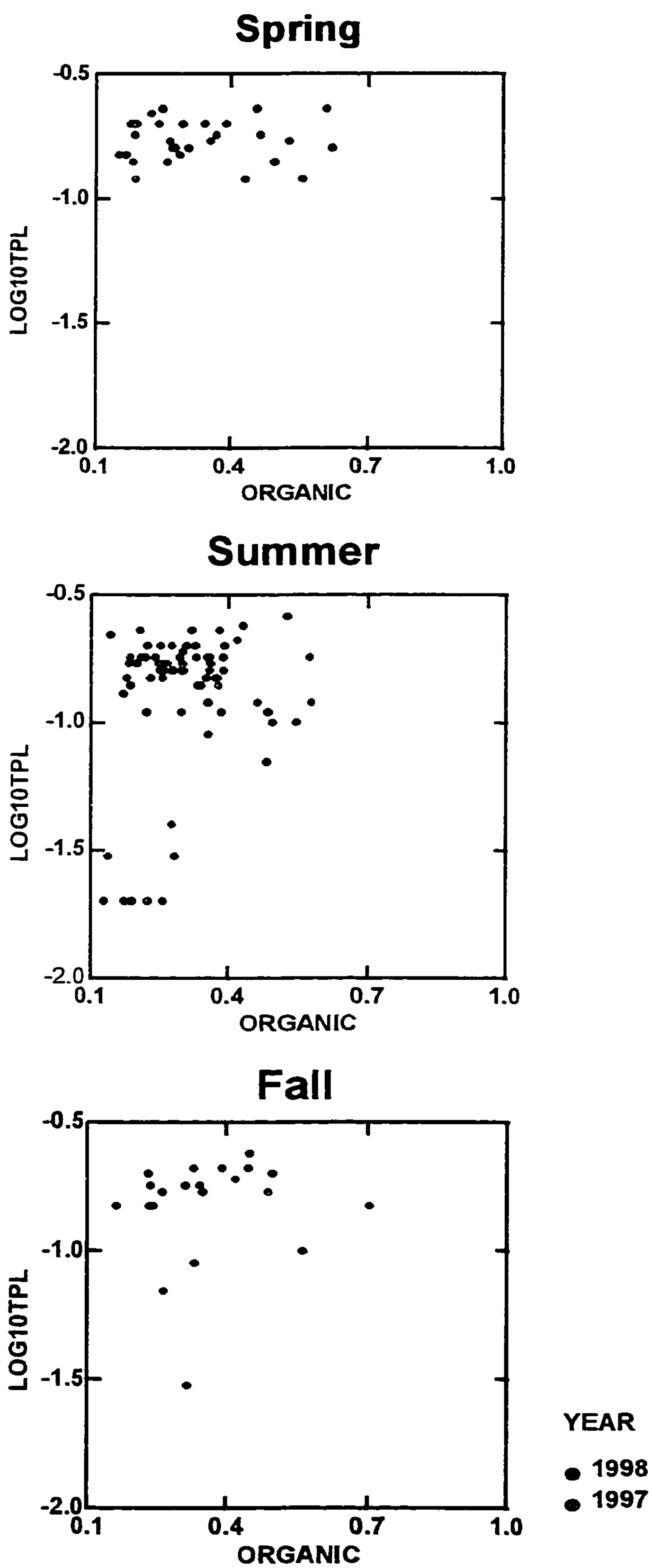

Fig. 10: The relationship between log10 total phosphorus concentrations, % by weight, and percent organic matter (by weight) in mixed litter during the two year study period.

Year	Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	Spring	-0.219	0.945	0.187	9	0.002	0.164	-1.172	0.280
1997	Summer	0.366	1.240	0.242	36	0.013	0.063	1.515	0.139
1997	Fall	1.143	1.663	0.726	9	0.015	0.262	1.575	0.159
1998	Spring	0.048	-0.806	0.127	24	0.007	0.006	0.375	0.711
1998	Summer	-1.838	0.753	0.508	36	0.125	0.278	-3.617	0.001
1998	Fall	0.470	-1.376	0.709	12	0.061	0.042	0.663	0.522

Table 6: Regression of log10 total phosphorus concentrations (% dry weight) on arcsine square root transformed percent organic matter in mixed litter within a given season. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

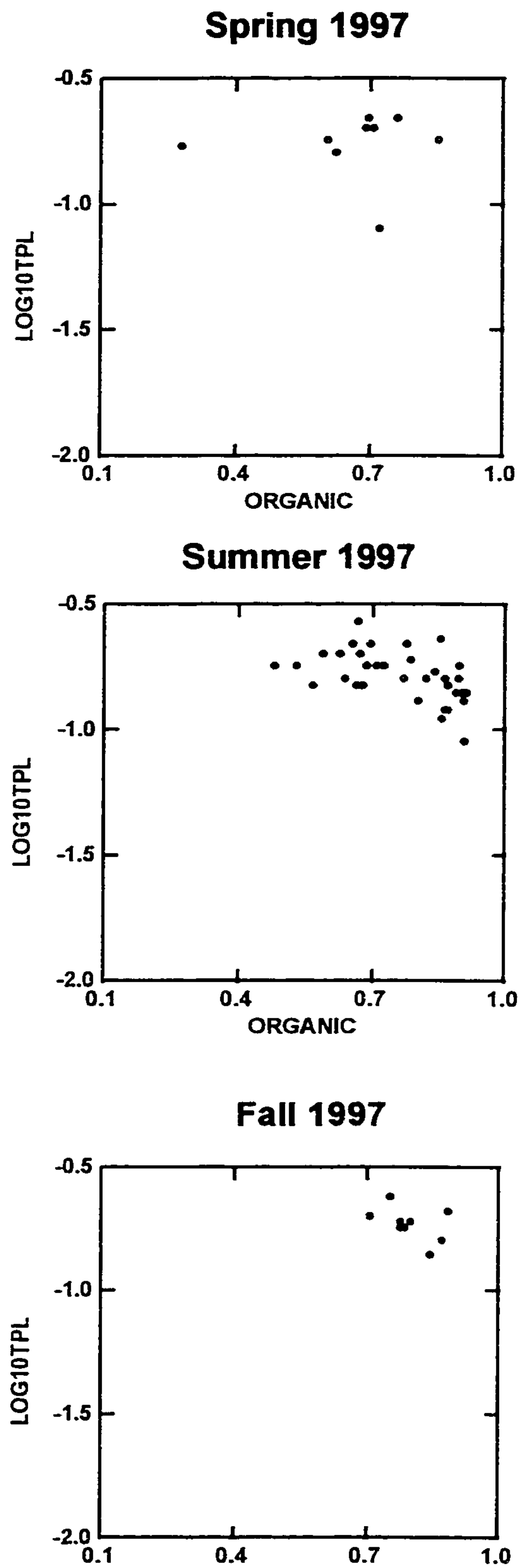

Fig. 11: The relationship between log10 total phosphorus concentrations, % dry weight, and percent organic matter (by weight) in cattail litter during 1997.

Year	Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	Spring	-0.041	0.587	0.481	9	0.033	0.001	-0.085	0.934
1997	Summer	0.745	1.079	0.209	36	0.016	0.272	3.562	0.001
1997	Fall	0.464	0.801	0.384	9	0.005	0.172	1.207	0.266

Table 7: Regression of log10 total phosphorus concentrations (% dry weight) on arcsine square root transformed percent organic matter in cattail litter within a given season. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

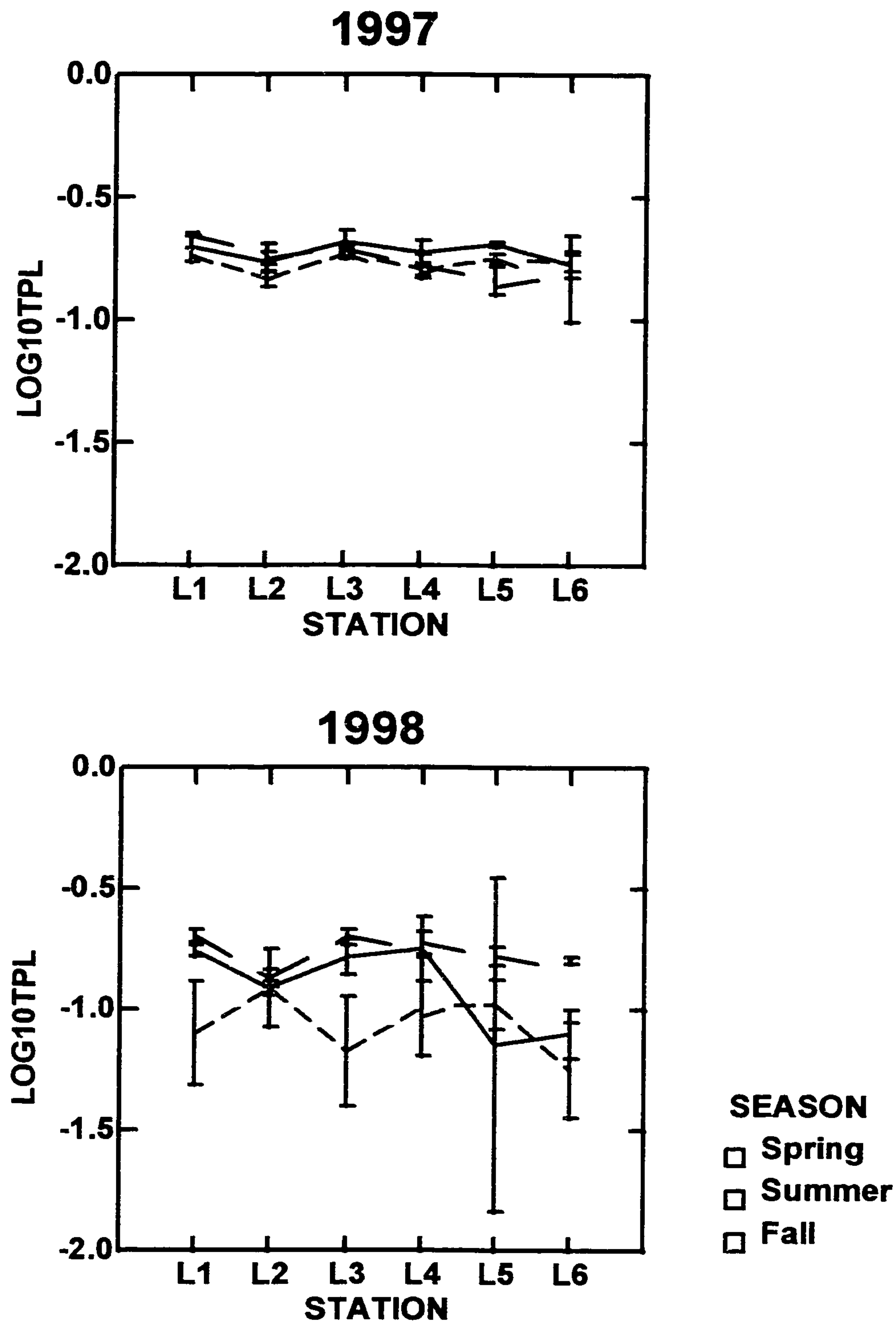

Fig. 12: Average seasonal log10 total phosphorus concentrations. % dry weight (+/-se) in mixed and cattail litter at each sampling station in 1997 and 1998.

More detailed examination of spatial variations in mixed and cattail litter phosphorus concentrations within the wetland (Fig. 12) shows a great deal of uniformity among stations and seasons during 1997, followed by a much greater degree of variability among both stations and seasons in 1998. A 3-way Model III non-parametric ANOVA with log10 total litter phosphorous concentrations as the dependent variable and station (L1 to L6), year (1997 or 1998) and season (spring, summer, fall) as the factors, shows a weak significant 2-way interaction between year and season ($H_{year*season}$=7.025, df=2, p=0.030).

3.3.2 Nitrogen

Total nitrogen concentrations in litter ranged from 1.08 to 3.35 percent dry weight in 1997 and from 0.84 to 3.58 percent dry weight in 1998.

Using data for all sampling stations, a two-way non-parametric factorial ANOVA with log10 mixed and cattail litter nitrogen concentrations as the dependent variable, and season and year as factors only shows a significant effect for season (H_{season}=18.301, df=2, p= 0.000). Fig. 13 shows average monthly total nitrogen concentrations in mixed and cattail litter, pooled over all sampling stations in the wetland. A similar pattern results when only mixed litter is considered. Fig. 13 shows that total litter nitrogen concentrations similar for 1997 than 1998.

An analysis (Table 8) comparing L1 and L6 concentrations within a given (season, year) class, shows consistently smaller concentrations at the outflow station (L6) than at the inflow station (L1), although this difference is not statistically significant in any season or year.

Fig. 14 shows the relationship between log10 mixed litter nitrogen concentrations and percent organic matter during the two year study period.

Linear regressions (Table 9) of total mixed litter nitrogen concentrations and transformed percent organic matter (by weight) within a given season shows a significant positive relationship during spring and summer of both 1997 and 1998, while no relationship is indicated for the fall of either year.

Fig. 15 shows the relationship between log10 cattail litter nitrogen concentrations and percent organic matter (by weight) during the first year of the study, which was the only year this type of litter was collected. Linear regressions (Table 10) of this relationship using transformed percent organic matter within a given season only shows a significant positive relationship during the spring of 1997. By comparison, linear regressions of total leaf litter nitrogen concentrations and transformed percent organic matter (by weight) do not show a significant relationship for any season during the 1997 study period.

Fig. 16 shows the relationship between log10 mixed and cattail litter nitrogen concentrations and water temperatures during the two year study period. This figure shows a general decrease in the total nitrogen of the litter as water temperature increases. A similar set of graphs is observed when only mixed litter is used. A linear regressions (Table 11) of total litter nitrogen concentrations and water temperatures within a given year shows a significant negative relationship for both years.

This provides some indication that decomposition and subsequent plant uptake are significant mechanisms for nitrogen loss from the litter compartment.

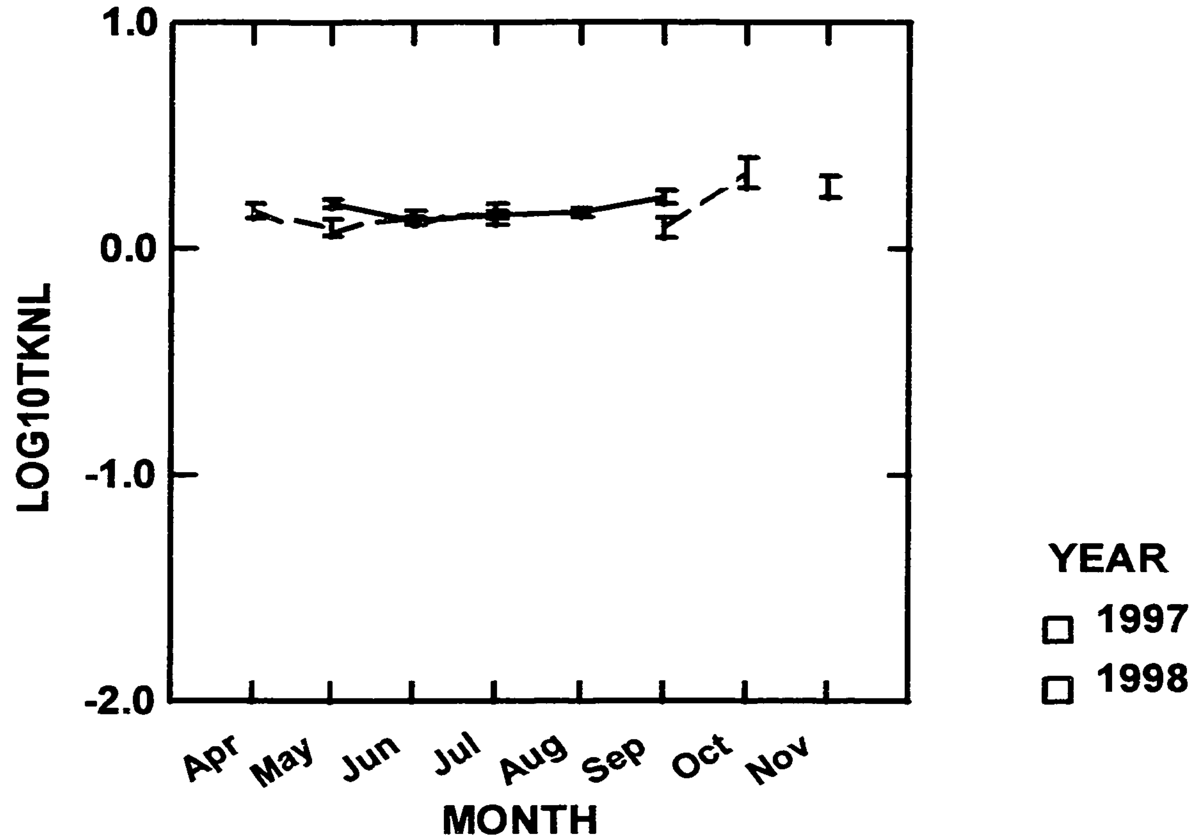

Fig. 13: Average monthly log10 total nitrogen concentrations, % dry weight (+/-se) in mixed and cattail litter for 1997 (n=108) and 1998 (n=72), pooled over all sampling stations within the wetland.

Parameter	1997		
	Spring	Summer	Fall
Station L1, (N)	1.750±.234 (3)	1.470±.062 (12)	2.397±.672 (3)
Station L6, (N)	1.410±.036 (3)	1.497±.057 (12)	1.543±.105 (3)
Mann-Whitney U, (p)	8 (0.127)	64 (0.644)	8 (0.127)
% Retention (*R*)	19.4	-1.8	35.6

Table 8a

Parameter	1998		
	Spring	Summer	Fall
Station L1, (N)	1.482±.241 (4)	1.317±.079 (6)	1.775±.135 (2)
Station L6, (N)	1.132±.042 (4)	1.500±.112 (6)	1.245±.285 (2)
Mann-Whitney U, (p)	12 (0.248)	8 (0.109)	4 (0.121)
% Retention (*R*)	23.6	-13.9	29.9

Table 8b

Mean ± 1 standard error (sample size) total nitrogen concentrations ((% dry weight) in mixed and cattail litter near inflow (station L1) and outflow (station L6) during 1997 (Table 8a) and 1998 (Table 8b), and Mann-Whitney U and probability (p) associated with a test of the two-tailed null hypothesis that average concentrations are the same at the inflow and outflow stations. Percent change (*R*) is calculated $R=[(\overline{c}_I-\overline{c}_O)/\overline{c}_I]\times 100$, where $\overline{c}_I,\overline{c}_O$ are the average L1 and L6 concentrations respectively. A positive value indicates a reduction, while a negative value indicates an increase.

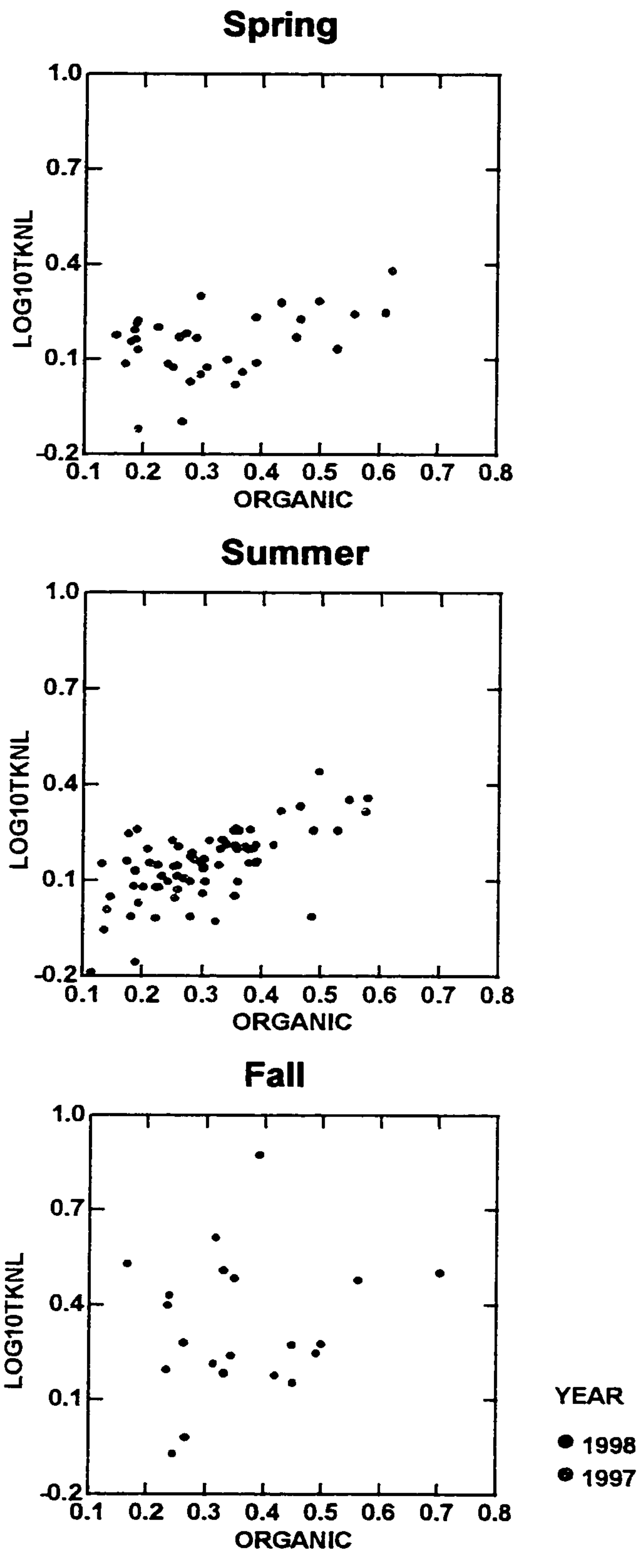

Fig. 14: The relationship between log10 total nitrogen concentrations, % dry weight, and organic matter, % dry weight, in mixed litter during the two year study period.

Year	Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	Spring	-0.770	1.261	0.237	9	0.001	0.601	-3.249	0.014
1997	Summer	-1.098	1.153	0.139	36	0.005	0.647	-7.898	0.000
1997	Fall	-0.122	0.886	0.214	9	0.019	0.045	-0.572	0.585
1998	Spring	-0.633	0.717	0.128	24	0.007	0.525	-4.929	0.000
1998	Summer	-0.617	0.758	0.164	36	0.013	0.294	-3.762	0.001
1998	Fall	-0.404	0.738	0.651	12	0.051	0.037	-0.620	0.549

Table 9: Regression of log10 total nitrogen concentrations (% dry weight) on arcsine square root transformed percent organic matter (by weight) in mixed litter within a given season. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

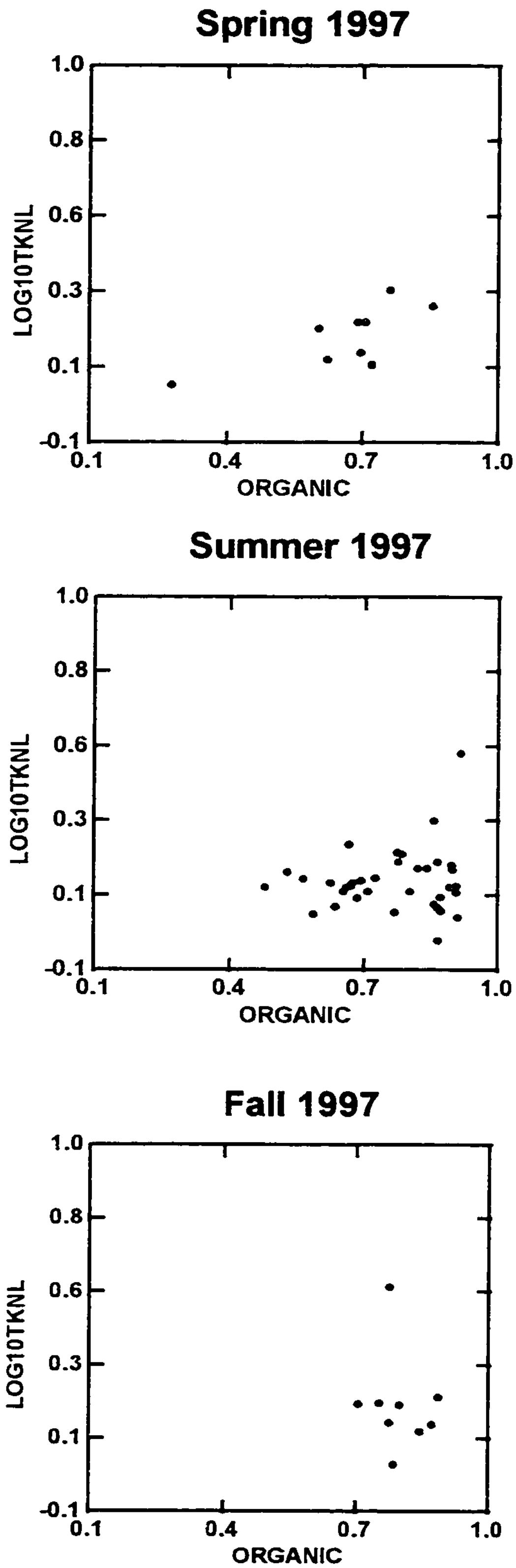

Fig. 15: The relationship between log10 total nitrogen concentrations and percent organic matter in cattail litter during the first year of the study.

Year	Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	Spring	-0.377	0.441	0.136	9	0.004	0.524	-2.773	0.028
1997	Summer	-0.087	0.205	0.109	36	0.009	0.018	-0.793	0.433
1997	Fall	0.310	0.078	0.724	9	0.024	0.026	0.428	0.681

Table 10: Regression of log10 total nitrogen concentrations (% dry weight) on arcsine square root transformed percent organic matter (by weight) in cattail litter within a given season in 1997. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

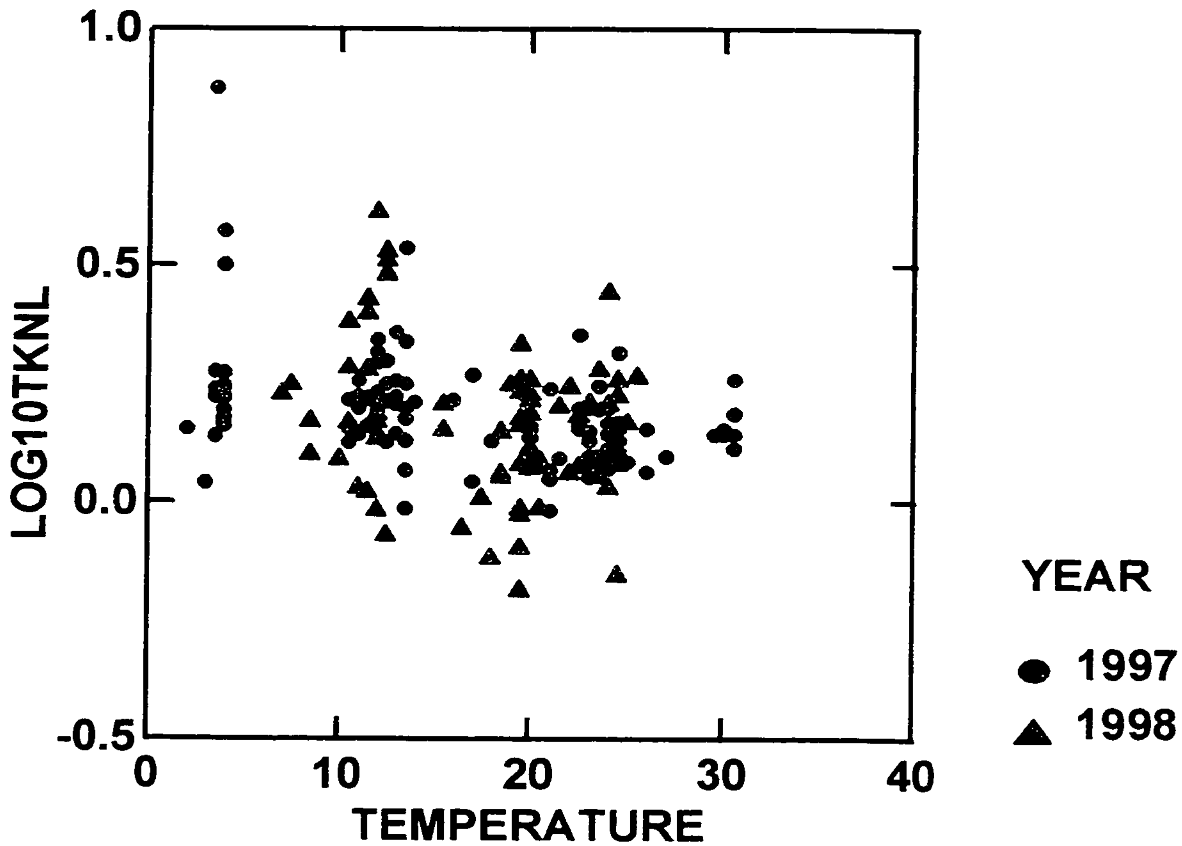

Fig. 16: Relationship between log10 total nitrogen concentrations (% dry weight) in litter (mixed and cattail) and water temperature (°C) at time of sampling at each station for each study year.

Year	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	-0.005	0.281	0.001	108	0.012	0.149	-4.313	0.000
1998	-0.008	0.307	0.004	72	0.025	0.149	-2.209	0.030

Table 11: Regression of log10 total nitrogen concentrations (% dry weight) in mixed and cattail litter and water temperature (°C) at time of sampling at each station within a given year. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

More detailed examination of spatial variations in mixed and cattail litter nitrogen concentrations within the wetland (Fig. 17) shows a fair amount of uniformity among seasons in 1997. A greater degree of variability among stations and seasons appears to occur in 1998. A 3-way Model III non-parametric ANOVA with log10 total litter nitrogen concentrations as the dependent variable and station (L1 to L6), year (1997 or 1998) and season (spring, summer, fall) as the factors, shows a significant 2-way interaction between station and year ($H_{station*year}$=24.513, df=10, p=0.006).

3.4 Different Types of Plant Litter and Nutrient Retention

As outlined in Section 2.2, above, three different types of plant litter were used: an off-site source of fallen hardwood leaf litter; deadfall cattail litter collected from within the water column; and mixed decomposing litter which had accumulated in the wetland. Table 12 provides a summary of the percent (dry weight) organic content of the three litter types. Mixed litter consists of samples collected during both 1997 and 1998. The other litter types were only collected during 1997.

Figure 18 clearly indicates that mixed litter contains a much lower organic content than leaf or cattail litter. Mixed litter was collected from the bottom sediments of the wetland. While an effort was made to sift out excessive mineral fractions from the collected material, many samples still contained considerable amounts of mineral sediments. A one-way non-parametric factorial ANOVA with transformed percent organic matter as the dependent variable, and litter type as the factor shows considerable variation among litter types (H_{type}=93.907, df=2, p= 0.000).

A 2-way Model III non-parametric ANOVA with log10 total litter phosphorous concentrations as the dependent variable and season (spring and summer) and litter type (cattail, leaf and mixed) as the factors, shows a significant result (Fig. 19a) for litter type (H_{type}=24.667, df=2, p=0.000), with leaf < cattail </= mixed litter, however no significant results were found for season (H_{season}=1.500, df=1, p=0.221), nor season•type ($H_{season*type}$=1.547, df=2, p=0.453). A similar analysis for log10 total litter nitrogen concentrations show no significant difference among litter types or season. Leaf samples were not collected during the fall of 1997, nor in 1998.

A 2-way Model III non-parametric ANOVA with log10 total litter phosphorous concentrations as the dependent variable and seasons (spring, summer and fall) and litter type (cattail and mixed) as factors, yields no significant results for season (H_{season}=3.625, df=2, p=0.163), litter type (H_{type}=875, df=1, p=0.350), or season•type ($H_{season*type}$=0.500, df=2, p=0.779). However, a similar analysis for total litter nitrogen concentrations as the dependent variable and seasons (spring, summer and fall) and litter type (cattail and mixed) as factors, yields a significant result for season (H_{season}=12.286, df=2, p=0.002), however no significant results were found for litter type (H_{type}=1.429, df=1, p=0.232), nor season•type ($H_{season*type}$=2.929, df=2, p=0.231). Fig. 19b which depicts the relationship between log10 total nitrogen concentrations in mixed litter and season, only shows a significant seasonal result in the fall of 1997. No significant seasonal differences were found for total nitrogen concentrations in cattail litter.

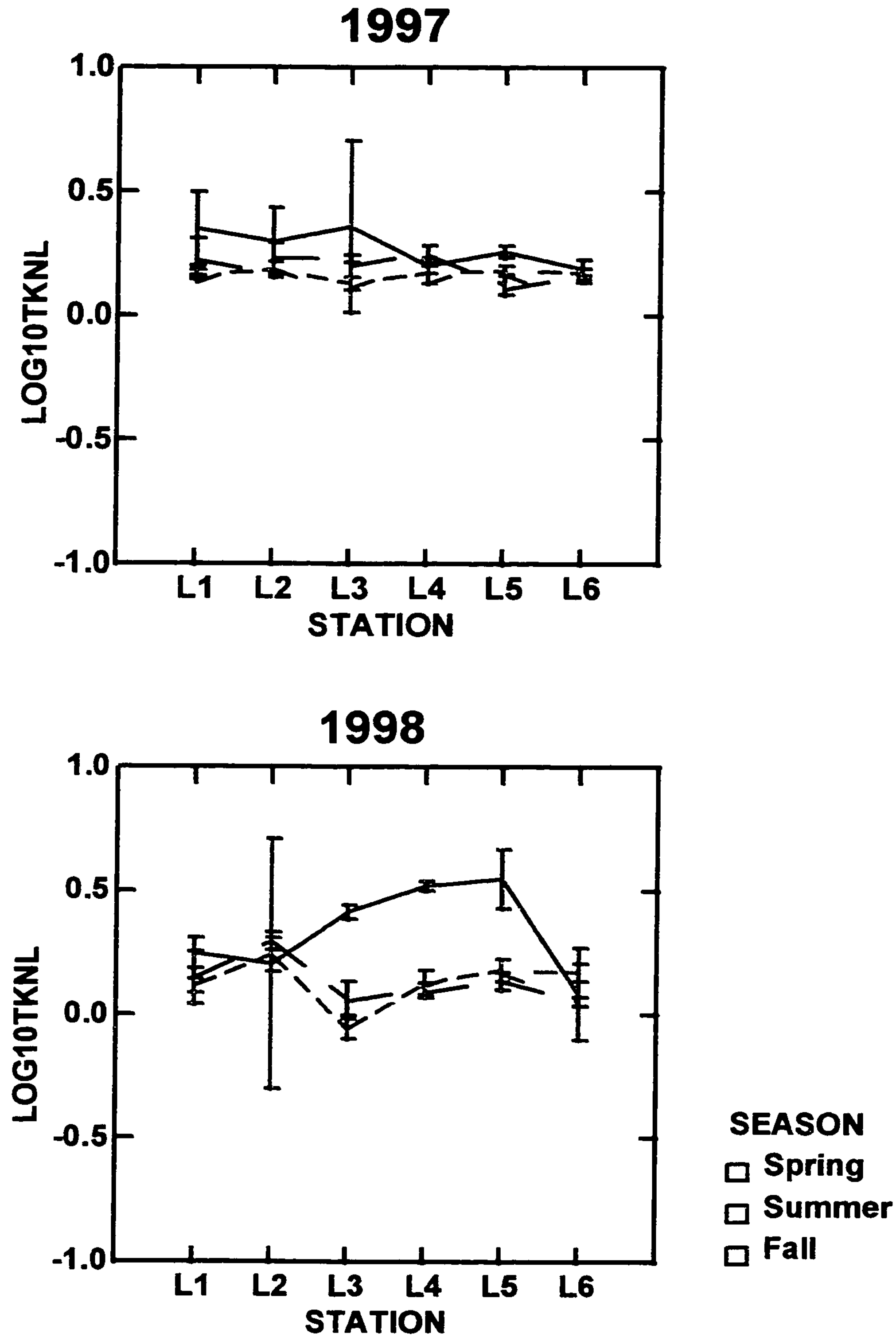

Fig. 17: Average seasonal log10 total nitrogen concentrations, % dry weight (+/-se) in mixed and cattail litter at each sampling station in 1997 and 1998.

Type	Mean	Min.	Max.	N
leaf	80.6	58.1	88.8	18
cattail	75.3	27.9	91.6	54
mixed	31.8	11.4	70.2	126

Table 12: Percent (dry weight) organic content by litter type.

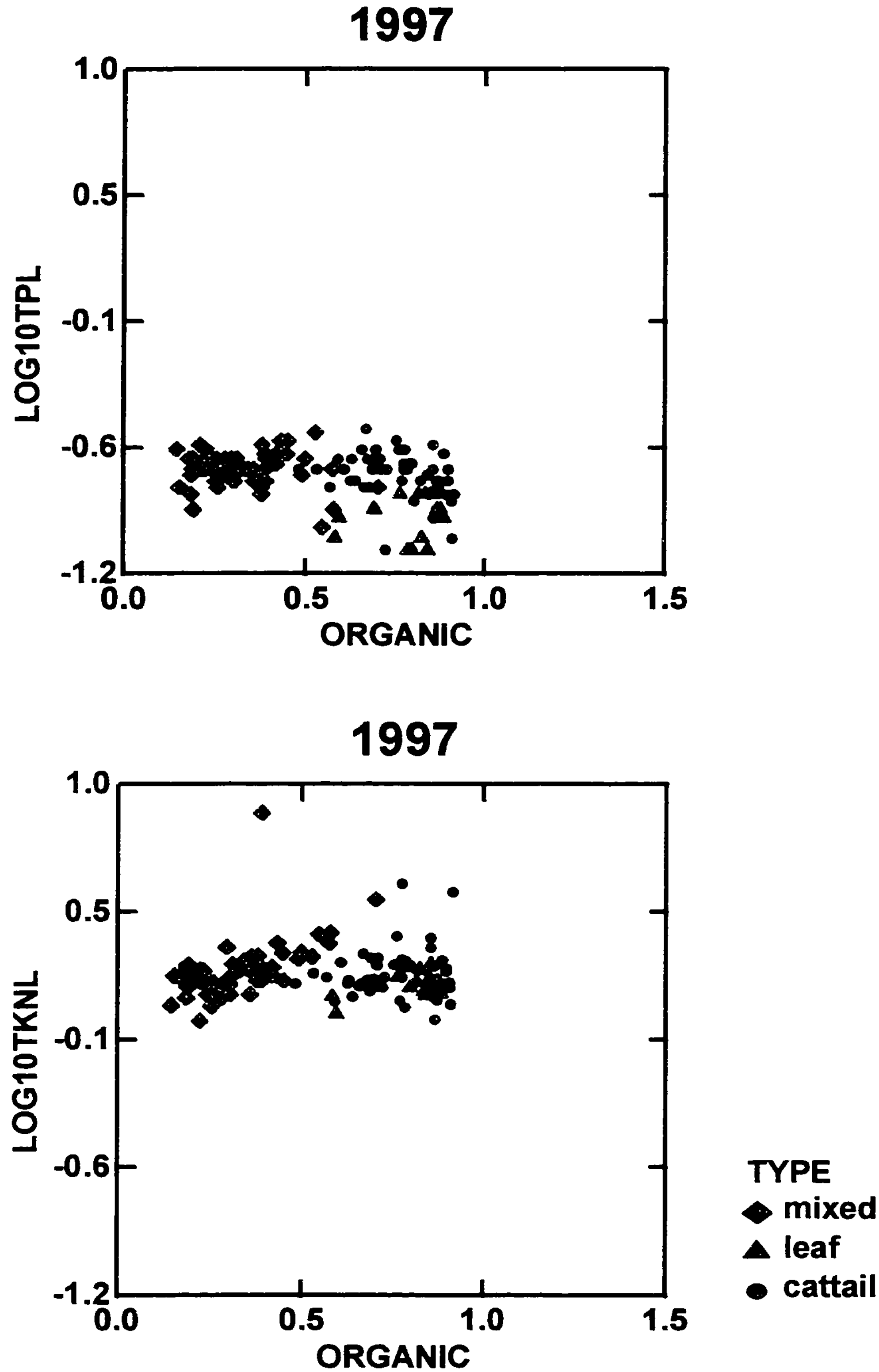

Fig. 18: Relationship between log10 TP and TKN concentrations (% dry weight) in litter and percent organic matter (by weight) during 1997 when all three litter types were collected.

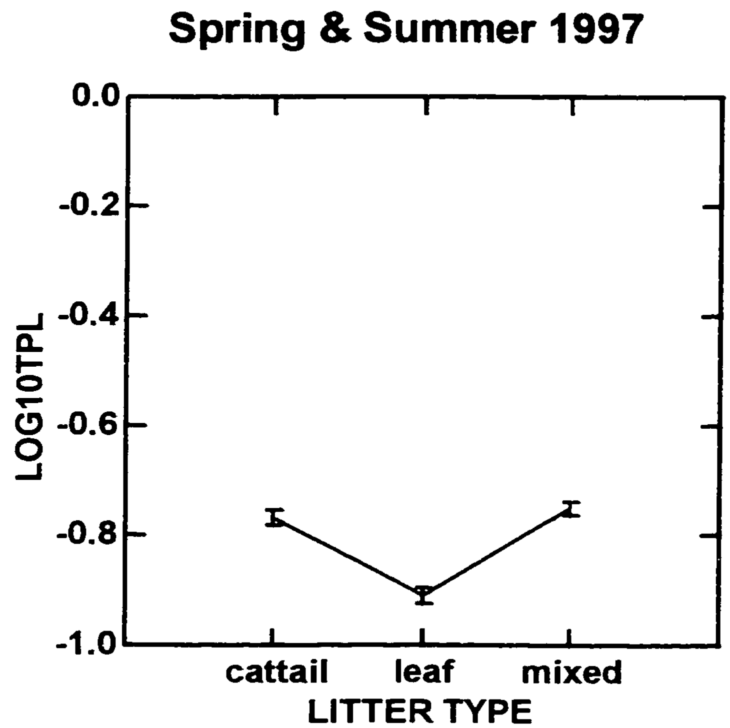

Fig. 19a: Relationship between mean log10 TP concentrations, % dry weight (+/-se) in litter and litter type.

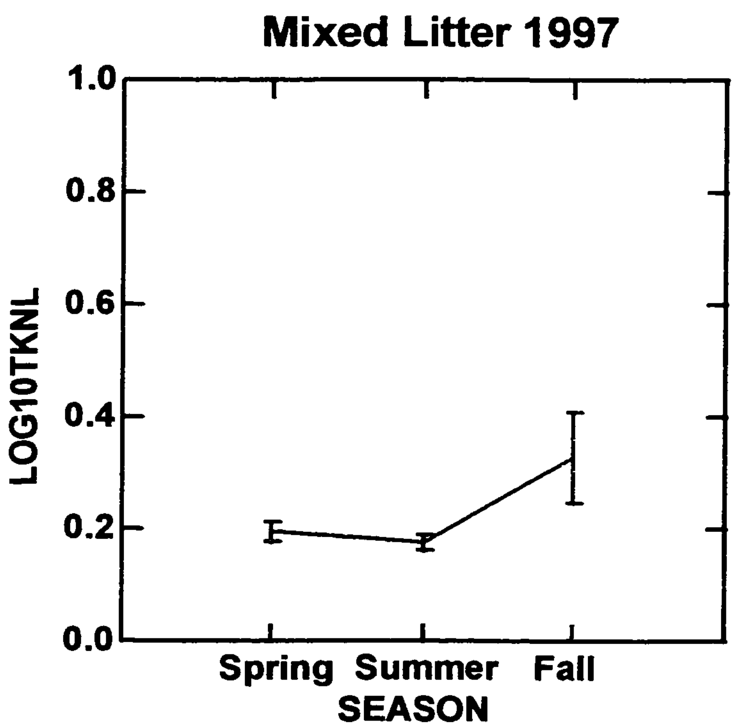

Fig. 19b: Relationship between mean log10 TKN concentrations, % dry weight (+/-se) in mixed litter and seasons in 1997.

3.5 Relationship Between Nutrient Concentrations in Litter and Water Column

Figure 20 shows the relationship between mean log10 mixed litter and water phosphorus concentrations on a seasonal basis combining the two year study period. Linear regressions (Table 13) for this relationship, within a given season and combining both years, only show a significant positive result for the summer season.

Figure 21 shows the relationship between mean log10 mixed litter and water nitrogen concentrations on a seasonal basis combining the two year study period. Linear regressions (Table 14) of this relationship, within a given season and combining both years does not show significant results for any season.

3.6 Iron as an Indicator of Phosphorus Release from Sediments

Fig. 22 shows the seasonal relationship between log10 water phosphorus and iron concentrations during the two year study period. In the spring of each year there appears to be a tight positive relationship between water phosphorus and iron concentrations. During the summer, this positive relationship begins to break up, particularly in 1998. Early fall of 1998 displays a positive relationship similar to that of the summer months. However, in the fall of 1997, a dramatic disassociation is seen between concentrations of phosphorus and iron within the two wetland cells, with two very distinct levels occurring.

Linear regressions (Table 15) of total water phosphorus and iron concentrations within a given season shows a significant positive relationship in all seasons and years, except late fall of 1997.

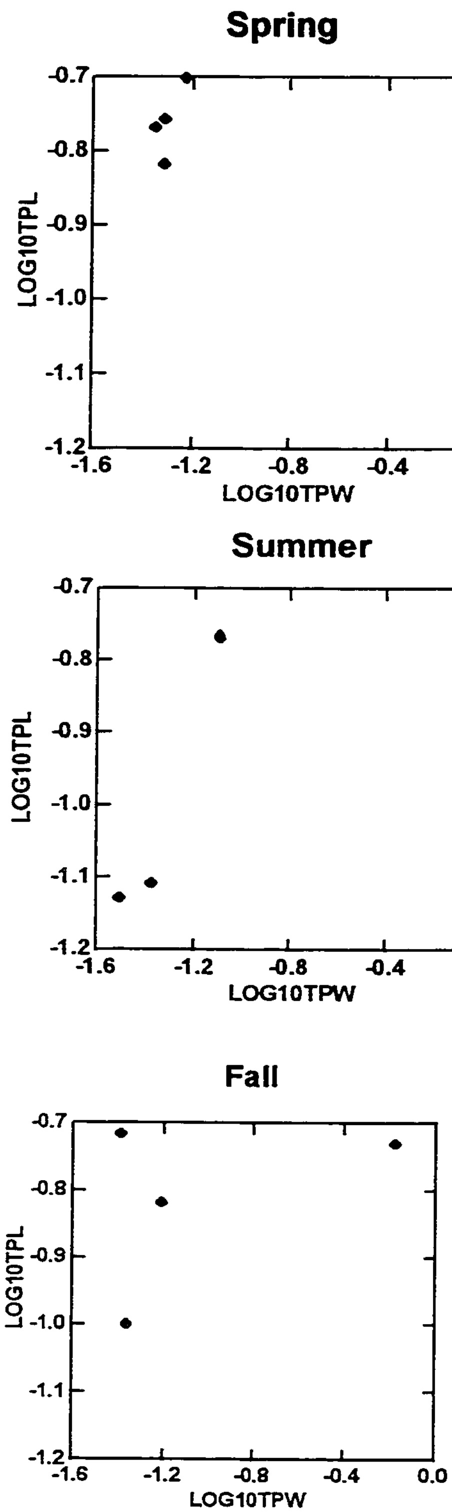

Fig. 20: Relationship between mean log10 total phosphorus concentrations in mixed litter (% dry weight) and water (mg/L) on a seasonal basis combining both years. Points represent means over replicates and stations within a cell. Standard error: TPL 0.015-0.127, TPW 0.023-0.091.

Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
Spring	0.671	0.114	0.446	4	0.002	0.531	1.506	0.271
Summer	0.965	0.280	0.154	4	0.003	0.952	6.269	0.025
Fall	0.099	-0.714	0.143	4	0.021	0.191	0.687	0.563

Table 13: Regression of log10 total phosphorus concentrations in mixed litter (% dry weight) and water (mg/L) within a given season combining both years. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

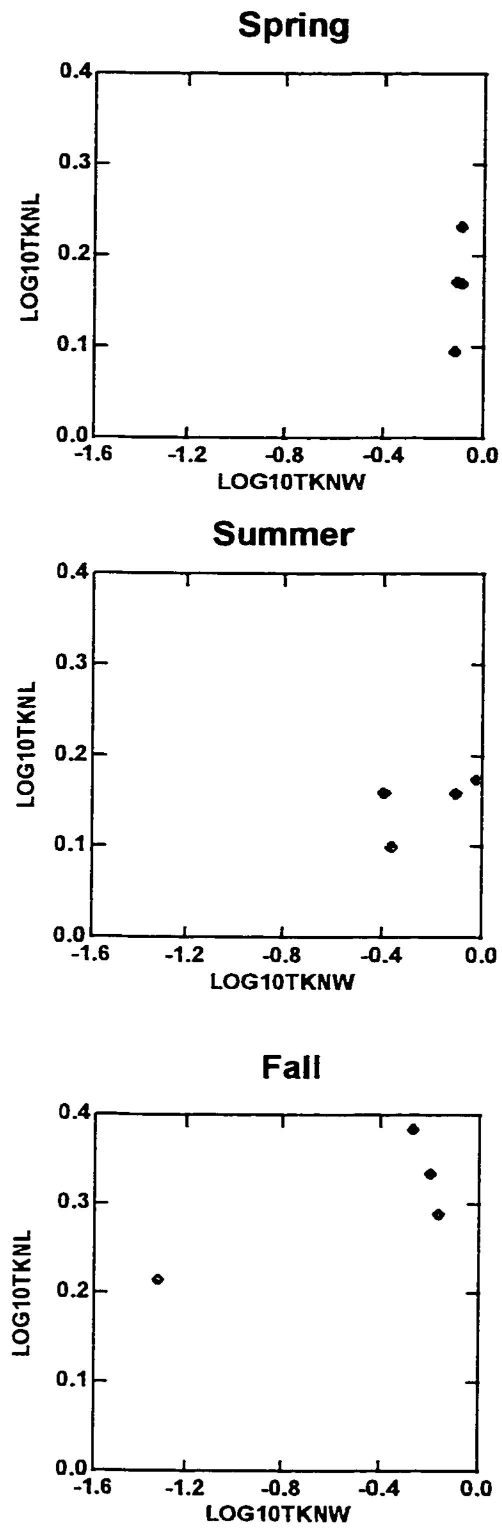

Fig. 21: Relationship between mean log10 total nitrogen concentrations in mixed litter (% dry weight) and water (mg/L) on a seasonal basis combining both years. Points represent means over replicates and stations within a cell. Standard error: TKNL 0.014-0.100, TKNW 0.012-0.061.

Season	Regression Coefficient	Y-Intercept	SE	N	RMS	R^2	t	p (2 Tail)
Spring	3.176	0.489	1.999	4	0.002	0.558	1.589	0.253
Summer	0.111	0.173	0.099	4	0.001	0.386	1.120	0.379
Fall	0.103	0.356	0.055	4	0.003	0.635	1.865	0.203

Table 14: Regression of log10 total nitrogen concentrations in mixed litter (% dry weight) and water (mg/L) within a given season and combining both years. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

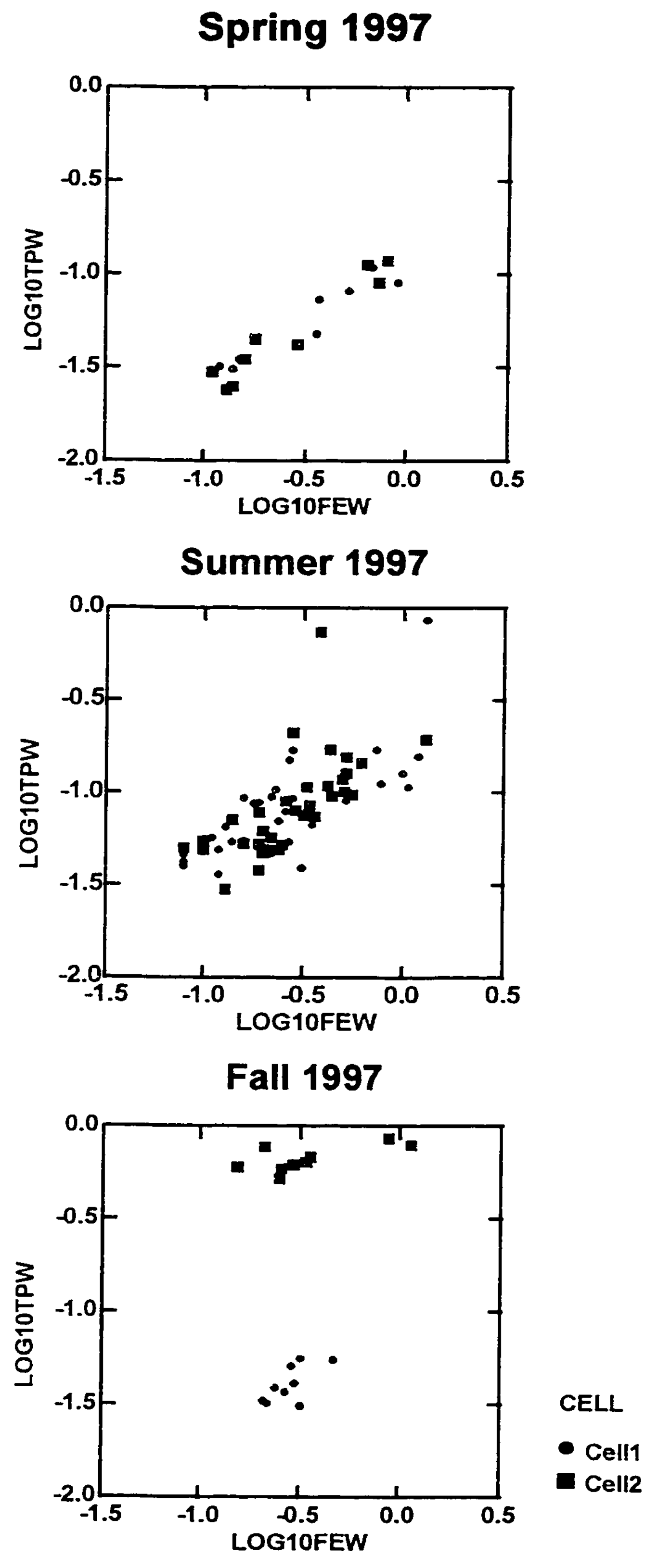

Fig. 22a: Relationship between log10 total phosphorus and total iron concentrations (mg/L) in water by season and cell in 1997.

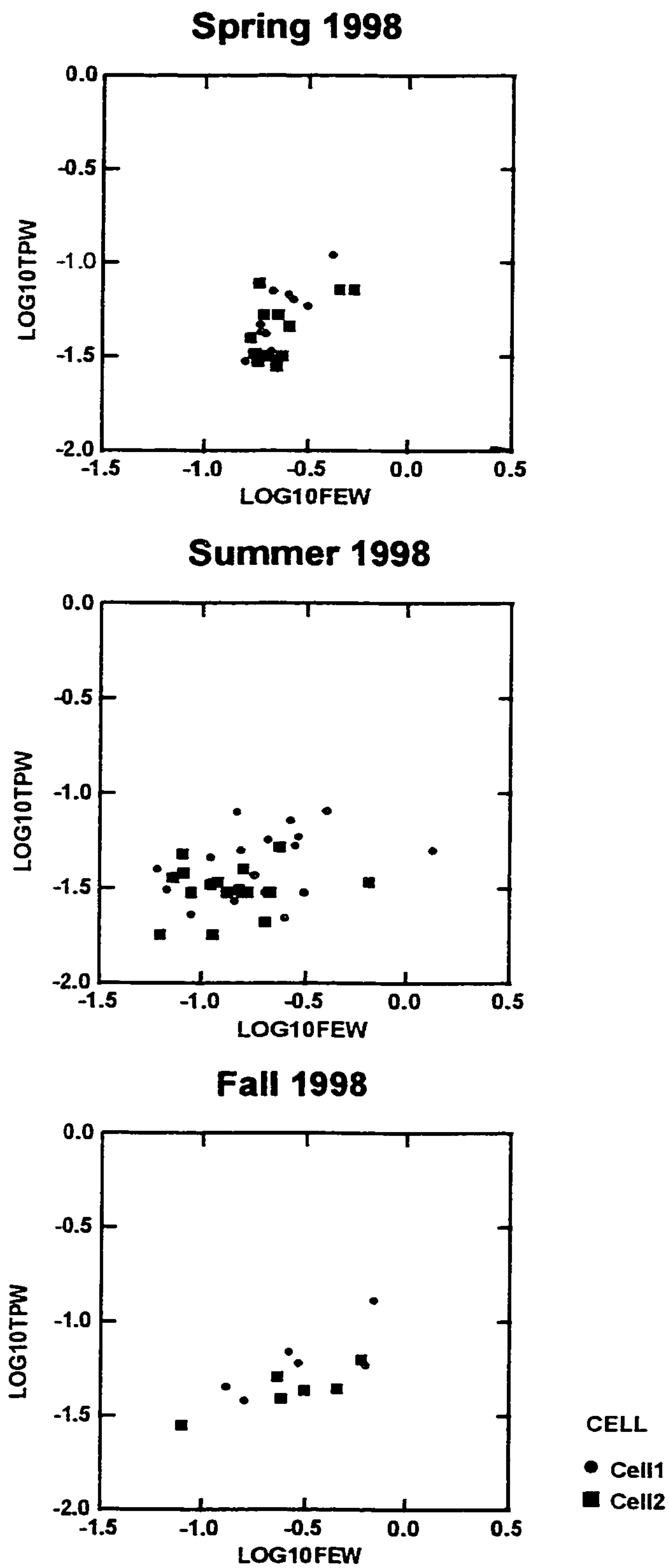

Fig. 22b: Relationship between log10 total phosphorus and total iron concentrations (mg/L) in water by season and cell in 1998.

Year	Season	Regression Coefficient	Intercept	SE	N	RMS	R^2	t	p (2 Tail)
1997	Spring	1.329	1.176	0.102	18	0.010	0.914	13.076	0.000
1997	Summer	0.810	0.297	0.099	72	0.047	0.490	8.209	0.000
1997	Fall	0.092	-0.431	0.081	18	0.044	0.074	1.134	0.274
1998	Spring	0.505	0.038	0.137	24	0.012	0.381	3.676	0.001
1998	Summer	0.576	0.043	0.278	36	0.073	0.112	2.070	0.046
1998	Fall	1.268	1.083	0.368	12	0.042	0.542	3.442	0.006

Table 15: Regression of log10 total phosphorus and total iron concentrations (mg/L) in water within a given season. SE: stand error of regression coefficient; N: sample size; RMS: residual mean square; and R^2: coefficient of determination.

Chapter 4 - DISCUSSION

4.1 Constructed Wetlands as Nutrient Sinks

Phosphorus retention in the Monahan Drain constructed wetland during the growing season (spring and summer) was somewhat consistent with results reported by Kadlec and Hey (1994) for the first three years after construction of the Des Plaines River Wetlands demonstration project (concentration change calculation). They reported spring retention of 23-99% as compared to 21-53% for the Monahan Drain constructed wetland. Summer retention for Des Plaines ranged from 47-92% as compared to 48-57% for Monahan Drain. While the lower ranges are quite comparable, the upper ranges for the Des Plaines Wetlands were much higher for both seasons. This may be due to the use of four marsh designs in the Des Plaines project, including variable water retention rates within the different wetlands. Fall retention results for the two constructed wetland projects were very different. The Des Plaines wetlands reported fall and winter retention rates of 47-88% and 21-68%, respectively. Results from the Monahan Drain wetland indicate retention of 21% in early fall (mid-October 1998), with a huge release of phosphorus in late fall (mid-November 1997) of -1869%. Only Devito et al. (1993) reported similar results, -34% (mass balance calculation) during the winter as part of a two year study of a natural beaver pond. It should be noted that this beaver pond is located at the same latitude and longitude as the Monahan Drain wetland, (45°N, 75°W).

The Final Design Report for the Monahan Drain Constructed Wetland (J.L. Richards, 1993) recommends an objective of <0.03 mg/L for total phosphorus at the outlet as part of a start-up monitoring plan for the treatment facility. Based on seasonal mean total phosphorus concentrations in water samples during the two year study period, this threshold was exceeded at the outlet in all seasons and years, except for the summer of 1998 when a mean value of 0.03 mg/L was calculated.

Nitrogen reduction results (Table 16) for the Monahan Drain constructed wetland differ from two research studies that reported seasonal results (Lemon et al., 1996 and Devito et al., 1993).

This variability could be explained in several ways: by the type of system used by Lemon et al. (1996), a subsurface flow system which is generally considered to be more effective in the treatment of nitrogen than surface flow systems (Hammer and Knight, 1994); the mass balance method used to calculate nitrogen reduction by Devito et al. (1993); and/or the much greater age of the natural beaver pond system used by Devito et al. (1993).

Since a late fall flux in phosphorus concentrations was not recorded for Cell 1, including at inflow station L1, it is likely that the major increase observed in the Cell 2 water column was derived from internal sources. The project engineer (D. Lalande, J.L. Richards & Associates, personal communication) indicated that it is unlikely that the significant increases observed would have resulted from groundwater sources, since in addition to lining the wetland with an impervious clay layer, several pre-existing springs and wells were sealed as part of the construction of the wetland. The late fall release of large concentrations of phosphorus in 1997 is consistent with work by Devito et al. (1993) and it is considered characteristic of reduced conditions resulting from winter ice cover in cold climate aquatic systems (Devito et al., 1993; and Lemon et al., 1996).

Season	Monahan Drain (conc. change calculation)	Lemon et al. (conc. change calculation)	Devito et al. (mass balance calculation)
Spring	19 to 27%	<-50%	-23%
Summer	-22 (1997) to 23%(1998)	>65%	71%
Fall	-94%(1997) to 28% (1998)	<-50%	-8%

Table 16: Comparison of total nitrogen reduction results

Although there is a major reduction in the concentration of nitrogen in the Cell 2 water column in late fall of 1997, there was no corresponding increase in the concentration of nitrogen in the litter compartment. The timing and magnitude of this decline in water nitrogen concentrations is consistent with what could be expected in the presence of reduced conditions. Denitrification is only significant when very low or zero oxygen conditions are present (Lee et al., 1975). A high rate of denitrification would result in the transformation of large amounts of soluble inorganic nitrogen to atmospheric nitrogen. Nitrogen would therefore be lost to the aquatic system, and would not build up in the substrate or water column. Both Kadlec and Hey (1994) and Lemon et al. (1996) reported significant nitrogen reduction (concentration change calculations) by denitrification within recently constructed wetland systems.

The large decrease in the concentration of nitrogen in the water column, coupled with the large increase in phosphorus concentrations, are both consistent with what would be expected from anoxic conditions. Anoxic conditions could be produced in temperate regions by the cessation of photosynthesis at the end of the growing season, depletion of dissolved oxygen in the water due to an increase in bacterial respiration breaking down newly deposited litter and the onset of ice cover as the temperature falls below zero in late fall (Richardson et al. 1985; and Waddell 1985). Much more extensive submerged aquatic plant growth in Cell 1, as compared to Cell 2 (Photos 1 and 2) was observed throughout the growing seasons of both years. Submerged vegetation was also observed to persist well past completion of the life cycle of emergent aquatic vegetation. This prolonged growing season for submerged aquatic plants, producing an ongoing supply of oxygen to the water column, may be at least partially responsible for the delayed anoxic conditions in Cell 1 during late fall of 1997. Although, the wetland was free of ice cover on November 9, 1997, during field sample collection, Environment Canada data, from the Macdonald-Cartier Airport weather station recorded sub-zero temperatures and snowfall for several weeks beginning on October 18, 1997. Therefore, periods of ice cover could have occurred in advance of the November 9, 1997 sampling date. Work by Fortin et al. (in press), at the Monahan Drain constructed wetland, recorded that some sediments in Cell 1 were anoxic during October 1997 and in Cell 2 during August 1998. It may be that the cusp season, between late fall/early winter, with potentially alternating periods of ice cover and open water, produces ideal conditions for the production and escape of gaseous nitrogen from aquatic systems.

Water depth may be a contributing factor explaining the lower phosphorus retention capability of the Monahan Drain constructed wetland as compared to other wetland systems. In an examination of 101 data points compiled by the Environmental Protection Agency for free surface water wetland treatment systems in the United States, Moustafa (1999) reported that the highest P-removal efficiency (70 to 95%) is obtained at water depths less than 20 cm, with a rapid decrease to less than 50% at water depths greater than 80 cm. Post-construction contour plans of the Monahan Drain constructed wetland indicate that >70% of the wetland has a water depth of more than 50 cm. Lower P-retention efficiency could therefore be expected for this wetland during the growing season.

Although Hammer and Knight (1994) reported that the optimum size of deep zones (minimum 1.5 metres deep) for best treatment of nitrogen is between 10-20% of the area of a treatment wetland, with deep zones arranged intermittently along the length of the wetland, this was not apparent in this study. Approximately 18% of Cell 1 and 4% of Cell 2 are deeper than 1.5 metres, and deep areas tended to be concentrated in one area. Significant treatment of nitrogen was only recorded in Cell 2, in November of 1997. This cell contained the smallest percentage of deep zones.

Photo 1: Extensive growth of submerged aquatic plants in Cell 1.

Photo 2: Aquatic plant growth mainly confined to littoral emergent species in Cell 2.

4.2 Relative Retention of Nutrients

Results of the relative change in average phosphorus and nitrogen concentrations at the inflow and outflow stations indicate no statistical evidence of greater phosphorus than nitrogen retention by the wetland. In addition, analysis of seasonal N/P ratios shows that all values, except for late fall of 1997, are in the general range of 16:1, the ratio at which most aquatic organisms typically use nitrogen and phosphorus. Ratios in the range of 5:1 during the growing season would have indicated possible nitrogen limitations, while ratios in the range of 30:1 would have indicated possible phosphorus limitations. According to Schindler (1977) low N/P ratios in the range of 5:1 also favour the growth of objectionable nitrogen-fixing blue-green algae as the system tries to make up for a nitrogen deficiency in an aquatic system. This is consistent with observations by F. Pick, University of Ottawa, Department of Biology (personal communication), who found little indication of nitrogen fixing algal species in samples taken from the Monahan Drain constructed wetland during the period covered by this research. Therefore, during the first two years after construction of the wetland, neither phosphorus or nitrogen appear to be limiting the growth of wetland biota. However, if the trend toward increasing N/P ratios continues with subsequent years of the wetlands development, an excess of nitrogen may cause future phosphorus limitations during the growing season, even though high denitrification during late fall causes a steep decline in this ratio.

4.3 Plant Litter and Nutrient Retention

Analyses of phosphorus and nitrogen litter concentrations reveals a great deal of uniformity among stations (phosphorus only) and seasons during 1997, followed by a much greater degree of variability among both stations and seasons in 1998. It may be that during the first year after construction, microhabitats throughout the wetland are very similar from station-to-station. In the second year, as local differences begin to develop, such as the growth of variable densities and types of aquatic plants and different accumulation patterns for plant litter, greater spatial variability would result in more diverse nutrient retention in the litter compartment.

Mixed litter showed different patterns of phosphorus retention than cattail litter. Results for mixed litter are consistent with the prediction that microbial assimilation is an important mechanism for phosphorus retention, while results for cattail litter are consistent with the prediction that sediment adsorption and precipitation are important factors. These results are surprising in that it would have been expected that mixed litter, with the lowest organic matter content of the three litter types, would have been more likely to support the sediment adsorption and precipitation mechanism, and vice versa. It may be that floating cattail litter is subject to little biological activity, compared to mixed litter which had accumulated naturally on the bottom of the wetland, and therefore a good assessment of the full scope of its nutrient retention dynamics would be difficult. It should also be noted that a significant positive relationship between mixed litter phosphorus concentrations and percent organic matter was only observed in the summer of 1998, during the second year of the wetlands maturing process. This is consistent with work by Kadlec (1989), in which he stated that development of a microbial population in the sediment compartment typically takes 4 to 6 years in newly constructed wetlands. Therefore, the impact of the microbial population can be expected to produce a greater effect in the second year of the wetland's development, and since microbial activity is enhanced with increasing temperature, microbial assimilation could also be expected to be greatest during the summer. In general, it is likely that a good assessment of the retention capability of plant litter is best undertaken at a later stage of the wetland's development. However, such preliminary data provides an important point of comparison in determining the magnitude of net retention over time.

Since nitrogen does not tend to build-up in wetland sediments, it is very difficult to test the effectiveness of plant litter for nitrogen retention. This can be seen in Fig. 13 which shows very little variation in mixed and cattail litter nitrogen concentrations both within a month, from month-to-month, and from year-to-year. Results of an analysis of litter nitrogen concentrations and percent organic matter content really only confirm that nitrogen is a major component of organic matter. Results from an analysis of litter nitrogen concentration and water temperature give a better indication of how nitrogen cycles within the wetland by indicating that decomposition is an important mechanism for nitrogen release from the litter compartment.

Plant litter generally shows consistently smaller phosphorus and nitrogen concentrations at the outflow station (L6) than at the inflow station (L1). This is consistent with a similar pattern for water nutrient concentrations. In addition, there is a pattern of a decrease in water and litter phosphorus concentrations from year one to year two, particularly during the growing season from mid-June to mid-September. This is likely due to greater phosphorus demand by the more extensive, second-year growth of emergent vegetation in the littoral areas from which litter samples were collected. Litter nitrogen concentrations do not show a similar pattern during the growing season. It may be that the nitrogen requirements of aquatic vegetation are being met by dissolved inorganic nitrogen from the water column. Such a pattern of reduced nitrogen concentrations is evident for 1998 water samples. A pattern of decreasing water and litter nitrogen concentrations from year-to-year might also be expected due to the large amount of nitrogen lost from the wetland system through denitrification in late fall of 1997.

4.4 Different Types of Plant Litter and Nutrient Retention

The results indicate that in the two seasons for which all three litter types were collected (spring and summer of 1997), mixed and cattail litter were significantly better at retaining total phosphorus than leaf litter, while no significant difference was found in the ability of the three litter types to retain total nitrogen. This differs somewhat from similar studies by Mason and Bryant (1975); Day (1982); and Davis (1991) where similar phosphorus and nitrogen retention was recorded among the litter of different species. For litter types consisting of a high percentage of organic matter (cattail and leaf litter) this might be expected to occur, since variable nutrient retention resulting from differences in the growth dynamics of different living plant species and plant parts might disappear once the plants die and their plant tissue is broken down into simple compounds and elements. At such a state of decomposition, the resulting litter might be expected to behave in a similar manner. In the case of mixed litter, which was found to have a much higher mineral fraction, significantly better phosphorus retention would have been expected as compared to cattail litter.

Results, for the two litter types that were collected over all three seasons, demonstrate that although mixed and cattail litter were similarly effective in retaining both total phosphorus and total nitrogen during the spring and summer of 1997, mixed litter demonstrated significantly better total nitrogen retention in late fall of 1997. For cattail litter, these results are consistent with work by Day (1982) in which he found seasonal nutrient dynamics in decaying litter to be similar irrespective of litter type.

These results imply that no special effort or cost is needed in specifying the use of organic matter in constructed wetland design, since the imported litter type (leaf litter) was at best equally effective at nitrogen retention, and at worst the least effective litter type at phosphorus retention. These results also appear to indicate that naturally accumulating mixed litter is more effective in retaining nitrogen in the fall than deadfall cattail litter. This may be because the naturally accumulating litter is more biological developed and complex than either cattail or leaf litter and/or the significant fall increase in litter nitrogen may be related to new litter inputs at the end of the growing season. In any case, at this time there does not appear to be any compelling reason to accelerate the development of the wetland's litter compartment through the application of an organic layer as part of the work of the initial wetland construction.

4.5 Relationship Between Nutrient Concentrations in Litter and Water Column

Because interaction between the sediment and water compartments is complex, involving few instantaneous processes, a rougher scale of analysis was employed for this part of the study. Therefore data were combined on a seasonal basis over replicates and stations within each cell (Cell 1 and Cell 2). Data were also combined over both years.

In this study a weak significant relationship between litter and water total phosphorus was only found during the summer season. The results of this analysis are somewhat consistent with the work of Davis (1991), where a relationship was been found between litter and water total phosphorus, but not between litter and water total nitrogen. Since his work dealt with a sub-tropical wetland exposed to a broad nutrient gradient, from eutrophic to oligotrophic waters, a relationship might be more likely to be apparent than would be the case for a temperate wetland with mostly low eutrophic waters. For the Monahan Drain constructed wetland, a more frequent sampling interval might have improved the ability to detect a relationship between nutrient concentrations in the litter and water column. A sampling interval (bi-weekly) which better reflected the water retention time of the study wetland would likely have been preferable for this work. It is also possible that the substrate in this two year old constructed wetland may not be sufficiently saturated with phosphorus to clearly observe a dynamic exchange between the two compartments. In addition, if the relationship between different forms of phosphorus and nitrogen were assessed, instead of just total phosphorus and total nitrogen, the results might have been quite different. For example, Puriveth (1980) demonstrated that soluble reactive phosphorus and ammonium-nitrogen in the water column showed significant intercorrelation with litter phosphorus and nitrogen.

4.6 Iron as an Indicator of Phosphorus Release from Sediments

Results of the relationship between iron and phosphorus concentrations in the water column do not provide any direct evidence of a relationship between late fall phosphorus release to the water column and the iron content of the wetland sediments. They do however, tell us that iron is well represented within the wetland system, and that there is a significant positive relationship between water phosphorus and iron concentrations in all seasons and years studied, except late fall of 1997.

These results also tell us that there is a strong significant positive relationship between water phosphorus and iron concentrations in the spring of each year, likely as a result of the annual input of runoff from snow melt within the watershed when phosphorus and iron are washed into the wetland in similar proportions. During the summer and early fall, as plants and other biota assimilate phosphorus and iron, this significant positive relationship begins to break up. As discussed in Section 4.3, this loosening of the relationship between the two elements may be particularly noticeable in 1998 as the wetland develops increasingly variable microhabitats. The dramatic late fall flux in water phosphorus concentrations is likely related to the release of previously adsorbed phosphorus from the sediment compartment and is consistent with depletion of water oxygen as a result of the death and decay of wetland vegetation at the end of the growing season (Richardson et al., 1985) and/or the development of an ice cover which prevents wind induced mixing of atmospheric oxygen with the water column (Knowles and Lean, 1987). As previous discussed in Section 4.1, the very different pattern within the two wetland cells is likely related to delayed anoxia in Cell 1.

4.7 Implications for Design and Management

A reoccurring pattern of late fall or winter release of phosphorus from cold climate wetlands needs to be better addressed in the design and management of constructed treatment wetlands. While it is recognized that release of phosphorus outside the growing season is beneficial for the reduction of eutrophication within downstream aquatic systems during the time of the most serious algal blooms (Lee et al., 1975; Nichols, 1983; Mitsch and Gosselink, 1993), more attention and creativity needs to be focused on mitigating this pathway of nutrient release. Several authors have tried to provide some direction in this matter. Lee et al. (1975), Spangler et al. (1977), Richardson et al. (1985) and Mitsch and Gosselink (1993) thought that there was good potential to use phosphorus flushing periods to an advantage, by using lagoons or some other method of impoundment for temporary storage of flushed water. This stored water could then be recycled through the wetland at a later time, or diverted to a conventional treatment plant. Sloey et al. (1978) suggested diverting or pumping flushed nutrients onto agricultural land by means of an overland flow type of operation.

Nichols (1983) thought that if the bottom of a wetland were dredged to expose fresh mineral soil, some of the phosphorus retention capacity of the wetland could be restored. According to Welch and Lindell (1992) the phosphorus retention capability of a Swedish lake was restored by removal of the top 1 metre of nutrient rich sediment However, work by Kleeberg and Kohl (1999) on sediment of a shallow eutrophic lake in Germany, showed that dredging, without reducing the external phosphorus loading in the catchment area, may give only temporary improvement followed by a slow return to pre-dredge conditions.

Richardson et al. (1985) also had several other interesting suggestions for dealing with the phosphorus flush:

1. A two wetland system in which inflowing water is alternately applied to each wetland, allowing recovery periods between treatments;
2. Pretreatment of the water to remove most of the phosphorus by chemical precipitation (e.g. alum treatment), prior to discharge to downstream parts of the watershed;
3. Periodic flushing of the wetland to control the build-up and/or release of phosphorus, at a time when downstream impacts could be minimized;
4. Introduction of silt-laden water into the wetland to increase or restore the adsorption characteristics for phosphorus removal.

Richardson et al. (1985) also stated that while aquatic vegetation is useful in increasing uptake of nutrients, this storage compartment is not considered a major nutrient sink, and therefore biomass harvesting to permanently remove some phosphorus and nitrogen from the system would have minimal benefits while significantly increasing the treatment cost of the wetland facility.

There was concern regarding the installation of a self-weathering steel bridge across the middle of the Monahan Drain constructed wetland, dividing Cells 1 and 2. Communication with the suppliers of this type of bridge revealed that for the first five to six years after installation, this type of bridge would leach iron oxide into the water, ultimately forming a tough skin layer. It seems that the environmental impacts associated with this type of leaching on water quality have not been addressed to any great extent. While individual impacts may be small, cumulative impacts could potentially be significant.

It is becoming evident that monitoring of the effectiveness of a temperate constructed wetland for nutrient mitigation should extend throughout the year. Current monitoring has largely been confined to sample collection within the May to September swimming season. As long as pollutant concentrations stay within stated guidelines focused on human health, there has been little emphasis on year round or cumulative ecological impacts. This narrow seasonal time frame produces misleading results. High nutrient retention is reported during the growing season, with fall/spring flushing or transformation not factored in to the calculation of net annual nutrient retention.

Chapter 5 - CONCLUSIONS

Based on this research, and other relevant studies of aquatic systems, it appears that temperate constructed wetlands act as good sinks for phosphorus and nitrogen, especially during the summer growing season. However, in late fall or winter, they act as a major source for phosphorus, and as a major transformer of nitrate-nitrogen to gaseous nitrogen. Although, from a water quality perspective, this late fall phosphorus flux has some mitigative ecological benefits, it still is a problem that requires greater attention in the design and management of constructed wetlands. Diversion of nitrogen into the atmosphere is a positive result from a water quality point of view, however, before widespread use of nitrogen diverting systems are utilized, further research addressing possible cumulative impacts of this source of nitrogen oxides is needed. Transferring pollution problems from water to atmospheric compartments does not seem to be desirable.

The release of phosphorus and the transformation of nitrogen in late fall appears to be largely related to anoxic conditions unique to cold climate wetlands. The fall die-off of photosynthesizing plants, low temperatures and the prevention of oxygen mixing with the water column by ice-cover provide anoxic conditions within the Monahan Drain constructed wetland. It is, therefore, important to recognize that the treatment benefits associated with a cold climate constructed wetland differ from those of more tropical wetlands (Elder, 1985; Reddy and Burgoon, 1996). Cold climate treatment wetlands must therefore be designed and managed differently.

While the plant species origin of organic matter does not appear to make a significant contribution to the nutrient retention capability of a constructed wetland, the total amount of organic matter certainly does make a difference. The percent content of organic matter within the wetland substrate is an important factor in the success of a constructed wetland for both the retention and transformation of nutrients. Wetlands with mineral sediments will be more effective in retaining phosphorus when aerobic conditions prevail, and organic sediments are essential for denitrification under anaerobic conditions. Some indication that water depth may play an important role in phosphorus mitigation is derived from work by Moustafa (1999), who reported the highest phosphorus removal rates in water less than 20 cm deep based on EPA data from a large number of surface flow treatment wetlands in the United States. It is also appears that the type of substrate and/or water depth that is effective for the treatment of phosphorus is not likely to be similarly effective for the treatment of nitrogen. A constructed wetland will therefore need to be design for the nutrient of greatest concern within the watershed, since the effective treatment of both nutrients will be difficult.

More research is needed on natural and constructed wetland treatment systems, especially on constructed wetlands that have been in operation for a decade or more. A meta-analysis of statistical data from a collection of such research might provide some clearer answers to two important questions that can only be properly address with this type of analyses:

1. Are natural and constructed wetlands equally effective in nutrient mitigation?
2. Does the presence, absence or duration of ice cover have a significant impact on the effectiveness of nutrient mitigation?

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REFERENCES

Amell, B. and B.K. Eastlick (1996) Design for multiple benefit constructed wetlands. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium*. Niagara-on-the-Lake, Ontario, June 4-5, 1996

Baker, L.A. (1993) Introduction to Nonpoint Source Pollution and Wetland Mitigation. In: *Created and Natural Wetlands for Controlling Nonpoint Source Pollution*. Smoley, C.K. (ed.), CRC Press, Boca Ration, Fl. pp. 7-41.

Boström, B, J.M. Andersen, S. Fleischer, and M. Jansson (1988). Exchange of phosphorus across the sediment-water interface. *Hydrobiologia* 170: 229-244.

Boström, B. and K. Pettersson (1982). Different patterns of phosphorus release from lake sediments in laboratory experiments. *Hydrobiologia* 92: 415-429.

Chamie, J.P.M. and C.J. Richardson (1978). Decomposition in northern wetlands. In: *Freshwater Wetlands* eds. Good, R.E, D.F. Whigham, and R.L. Simpson. pp. 115-130.

Davis, S.M. (1991) Growth, decomposition, and nutrient retention of *Cladium jamaicense* Crantiz and *Typha domingensis* Pers. In the Florida Everglades. *Aquatic Botany*, 40: 203-224.

Day, F.P. Jr. (1982) Litter decomposition rates in the seasonally flooded Great Dismal Swamp, *Ecology* 63(3): 670-678.

Devito, K.J. and P.J. Dillion (1993). Importance of runoff and winter anoxia to the P and N dynamics of a beaver pond. *Can. J. Fish. Aquatic. Sci.*, Vol. 50: 2222-2234.

Elder, J.F. (1985) Nitrogen and Phosphorus Speciation and Flux in a Large Florida River Wetland System, *Water Resources Research*, Vol. 21, No. 5: 724-732.

Environment Canada (1987) *The Canadian Wetland Classification System*. By the National Wetlands Working Group. Ecological Land Classification Series, No. 21. Canadian Wildlife Service.

Faulkner, S.P. and C.J. Richardson (1989) Physical and chemical characteristics of freshwater wetland soils. In: *Constructed Wetlands for Wastewater Treatment*. Hammer, D.A. (ed.), pp 41-72.

Fortin, D., R. Goulet and M. Roy (in press) Seasonal cycling of Fe and S in a constructed wetland: the role of sulfate-reducing bacteria. *Geomicrobiology Journal*.

Gessner, M.O. (2000) Breakdown and nutrient dynamics of submerged Phragmites shoots in the littoral zone of a temperate hardwater lake. *Aquatic Botany* 66: 9-20.

Gore & Storrie Limited (1992) *Environmental Analysis of the Monahan Drain Watershed*, July 1992. City of Kanata.

Government of Canada (1991) *The Federal Policy on Wetland Conservation*. Environment Canada.

Guntenspergen, G.R., F. Stearns, and J.A. Kadlec (1989) Wetland Vegetation. In: *Constructed Wetlands for Wastewater Treatment*. Hammer, D.A. (ed.), pp 73-88.

Hammer, D.A. and R.L. Knight (1994). Designing constructed wetlands for nitrogen removal. *Water Science and Technology*, 29(4): 15-27.

Hasset, J.J. and W.L. Banwart (1992). *Soils and their environment*. Prentice-Hall.

Hietz, P., 1992. Decomposition and nutrient dynamics of reed (*Phragmites australis* (Cav.) Trin. Ex Steud.) litter in Lake Neusiedl, Austria. *Aquatic Botany*, 43: 211-230.

Horne, A.J. and C.R. Goldman (1994). *Limnology*. McGraw-Hill, New York. pp. 115-151.

J.L. Richards & Associates Limited (1993) *Monahan Drain Constructed Wetlands, Final Design Report*. City of Kanata.

Kadlec, R.H. (1989) Decomposition in wastewater wetlands. In: *Constructed Wetlands for Wastewater Treatment: municipal, industrial and agricultural*. Hammer, D.A. (ed.), pp. 459-468.

Kadlec, R.H. and D.L. Hey (1994) Constructed wetlands for river quality improvement. *Water Science and Technology*, 29(4): 159-168.

Kamp-Nielsen, L. (1974) Mud-water exchange of phosphate and other ions in undisturbed sediment cores and factors affecting the exchange rates. *Arch. Hydrobiol*. 73: 218-237.

Kleeberg, A. and J.G. Kohl (1999) Assessment of the long-term effectiveness of sediment dredging to reduce benthic phosphorus release in shallow Lake Müggelsee (Germany). Hydrobiologia 394: 153-161.

Knowles, R. and D.R.S. Lean (1987) Nitrification: a significant cause of oxygen depletion under winter ice. *Can. J. Fish. Aquat. Sci.* 44: 743-749.

Lee, G.F, E. Bentley, and R. Amundson (1975) Effects of marshes on water quality. In *Coupling of Land and Water Systems*. A.D. Hasler (ed.), Springer-Verlag, New York, N.Y. pp. 105-127.

Lemon, E., G. Bis, L. Rozema and I. Smith (1996) SWAMP pilot scale wetlands - design and performance. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium*. Niagara-on-the-Lake, Ontario, June 4-5, 1996

Livingston, R.J. and O.L. Loucks (1978) Productivity, trophic interactions, food-web relationships in wetlands and associated systems. In: P.E. Greeson, J.R. Clark, and J.E. Clark (eds.), Wetland Functions and Values: The State of Our Understanding. Proceedings of the National Symposium on Wetlands, American Water Resources Association, Minneapolis, Minnesota. pp. 101-119.

Loucks, O.L., R.T. Prentki, V.J. Watson, B.J. Reynolds, P.R. Weiler, S.M. Bartell, and A.B. D'Alessio (1977) *Studies of the Lake Wingra Watershed, An interim report.* Center for Biotic Systems, Inst. for Environmental Studies, Univ. of Wisconsin-Madison, Report 78.

Mackenzie, A., A.S. Ball and S.R. Virdee (1998) *Instant Notes in Ecology*, Springer-Verlag, New York.

Mason, C.F. and R.J. Bryant (1975) Production, nutrient content and decomposition of *Phragmites communis* Trin. and *Typha angustifolia* L., *J. Ecol.* 63: 71-95.

Miller, G. (1989) Use of artificial cattail marshes to treat sewage in northern Ontario, Canada. In: *Constructed Wetlands for Wastewater Treatment: municipal, industrial and agricultural.* D.A. Hammer (ed.) Lewis Publishers, Inc. pp. 636-642.

Ministry of the Environment (1983) Handbook of Analytical Methods for Environmental Samples, Laboratory Services and Applied Research Branch, Rexdale, Ontario.

Mitsch, W.J. and J.G. Gosselink (1993) *Wetlands.* Van Nostrand Reinhold, New York.

Mitsch, W.J. (1993) Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. In: *Created and Natural Wetlands for Controlling Nonpoint Source Pollution.* Smoley, C.K. (ed.), CRC Press, Boca Ration, Fl. pp. 43-69.

Mortimer, C.H. (1971) Chemical exchanges between sediments and water in the Great Lakes - Speculations on probable regulatory mechanisms. *Limnology and Oceanography*, 16(2): 387-404.

Moustafa, M.Z. (1999) Analysis of phosphorus retention in free-water surface treatment wetlands. *Hydrobiologia* 392: 41-53.

Murphy T.P. and B.G. Brownlee (1981) Ammonia volatilization in a hypertrophic prairie lake. *Can. J. Fish. Aquat. Sci.* 38: 1035-1039.

Nichols, D.S. (1983). Capacity of natural wetlands to remove nutrients from wastewater. *Journal WPCF*, 55(5): 495-505.

Nürnberg, G.K. (1984). The prediction of internal phosphorus load in lakes with anoxic hypolimnia. *Limnology and Oceanography*, 29(1): 111-124.

Ohlendorf, H.M. (1996) Ecological risk assessment for constructed wetlands. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium.* Niagara-on-the-Lake, Ontario, June 4-5, 1996

Pries, J. (1994) *Wastewater and Stormwater Applications of Wetlands in Canada.* Canadian Wildlife Service, Environment Canada and CH2M Hill Engineering Ltd, Report No. 1994-1.

Pries, J. (1996) Constructed treatment wetland systems in Canada. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium*. Niagara-on-the-Lake, Ontario, June 4-5, 1996

Puriveth, P. (1980) Decomposition of emergent macrophytes in a Wisconsin marsh. *Hydrobiologia* 72: 231-242

Reddy, K.R. and P.S. Burgoon (1996). Influence of temperature on biogeochemical processes in constructed wetlands: Implications to wastewater treatment. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium*. Niagara-on-the-Lake, Ontario, June 4-5, 1996

Richardson, C.J. and D.S. Nichols (1985) Ecological Analysis of Wastewater Management Criteria in Wetland Ecosystems, In: *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*, P.J. Godfrey, E.R. Kaynor and S. Pelczarski (eds.), Van Nostrand Reinhold, New York. pp. 351-391.

Richardson, C.J. (1985) Mechanisms Controlling Phosphorus Retention Capacity in Freshwater Wetlands, *Science*, 228: 1424-1427.

Richardson, C.J., D.L. Tilton, J.A. Kadlec, J.P.M. Chamie, and W.A. Wentz (1978) Nutrient Dynamics of Northern Wetland Ecosystems. In: *Freshwater Wetlands, Ecological processes and management potential*, R.E. Good, D.F. Whigham, and R.L. Simpson (eds.), Academic Press, New York. pp. 217-241.

Riemer, D.N. (1984) *Introduction to Freshwater Vegetation*. AVI Publishing Co., Westport, Connecticut.

Schindler, D.W. (1977) Evolution of phosphorus limitation in lakes. *Science*, 195: 260-262.

Schueler, T.R. (1992) *Design of stormwater wetland systems: guidelines for creating diverse and effective stormwater wetlands in the mid-atlantic region*, Metropolitan Washington Council of Governments, Washington, D.C.

Sloey, W.E., F.L. Spangler, and C.W. Fetter, Jr. (1978). Management of Freshwater Wetlands for Nutrient Assimilation. In: *Freshwater Wetlands, Ecological processes and management potential*, R.E. Good, D.F. Whigham, and R.L. Simpson (eds.), Academic Press, New York. pp. 321-340.

Spangler, F.L., C.W. Fetter, Jr., and W.E. Sloey (1977) Phosphorus accumulation - discharge cycles in marshes. *Water Resources Bulletin*, 13(6): 1191-1201.

Van Oostrom, A.J. and J.M. Russell. (1994) Denitrification in constructed wastewater wetlands receiving high concentrations of nitrate. *Wat. Sci. Tech.* 29(4): 7-14.

Verhoeven, J.T.A. and H.H.M. Arts (1992) *Carex* litter decomposition and nutrient release in mires with different water chemistry. *Aquatic Botany*, 43: 365-377.

Vitousek, P.M., J.Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and G.D. Tilman. (1997) Human alteration of the global nitrogen cycle: causes and consequences. *Ecological Applications*, Vol. 7.

Waddell, K.M., D.W. Darby and S.M. Theobald (1985) *Chemical and Physical Characteristics of Water and Sediment in Scofield Reservoir, Carbon County, Utah,* United States Geological Survey Water-Supply Paper 2247.

Ward, A.D. and W.J. Elliot (1995) *Environmental Hydrology*, CRC Press, Inc. pp. 327-334.

Watson, J.T., S.C. Reed, R.H. Kadlec, R.L. Knight and A.E. Whitehouse (1989) Performance expectations and loading rates for constructed wetlands. In: *Constructed Wetlands for Wastewater Treatment: municipal, industrial and agricultural*. D.A. Hammer (ed.) Lewis Publishers, Inc. pp. 319-351.

Welch, E.B. and T. Lindell (1992) *Ecological effects of wastewater: applied limnology and pollutant effects*. Chapman & Hall, London.

Willard, D.E. and Hiller (1990) Wetland dynamics: considerations for restored and created wetlands. In: *Wetland Creation and Restoration, the Status of the Science*. Kusler, J.A. and M.E. Kentula (eds). Island Press, Washington, D.C. pp. 459-466.

Wren, C., C. Bishop and D. Stewart (1996) Potential risk of chemicals in constructed wetlands to fish and wildlife. In: *Proceedings from the Constructed Wetlands in Cold Climates, Design, Operation and Performance Symposium*. Niagara-on-the-Lake, Ontario, June 4-5, 1996.