4.0 The Effects of Nutrient Addition on Ecosystems

Highlights

- Addition of nutrients to an aquatic or terrestrial ecosystem increases the biomass of plants and, ultimately, decreases the number of species. Although ecosystems managed for plant yield, such as agriculture and forestry, reap economic benefits from added nutrients, natural ecosystems generally suffer an undesirable change in plant and animal communities.

- Nutrient addition to lakes causes eutrophication, characterized by the development of algal scum (including blooms of toxic algae), changes in the abundance and composition of aquatic animals including an increase in the numbers of "coarse" fish, declines in oxygen in the lake water, and taste and odour problems of drinking water supplies. Examples of eutrophic lake waters are found in some of the sheltered bays of Lakes Erie and Ontario.

- Nutrient addition to rivers increases the abundance of aquatic weeds or algae. This increased plant growth may cause problems to boaters and clog the screens of intake pipes. Many rivers in Canada show signs of moderate nutrient enrichment downstream of municipal wastewater discharges or areas of intensive agriculture.

- Water quality on both coasts is generally excellent, as the ocean's natural circulation serves to stop accumulation of nutrients from anthropogenic sources. Certain small bays and harbours where oceanic exchange is limited do, however, show evidence of eutrophication, especially on the East Coast where settlement has affected the environment for over 200 years.

- Wetlands are unique ecosystems that straddle the boundary between terrestrial and aquatic habitats. Although the conversion of wetlands to urban and agricultural land has been the major cause of wetland loss, a less obvious cause of undesirable changes in wetland plant communities is nitrogen addition. Cootes Paradise is a freshwater wetland in southern Ontario showing signs of recovery now that nutrient addition from municipal sewage has been reduced.

- Nitrate contamination of ground water is usually due to over-application of fertilizer or manure on agricultural land, domestic waste from septic disposal systems and losses of soil organic nitrogen. Contaminated ground water may contribute to the enrichment of some surface waters in Canada.

Nutrient addition, whether it be the purposeful addition of fertilizers to agricultural fields or the unintentional consequence of waste discharge, promotes plant growth in all ecosystems except those few that are nutrient sufficient. Terrestrial plant production is limited by N supply for much of the Earth's surface, particularly in temperate and boreal regions (Vitousek and Howarth 1991). Agricultural crops are a clear example of production gains to be realized from the addition of N, the dominant ingredient in commercial fertilizers. However, native vegetation is also N limited (i.e., maximum growth is prevented by insufficient N). Consequently, native vegetation responds to N addition by increased biomass if not limited by other factors such as moisture, light, or grazing (Rasmussen et al. 1993; Hudson et al. 1994; Aber et al. 1995; Townsend et al. 1996).

In aquatic ecosystems, the addition of inorganic nitrogen to N-limited systems will also increase plant production to excessive levels, a condition known as eutrophication. Most marine waters, particularly those in temperate regions, are N-limited and N addition as a result of human activity is arguably the greatest threat to the integrity of coastal ecosystems (NRC 1993a; Howarth et al. 1996). In contrast, most lakes tend to be limited by P supply (Dillon and Rigler 1974; Schindler 1974). In the case of river and freshwater wetlands, less information is available about their nutrient status; however, evidence to date suggests these systems may be N or P-limited when little affected by human activity (Morris 1991, Scrimgeour and Chambers 2000). As well as increased plant growth resulting as a direct consequence of nutrient addition to inland and coastal waters, indirect consequences include changes in the abundance and diversity of higher trophic levels (benthic and planktonic fauna), increased abundance of toxic algae, and loss of oxygen from the water. This last effect is due to increased oxygen consumption by the greater abundance of bacteria supported by decomposition of nutrient-enhanced plant production.

The primary consequence of nutrient addition is similar for all ecosystems, namely increased plant growth. However the secondary consequences (e.g., changes in faunal diversity, effects on nutrient cycling, impacts on adjacent ecosystems) differ. This chapter reviews the effects of nutrient addition on forests, lakes, rivers, wetlands, coastal waters and ground waters. Canadian case studies are presented to support the general reviews, and the geographic extent of adverse consequences to nutrient addition is identified.

4.1. Terrestrial Ecosystems

Nitrogen is considered to be the nutrient that most often limits net primary production in forest ecosystems, particularly those in temperate and boreal regions (e.g., Vitousek and Howarth 1991). Fertilization studies involving single N applications have confirmed that N is the major limiting element for tree growth in many Canadian forest ecosystems (Weetman et al. 1987; Morrison and Foster 1990). Not all forests in Canada are N limited, however. Maple-dominated deciduous forests in eastern Canada have not responded to added N, likely because of an abundance of soil N (Cote et al. 1993; I.K. Morrison, Natural Resources Canada, personal communication). However, the general tendency of N limitation in terrestrial ecosystems, together with the documented increase in atmospheric N deposition as a result of human activities, have raised the question as to how much of the carbon stored in terrestrial ecosystems is a consequence of anthropogenic N deposition and fixation.

The primary source of N to forest ecosystems is the atmosphere. Prior to industrial development, N forms taken up by plants were produced only by biological nitrogen fixation (i.e., microbially mediated production of ammonia) and lightning, which results in the production of nitric oxide, NO. Although plants have the capacity to remove nitrogen oxides from the atmosphere (Bennet 1975; WHO 1997a), most N is taken up from the soil where it has been fixed in biologically available forms by N-fixing bacteria.

Human activity has greatly increased emissions of airborne N that, in turn, have led to increased deposition of biologically-available N on the Earth's surface. Experiments conducted in Europe and North America indicate a large portion of this extra anthropogenic N is retained by forest, wetland, and tundra ecosystems and stimulates production, resulting in increased carbon uptake and storage (see Vitousek et al. 1997 for discussion). N deposition can also stimulate microbial decomposition and thus

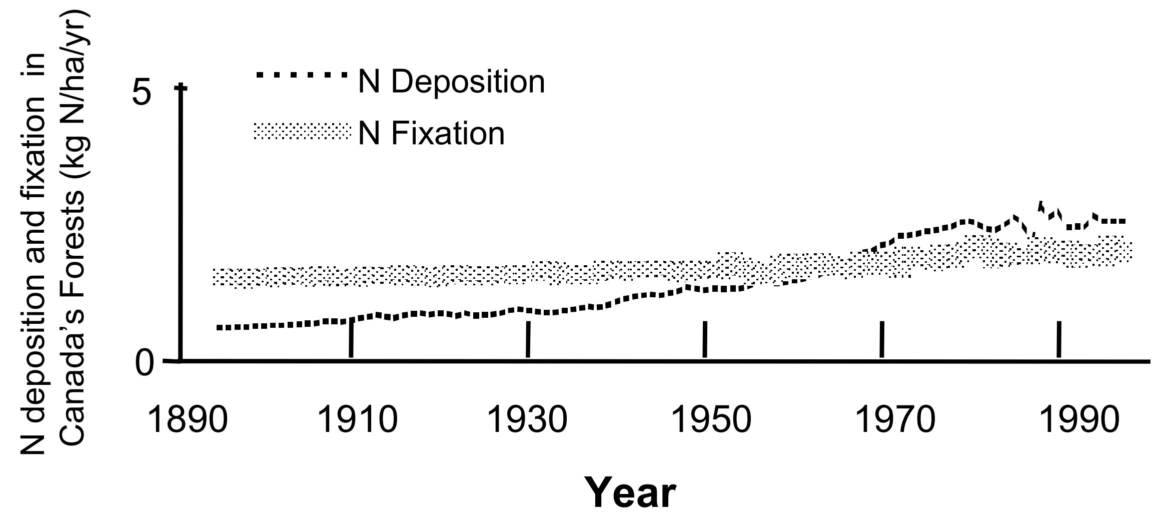

Figure 4.1. Atmospheric N deposition in Canada and N fixation by Canada's forest ecosystems (from Chen et al. 2000 a, b).

the breakdown of any N-enhanced productivity. However, the stimulatory effects of N appear, on average, to out-weigh productivity losses on a global basis. Global estimates of additional plant growth (expressed as carbon storage) resulting from anthropogenic N addition range from $1x10^8$ to $13x10^8$ tonnes of carbon, with higher numbers tending to reflect our better understanding in recent years of the impact of humans on the global N cycle (Peterson and Melillo 1985; Schindler and Bayley 1993; Hudson et al. 1994; Townsend et al. 1996).

In Canada, atmospheric deposition of inorganic N (ammonium plus nitrate) has steadily increased since the 1900s until the 1990s when it was estimated to supply, on average, 2.5 kg/ha/yr of N to forests (Figures 3.14 and 4.1). This value is five-fold greater than pre-industrial values of 0.5 kg/ha/yr N. Because most of Canada's forests are N limited, increased N deposition has likely increased the productivity of forest vegetation. It is still not clear, however, if productivity gains in Canada's forests resulting from increased N deposition, along with a warming climate and elevated CO_2 levels in the atmosphere, have offset the carbon loss caused by timber harvest, fire and insect-induced mortality. Kurtz and Apps (1999) estimated Canadian forest ecosystems were a sink for atmospheric carbon from 1920 to 1980 but became a carbon source in the 1980s as a result of stand-replacing disturbances, largely of natural origin, that reduced the quantity of stored carbon (Figure 4.2b). In contrast, Chen et al. (2000a, b) estimated that the productivity gains resulting from increased N deposition (about 70 million tonnes carbon per year from 1985 to 1998), along with a warming climate and elevated CO_2 levels in the atmosphere, have more than offset the carbon loss caused by timber harvest, fire and insect-induced mortality, so that Canada's forests are now a carbon sink (Figure 4.2a).

In addition to increases in forest growth, N deposition can also change species dominance and diversity by favouring the growth of plants better adapted to high concentrations of N. Experiments involving N addition have resulted in declines in the number of plant species in North American grasslands, European grasslands and European heathlands (e.g., Aerts and Berendse 1988; Bobbink et al. 1988; Tilman 1997). Moreover, the resulting species-poor plant communities experienced greater variations in productivity in response to environmental stress such as interannual climatic variation (Tilman and

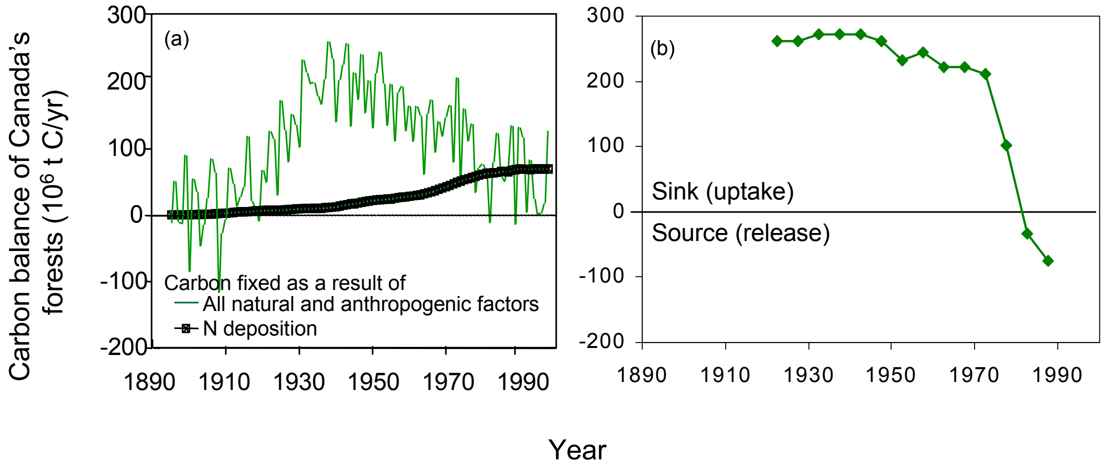

Figure 4.2. Carbon balance for Canada's forest ecosystems: (a) results of Chen et al. (2000a); (b) results of Kurz and Apps (1999).

Downing 1994; Tilman 1996). In Canada, there is little if any conclusive evidence of changes in forest diversity as a result of N addition (Hall 1995).

Although N is the element that most often limits net primary production in terrestrial ecosystems, a point may ultimately be reached where N supply as a result of N deposition is in excess of the total plant and microbial demand for N. This condition is known as forest ecosystem N saturation and is characterized by the conversion of excess N to nitrate, which is then leached to surface or ground waters. In addition to its effects on downstream waters, nitrate leaching has been found to cause changes in soil chemistry, in particular a loss of nutrient cations (such as calcium and magnesium), an increase in soil acidity, and an increase in the availability of aluminum (due to the increased acidity) (see Aber et al. 1998 for review). Leaching of nitrate and associated cations can, in turn, cause nutrient imbalances in trees, in particular changes in calcium:aluminum and magnesium:nitrogen ratios. These imbalances have been linked to reductions in net photosynthesis, photosynthetic N-use efficiency, forest growth, and tree mortality (Schulze 1989; McNulty et al. 1991, 1996; Aber et al. 1995; Cronan and Grigal 1995). Nitrogen imbalances in trees as a result of high levels of N deposition may also increase the likelihood of damage caused by weather extremes, particularly high temperatures in late winter, hard frosts in early spring or prolonged drought (Cote and Ouimet 1996).

Much of the research on forest ecosystem N saturation has been undertaken in Europe and several studies have concluded that the forests of Europe are becoming N saturated. The risk of N saturation in North America is generally lower because most forest ecosystems are N limited and because N deposition is still relatively low (averaging about 2.5 kg/ha/yr in Canada; Figure 4.1). By comparison, forests in some parts of northern Europe are receiving 10 times the natural level of atmospheric N deposition, with the Netherlands receiving the highest rates of atmospheric N deposition in the world (40 to 90 kg/ha/yr; Vitousek et al. 1997). Nevertheless, studies in the USA have shown that certain forests are already N saturated (e.g., Fernow experimental forest, West Virginia; Gilliam et al. 1996) and that other forests could be experimentally N saturated by several years of low-level additions of N (e.g., in a high-elevation spruce-fir stand, Vermont [McNultry et al. 1996]; Bear Brook, Maine [Kahl et al. 1993]; Harvard Forest, Massachusetts [Aber et al. 1993]).

In Canada, an analysis of nitrate concentrations in runoff water to Turkey Lake in central Ontario, Lac Laflamme in southern Québec, Plastic Lake in south-central Ontario, the Experimental Lakes Area in northwestern Ontario, and Kejimkujik Lake in southern Nova Scotia identified the forests surrounding Turkey Lake and Lac Laflamme as being in an early stage of N saturation (Jeffries 1995). The lack of retention of atmospheric N by vegetation (maple-dominated forest) and soils in the Turkey lakes watershed is due to the near-N saturated soils that exhibit high rates of endogenous nitrification (Mitchell et al. 1992). Depletion of base cations from forest soils has also been observed in areas of high sulphate and nitrate deposition (Houle et al. 1997; Markewitz et al. 1998) and predictions based on cumulative loss of base cations indicate forest productivity will decline with continued sulphate and nitrate deposition (Arp and Oja 1992; Arp et al. 1996). The risk of N saturation is greatest in eastern Canada where N deposition rates are highest, with wet deposition being about 10 kg/ha/yr and total deposition (wet + dry deposition) being 13.5 kg/ha/yr (Chen et al. 2000a).

4.2. Lakes

Nutrients are essential to lakes because they provide the raw material for the growth of algae the food sources of zooplankton, which, in turn, are eaten by fish. The concentration of nutrients in a lake is determined by the interplay of the magnitude, timing and bioavailability of the nutrient load, the rate of water supply compared to the volume of the lake (flushing time), and the depth of the lake. Lakes have a capacity to absorb nutrients but when this level is exceeded concentrations rise and obnoxious growths of algae and other effects occur.

Phosphorus in lakes is highly reactive, binding with soil particles and minerals in the water column. Once this binding has occurred the P is largely unavailable to bacteria and algae for growth. It is the amount and rate of turnover of the remaining dissolved phosphorus that dictates the extent of algal growth. The ability of a lake to retain P depends on net sedimentation, which in turn depends on the flushing rate in the lake as well as the depth. Rapidly flushed lakes reflect the concentration of the inflowing waters. In contrast, the P concentration of a lake that flushes very slowly is largely determined by its sedimentation characteristics (Vollenweider 1976). However, even in lakes with high nutrient retention and low average concentrations, there can be nutrient problems. These problems occur because nutrients are loaded in the nearshore via sewage outfalls and rivers where dilution due to the physical movements of water in lakes is minimal. Although dilution is sometimes viewed as an undesirable solution to pollution, dilution is a key part of the nutrient assimilation capacity of lakes where effluents may have 10 to 100 times the P concentration desired in the ambient water. The concentration of nutrients is the crucial factor in nutrient pollution effects and dilution is critical to amelioration of pollution effects.

Nutrient supply strongly affects the growth of aquatic plants in lakes, particularly algae. In the majority of North temperate lakes, P is the nutrient most in demand, and algal growth and biomass is P-limited (Vollenweider 1976). Thus, in many lakes, algal growth is a function of P concentration (e.g., Watson et al. 1997). However, in some lakes, primary production is dominated by macrophytes, which tend to suppress algae, resulting in clear water even at high nutrient loads (Beklioglu and Moss 1996; Weisner et al. 1997). Lakes can be classified based on their nutrient or "trophic" status. Oligotrophic lakes have very low nutrient concentrations (typically less than 5 μg/L average total P). In these systems, algae are nutrient-limited and low in abundance; hence, the water is typically clear. Oligotrophic water can, however be highly coloured by humic matter. Moderately productive lakes (about 10-30 μg/L total P)

Lake Erie, Ontario: Recovery of a "Dead" Lake

Lake Erie is one of the most widely recognized examples of restoration of an aquatic ecosystem damaged by excessive nutrient loading. The damage began in the 1800s when land clearing for agriculture and settlement increased soil erosion and thus P loading to the lake. For one hundred years, development continued with accelerations occurring in the periods of the World Wars. Dramatic changes began in the 1940s as more and more people were connected to sewage systems that discharged to the lake and detergents with high P content came into use.

Concern about the persistent foam from detergents, the increasing degradation of visible water quality, and increasing general concern for the environment led to scientific studies of Lake Erie by federal government agencies. Surprisingly, there had been few previous studies. Investigations by U.S. agencies and Canadian universities in the late 1920s, early 1950s and early 1960s raised concerns about water pollution and disappearance of fish. In 1970, a bi-national study of the lake confirmed that there had been an increase in nutrient concentrations in the lake and that nuisance algae were appearing. Filamentous algae (*Cladophora*) washed up on shores in great quantities. Excessive algal populations sunk to the bottom where decomposition occurred such that the bottom waters of the central basin were depleted of oxygen by late summer; this caused regeneration of P from the sediments. There was a fear that the P regeneration would lead to an autofertilization condition (Burns 1976).

Results of modelling exercises to predict a safe P load were startling. Annual P loads needed to be reduced from a high of about 28 000 tonnes to about 11 000 tonnes to achieve reasonable algal abundance and restore year-round aerobic conditions in the hypolimnion of the Central Basin. Governments were deeply concerned by the evidence that the huge Great Lakes system could be so affected by anthropogenic nutrient loads. Eventually, the Canada U.S. Great Lakes Water Quality Agreement (GLWQA) was signed in 1972. The agreement incorporated the scientific advice to decrease the P load to 11 000 tonnes per year.

Four main strategies were employed to reduce the P load: (1) P in household laundry detergents (which accounted for about 25% of the P in sewage) was rapidly phased out; (2) sewage treatment plants (MWTPs) were constructed where none existed and primary treatment plants were upgraded to secondary treatment; (3) P precipitation by addition of alum, ferric chloride or other agents was employed to enable all MWTPs handling more than 3.8 million litres per day (one million U.S. gallons) to decrease effluent P to 1 mg P/L or less; (4) non-point source P from urban runoff and agriculture was addressed with ongoing changes in farm practices that reduce runoff and erosion. Although an accurate estimate of the expenditures is not possible, it is accepted that on the order of $15 billion U.S. was spent on the various infrastructure construction and improvement projects.

The results provide a dramatic example of how concerted action can reverse environmental damage. The load of total P to Lake Erie was reduced by more than 50% by the mid-1980s and now oscillates around the recommended 11 000 tonnes annually (Figure 4.3). Much of the damaging P load came from cities discharging sewage into the West Basin of the lake, which is where the main repair was effected as the P concentration in the water decreased from 40 µg P/L to about 20 µg P/L. The P response in the Central Basin offshore has been more modest because the basin is larger and nutrient loads were relatively smaller as the majority of controllable municipal sewage discharges are located in the West Basin.

About the time the P reduction goals were achieved, the zebra mussel (*Driessena polymorpha*) invaded the lake. Prodigious filter feeders, the mussels remove enormous quantities of plankton from the water column and deposit their waste to the sediments resulting in P reductions in the water and improved water quality. Because the mussels can have effects on water quality similar to nutrient reduction and lakes take some time to equilibrate, it is important to distinguish the effects of the load reductions from those of the mussels. Comparison of water quality over the period of nutrient controls showed that P concentrations were about 20% lower in the East and Central Basins in 1988 than in 1968 (Table 4.1). This decrease corresponded to about a 50% reduction in phytoplankton abundance (measured as chlorophyll *a* concentration) and a 50% improvement in water clarity (measured as Secchi disc transparency). Nitrate increased by more than 300% over the same period due to fewer algae to assimilate N and continued discharge of sewage N and agricultural fertilizers. In comparison to the period of nutrient controls, changes following the introduction of zebra mussels have been smaller except for P (Table 4.1). Although P declined to a record low concentration in 1995, there has been a tendency in the last few years (1996-1999) for a recovery back to pre-mussel concentrations. Data are less available for the West Basin and there was much variability due to resuspension of sediments; however, chlorophyll a declined by 76% by 1988 and declined by a further 34% by 1994/95/96. Thus, we do not yet know what the long-term impact of the mussels will be or even if the impact will be stable (Charlton et al. 1999). The mussels have caused a marked increase in

Lake Erie concluded

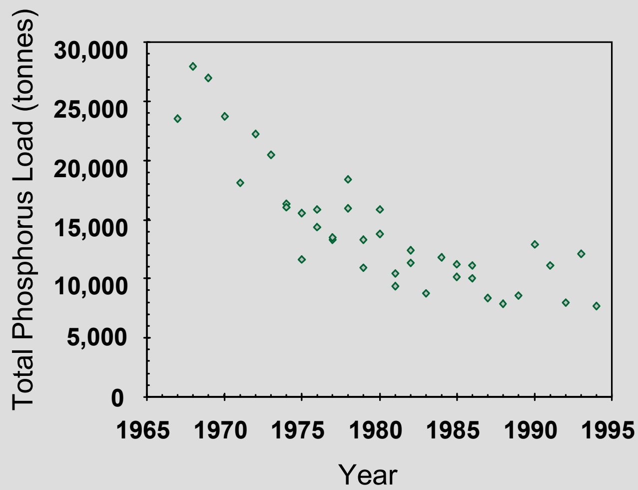

Figure 4.3. Phosphorus loading history of Lake Erie, 1967 to 1994. (Data sources: Fraser 1987; Lesht et al. 1991; D.M. Dolan, International Joint Commission, personal communication).

water clarity in nearshore areas especially in the west basin of the lake (116%).

Despite the success of nutrient controls in lowering phytoplankton abundance, the problem with filamentous algae, (*Cladophora*) has been reduced but not eliminated. Treated sewage contains P at 100 times the desired concentration of the lake water. In mixing zones near sewage outfalls and in nearshore areas receiving flows from rivers, P concentrations are sufficiently high to cause nuisance accumulations of *Cladophora*. Indeed, P concentrations in most Ontario tributaries exceed the provincial guideline of 0.030 mg P/L. Thus, even though the offshore water is of good quality, the conditions in specific locations can still be improved. Now with the advent of the mussel invasion there is some evidence that the excretion of the mussels helps supply nutrients to the attached *Cladophora* algae.

Table 4.1. Percentage change in phosphorus (P), nitrate (N), chlorophyll *a* (Chl*a*), and Secchi depth (Secchi) in the East and Central Basins of Lake Erie during the period of phosphorus reductions (1968 to 1988) and after the zebra mussel invasion (1988 to 1994/95/96).

Basin	1968 to 1988				1988 to 1994/95/96			
	P	N	Chl*a*	Secchi	P	N	Chl*a*	Secchi
East	-19	+325	-43	+48	-33	+53	-25*	+8*
Central	-23	+340	-54	+59	-23	+5*	-17	-5*

* Change is not statistically significant

The success of the nutrient controls in ameliorating the oxygen depletion of the Central Basin hypolimnion has been more equivocal. Beginning in 1993 and until 1997, oxygen concentrations in the hypolimnion were much higher than usual, raising hopes of a recovery in deep-water oxygen (Charlton 1994). However, in 1998, oxygen concentrations throughout most of the Central Basin were less than 1 mg/L oxygen by the end of August, typical of much earlier years. The situation continues to be a subject of research and monitoring.

Despite the inertia of the oxygen situation, the GLWQA nutrient controls have been a success. It is likely that the lake never fully equilibrated with the brief period of peak nutrient loadings so that damage was averted as well as reversed by the substantial reduction in P loads. Further reduction in loads will come with better land use practices; non-point sources now account for about half the total P load. Dealing with human sewage is a perpetual management problem. Maintenance of the hard won gains will be important to preserve the lake. Phosphorus removal at sewage plants increases operating expenses and populations continue to grow. Thus, it is important to maintain monitoring of loads and the performance of sewage plants.

Murray Charlton
Environment Canada

are said to be mesotrophic (see Heney Lake case study), whereas eutrophic lakes are very rich in nutrients, such that the water is green with algae throughout most of the growing season (see Qu'Appelle Lakes case study). At very high nutrient concentrations (generally about 100 µg/L total P), algal growth is independent of P concentration and becomes limited by other factors such as N, micronutrients (e.g., iron) or light, due to self-shading by high algal densities. Lakes with extremely high nutrient concentrations and excessive algal growth (such that the water can resemble pea soup) are termed hyper-eutrophic.

There are many deleterious effects of algal blooms. Initially, the most noticeable effects are increased water turbidity, reduced aesthetic appeal and even decreased recreational use, as visibility becomes insufficient for swimming. As enrichment continues, erratic and sometimes severe outbreaks of different algal taxa (e.g. Chrysophyta, Haptophyta, Dinophyta) can not only negatively affect water quality and the aquatic food web, but can cause fish kills (e.g., Adachi 1965; Kamiya et al 1979; Hansen et al. 1994; Hallegraeff et al. 1995; Leeper and Porter 1995), particularly, it seems, in acidified systems (Mills et al. 1995). Highly eutrophic systems tend to be predominated by blue-green algae that form dense, foul-smelling and noxious blooms, often as surface scums, because many of these algae are buoyant. These algae can even develop massive blooms under ice-cover, as they are highly efficient at capturing light. Many species of blue-green algae produce potent toxins, which can poison zooplankton, fish, avian waterfowl, terrestrial wildlife, livestock and even humans (Carmichael 1992; Kotak et al 1993a, 1994). Paradoxically, some blue-green algae may at times give the impression of greater water clarity, because their cells are amassed into macroscopic colonies, allowing more light penetration than a similar biomass of individual algal cells (Vollenweider 1976). More frequently, however, surface blooms of blue-green algae can shade out other algae or aquatic plants. Blue-green algae may have other negative effects on aquatic food webs, because they are too large for ingestion by most grazers. This effectively provides them with an escape from predation, and facilitates their growth and dominance, as grazers selectively remove smaller and less noxious algal species (Watson and McCauley 1988).

High algal densities increase costs for drinking-water filtration, and are associated with taste and odour problems. The decline and senescence of algal blooms can lower oxygen concentrations and generate toxic levels of ammonium, thereby shrinking the strata of water habitable by fish. On shorelines with sufficient rocky substrates and P concentrations, attached filamentous algae such as *Cladophora* can produce dense, long filamentous strands that crowd out other species, and provide surface area and nutrients for the growth of odourous mats of other algae and bacteria. These filaments break off during late summer storms and are concentrated on shores by wind action, sometimes to a thickness of over one metre, and extending a substantial distance from shore. This action not only precludes swimming and other forms of recreation, but subsequent decay generates odour and noxious ammonia levels, which may render adjacent property unusable and without value.

Algae (as well as bacteria such as actinomycetes) can cause widespread taste and odour in drinking water supplies. For example, many blue-green algae and actinomycetes produce the compounds geosmin (trans,trans-1,10-dimethyl-9-decalol) and MIB (2-methylisoborneol) which cause musty, earthy odours and tastes in drinking water (Jensen et al. 1994; Jüttner 1995). These two potent volatile organic compounds (VOCs) do not respond to conventional water treatment and their removal costs the water industry millions of dollars per year. Traditionally these VOCs have been associated with intense growths of nuisance algae in highly eutrophic lakes (Brownlee et al.1984; Kenefick et al. 1992; Watson

Qu'Appelle Lakes, Saskatchewan: Effects of Human Activity on Naturally Eutrophic Lakes

Lakes in the Prairies and Boreal Plains ecozones of southern Alberta, Saskatchewan and Manitoba lie in large, fertile agricultural catchments that supply high concentrations of nutrients. Eutrophication is generally the single most important water quality issue (Government of Canada 1996; Hall et al. 1999). Water quality in prairie lakes is controlled by the interplay of climate, resource use, and urban factors. The lakes of the Qu'Appelle Valley in south-central Saskatchewan are a good example of the combined effects of agricultural activity, human settlement, and climate on water quality.

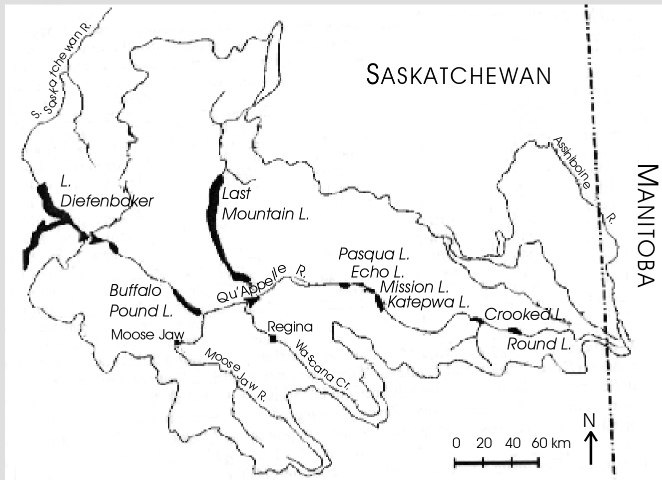

Figure 4.4. Qu'Appelle Valley drainage basin and lakes. Adapted from Hall et al. 1999.

The Qu'Appelle Valley extends over 400 km from its headwaters near Lake Diefenbaker in western Saskatchewan to its confluence with the Assiniboine River in western Manitoba (Figure 4.4). Its 52 000 km^2 basin is drained by the Qu'Appelle River and it, or its tributaries, provides water to approximately one-third of Saskatchewan's population, including the cities of Regina and Moose Jaw. Agricultural fields and pastures comprise more than 95% of land use in the drainage basin (Hall et al. 1999). A chain of eight lakes, including two headwater reservoirs (Lake Diefenbaker and Buffalo Pound Lake), a central group of four 'Fishing Lakes' (Pasqua, Echo, Mission and Katepwa lakes), and two eastern basins (Crooked and Round lakes) form a trophic status gradient in the valley with nutrient concentrations in the lakes increasing from the western to the eastern end of the chain. A ninth lake, Last Mountain Lake, drains into the Qu'Appelle River via Last Mountain Creek. These lakes represent a major recreational and economically-valuable resource for southern Saskatchewan as they are used for commercial (whitefish, cisco and buffalofish) and game (northern pike, walleye, yellow perch) fishing, recreation, irrigation, livestock watering, drinking water supply and sewage discharge, in addition to serving as flood control and waterfowl habitat (Munroe 1986, Chambers 1989).

The lakes of the Qu'Appelle Valley are glacial outwash features (Hall et al. 1999). Typical of this region, the lakes are shallow (mean depth 3.0-14.3 m), hypereutrophic (TP>300 μg/L) and produce immense blooms of blue-green algae throughout the summer (Munroe 1986; Kenney 1990; Hall et al. 1999). Their water quality is modestly influenced by annual fluctuations in precipitation and evaporation rates during the hot, dry summer months. Depending on the amount of rainfall in a given year, their salinity fluctuates between slightly saline (0.5 - 1.0 g/L total dissolved solids (TDS)) and moderately saline (1.0 - 5.0 g/L TDS). Lake Diefenbaker and Buffalo Pound Lake contain the freshest water (TDS < 0.5 g/L) whereas Last Mountain Lake is the most saline with TDS concentrations regularly exceeding 1.5 g/L (Chambers 1989; Hall et al. 1999).

Although the lakes are naturally eutrophic, present water quality is worse than before permanent settlement and intensive agricultural development of the region (Allan et al. 1980; Hall et al. 1999). Growing concern over the deterioration of water quality in the Qu'Appelle lakes during the last 30 years resulted in several Federal-Provincial studies that attributed excessive algal and plant growth to high nutrient concentrations in agricultural runoff and municipal wastewater discharge. At that time (1972), it was estimated that 70% of the P and N entering the river basin came from sewage discharge from

Qu'Appelle Lakes concluded

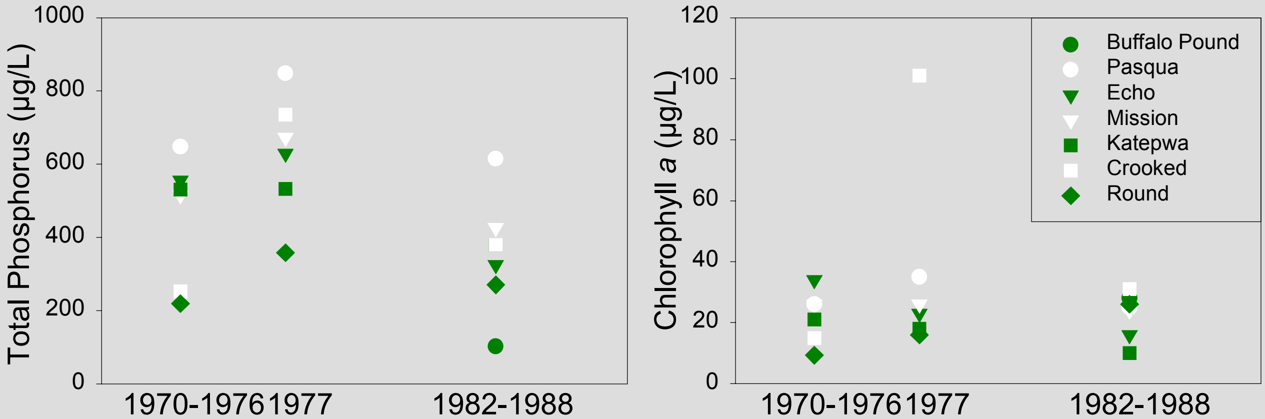

Figure 4.5. Total phosphorus and phytoplankton chlorophyll *a* concentrations for the Qu'Appelle Lakes before (1970-76) and after (1977; 1982-1989) upgrades at the Regina wastewater treatment plant. (Data from Chambers 1989).

Regina and Moose Jaw's wastewater treatment plants (Munroe 1986). As a result, the 1975 Qu'Appelle Agreement between the Governments of Canada and Saskatchewan set out to protect and improve the Qu'Appelle Valley's environment, resources and cultural heritage, and to promote economic growth in the region by developing tourism and recreational activities (Chambers 1989). Regina upgraded its sewage treatment facility to tertiary treatment in 1976 and Moose Jaw diverted all of its sewage to agricultural land through the use of spray irrigation by 1987 (Chambers 1989).

Tertiary wastewater treatment has not resulted in a clear improvement in water quality in the Qu'Appelle lakes. Comparing data collected before and after upgrades to Regina's wastewater treatment plant, open-water total P concentrations have decreased; however, chlorophyll *a* levels have not changed (Figure 4.5; Chambers 1989). Hall et al. (1999) also noted no clear improvement in water quality following sewage treatment despite a three-fold decrease in P loading. However, paleolimnological studies of environmental factors related to long term changes in water quality in the Qu'Appelle Lakes have identified that algal community composition is more variable in the years following tertiary treatment of Regina's sewage (ca. 1977-1990) as compared to a preceding era of high algal production (ca. 1925-1960; Dixit et al. 2000).

N loading to the Qu'Appelle lakes is currently at an all-time maximum (Hall et al. 1999). Nutrient enrichment bioassays (Graham 1997), analysis of elemental ratios (TN:TP; Munroe 1986) and the presence of N_2-fixing blue-green algae (Graham 1997) suggest primary production in the Qu'Appelle Valley is controlled by the influx of N, not simply P. Hall et al. (1999) concluded future management strategies must take into account that water quality in the upstream lakes is influenced by urban activities whereas downstream lakes are more affected by resource use (cropland area and livestock biomass).

Martha Guy
Environment Canada

et al. 1999), but more recently, they have been problematic in systems that have undergone nutrient abatement. High levels of these compounds were detected in water from Lake Erie, the Niagara River, eastern Lake Ontario and the upper St. Lawrence River (Lange and Wittmeyer 1997; Ridal et al. 1999; Watson et al. 2000). Odour outbreaks have increased in the Great Lakes and St. Lawrence River system coincident with the invasion of zebra mussels and it has been postulated that the greater frequency of these problems is due to increased benthic algal abundance caused by greater water

Lac Heney, Québec: Impacts of Aquaculture (1993-1998)

In November 1998, the Québec government closed a fish farm that had opened in 1993 on a tributary of Lac Heney, Québec (Bird and Mesnage 1996). In five years of operation, the fish farm's effluent was implicated in the near doubling of the lake's P loads and the associated dramatic decrease in water transparency and oxygen concentrations and increase in algal concentrations. The Québec Ministry of the Environment and Wildlife reported the fish farm's nutrient-rich wastewater added 400 kg of P to the lake annually via the lake's tributary. The wastewater's high nutrient concentration was attributed to the presence of waste food pellets in the effluent.

Lac Heney, located in the Outaouais region of Québec, supports a valuable trout fishery. The lake is deep (30- 33 m maximum depth) with an area of 12.4 km^2. Concentrations of total P in the surface water varied from 10 to 30 µg/L during the summer of 1995. During the period of spring overturn in 1996, P concentrations of 21 µg/L remained uniform in the water column. The concentration of P in Lac Heney classes it as mesotrophic or moderately nutrient rich.

Elevated P concentrations as a result of the fish farm increased the abundance of planktonic algae as well as macrophytes and filamentous algae in the lake (Bird and Mesnage 1996). In 1996, planktonic algal abundance (expressed as concentration of chlorophyll *a*) was fairly low and ranged from 1 to 3 µg/L with concentrations peaking at 14 µg/L in August. The dissolved oxygen profile was typical of a mesotrophic lake whereby concentrations were lower in the deep (hypolimnetic) waters. The result was that the waters of the hypolimnion may have been undesirable for the health of lake trout for most of July because the concentration of oxygen was below 6 mg/L (Prairie 1994). The lowered oxygen levels may also have had a long-term effect by reducing the growth and reproductive rates of sensitive species like lake trout. In addition, the growth of macrophytes was reported to destroy the spawning sites of lake trout.

To create a suitable lake-specific remediation plan for Lac Heney, it was necessary to determine the contribution of P from the fish farm relative to other sources (e.g., atmosphere, cottages, cultivated land in the basin, wastewater inputs, and lake sediments). Sampling undertaken during 1995 and 1996 showed that the quantity of total P entering the lake from all tributaries, including the aquaculture farm was 762 kg (Table 4.2). The aquaculture farm alone contributed 450 kg to this total. The quantity lost from the lake was 231 kg P. The fraction of the contribution that remained in the lake (i.e., the coefficient of retention) was therefore 0.81. Clearly, the fish farm was a major contributor of P to the lake and was responsible for the rapid deterioration in water quality.

Closure of the fish farm is the remediation strategy that the Québec government and the Association for the Protection of Lac Heney has undertaken to reduce P levels and hopefully to avoid further deterioration in the water quality of Lac Heney.

Table 4.2. Calculation of the total phosphorus (TP) balance in Lac Heney, Québec. Adapted from Bird and Mesnage (1996).

Components of the Balance		TP (kg)
Tributaries (excluding fish farm)		312
Aquaculture farm	+	450
Atmosphere	+	186
Cottage septic systems	+	195
TOTAL external contributions	=	1143
Loss of P from the lake	-	231
Quantity of P remaining in the lake	=	912

Christian Gagnon
Environment Canada

clarity and/or to nutrient release from the mussels fecal material (Lange and Wittmeyer 1997; Vogel et al. 1997). However, blue-green algae are not the only algal group that can cause taste and odour problems. Even nutrient-poor lakes and reservoirs may experience intermittent but severe taste and odour, often due to the production of fishy-smelling fatty acid derivatives by chrysophytes (Nicholls 1995; Watson et al. 1999). Public reaction to drinking water described as having an earthy, swampy or fishy taste has led to concern about the safety of the water supply and the implementation of various treatments to mitigate the problem, which have varying degrees of success (Kim et al. 1997; Chow et al. 1998; Satchwill et al. 2000). It should be noted, however, that these odour compounds are not toxic to most organisms, although at excessive levels, geosmin may affect some aquatic juvenile forms (trout, urchins) whereas fatty acid derivatives can be toxic to micrograzers (Nakajima et al. 1996; Gagne et al. 1999; Jüttner 1999).

Algae produce organic matter that forms the base of the food chain, but excessive algal populations have deleterious ecological effects. In the case of nutrient-poor lakes with low production of organic matter, artificial fertilization may be undertaken to provide added nutrients and stimulate plant growth. For example, N and P addition to oligotrophic British Columbia lakes has increased algal and zooplankton abundance and, in turn, increased the size of spawning and juvenile salmon (see Section 3.4). However, in eutrophic lakes, zooplankton go through boom and bust cycles and diversity may be reduced and production limited by blue-green algae blooms. So called “coarse” fish species, such as carp, may come to dominate the fish population while success of spawning and survival of more desirable species, such as trout or bass, becomes more and more tenuous due to the overwhelming amount of organic detritus and periodic de-oxygenation as algal blooms decompose. The result is that eutrophication tends to simplify the aquatic food web. In addition, the overabundance of organic matter caused by excessive algal populations can smother the lake bottom resulting in reduced biodiversity of benthic organisms. Organic matter also accumulates in crevices between rocks where decay can consume enough oxygen to damage the survival of fish eggs. As organic matter accumulates in sediments nuisance accumulations of aquatic rooted plants are more likely to occur. The plants can absorb nutrients from sediments and then release them to the water column either through excretion from the leaves and shoots or by decomposition when the plants die.

The concentration of oxygen in lakes is also related to aquatic plant abundance. Aquatic plants produce oxygen in the daytime as a result of photosynthesis and consume it at night when photosynthesis has ceased and only respiration is occurring. Large populations of aquatic plants can increase oxygen concentrations well above saturation so that the water can effervesce with oxygen bubbles. At night the plants take up oxygen such that oxygen levels may fall well below the saturation concentration (i.e., the maximum concentration of oxygen for its temperature and barometric pressure) thereby causing oxygen stress to fish and invertebrates (Vollenweider 1976). The generally accepted notion that algae cause low oxygen in lakes oversimplifies the situation. Although oxygen may be high during daylight hours, the oxygen decline at night caused by plant uptake can be offset largely by diffusion from the atmosphere into the lake if oxygen concentrations sink below saturation values.

Another way that algae can affect oxygen is by a phenomenon termed “hypolimnetic oxygen depletion”. In early summer, solar radiation warms the upper waters of a lake. Although the solar warming extends to a depth of several metres, the deeper waters remain cold. This cold water is denser and therefore does not mix easily with the warmer surface waters. Until the upper layer cools in autumn, the deeper cold water (or hypolimnion) has no direct contact with the air and thus, oxygen consumed there cannot be replaced by diffusion from the air. The rate of oxygen depletion in the hypolimnion depends on the temperature and thickness of the hypolimnion, respiration by bacteria (with the abundance of bacteria determined by the quantity of organic matter settling down from upper warm waters), and respiration by other biota in the hypolimnetic water and sediments (Vollenweider 1976). In lakes severely polluted by raw or treated sewage, the organic matter from sewage plants may be more than sufficient to drive the oxygen consumption. However, in most lakes, excessive algal growth provides the organic matter that bacteria in the lake hypolimnia use for their growth and which, in turn, results in consumption of oxygen (see Lake Erie case study).

Oxygen consumption in a lake's hypolimnion can reduce the oxygen concentration to zero. When this happens, most of the biota are driven from the hypolimnion water and sediment. Those organisms that cannot move fast enough are killed outright. Under less severe circumstances, chronic conditions of low oxygen (5 mg/L) in the hypolimnion can result in a shift in fish composition from cold-water species with requirements for high concentrations of dissolved oxygen (e.g., salmonids and sculpins) to warm-water species (e.g., walleye, pike and smallmouth bass) with lower oxygen requirements. Sometimes, due to wind, the layers slosh around in the lake. Under these conditions there may be fish kills in surface waters when the deoxygenated bottom waters move into lake areas normally overlain by surface water. Oxygen depletion, however, is a natural phenomenon that is exacerbated by anthropogenic pollution. Therefore, the mere occurrence of oxygen depletion in a hypolimnion is not definite proof of a water pollution problem. Rather, the extent and duration of oxygen depletion can be compared to models that take all factors into account to diagnose whether there are excessive sources of organic matter that may indicate pollution sources (Charlton 1980).

Under conditions of low (hypoxia) or no (anoxia) oxygen in the hypolimnion, P can be released from the bottom sediments as phosphate due to the chemical reduction of iron-phosphate complexes. The duration of P regeneration depends on the duration of anoxia and this depends on the amount of organic matter settling down into the hypolimnion. The concern with P regeneration is that it may lead to an in-lake fertilization process (or internal P loading) whereby P from the bottom sediments diffuses upward into the surface water and provides more P for even more algal production. In shallow lakes, decades of pollution can cause release of P from bottom sediments even under oxygenated conditions, slowing the lake's recovery despite reduction of external sources. Water quality in the Bay of Quinte, Lake Ontario responded positively to an 80% reduction in P load despite a strong summertime sediment phosphorus release (Nicholls 1999). Some of the regenerated P is swept from the system by increased river flows in the fall. Thus, internal phosphorus loading may retard but not prevent recovery of lakes that have sufficient flushing.

4.3. Rivers

Until recently, studies on the effects of nutrient addition to aquatic systems focused largely on lakes and reservoirs with comparatively little work conducted on streams and rivers. The relatively little attention originally directed to assessing the impact of nutrient addition on riverine biota likely relates to early beliefs that rivers were conduits for pollutants and could assimilate any loading of nutrients and organic waste. This thought may have been true before the rapid industrialization and acceleration in population growth of this century that resulted in severe degradation of many large rivers (e.g., Thames, England; Rhine, Europe; Elbe, Czechoslovakia/Germany; Cuyahoga, Ohio, U.S.A.). As a result of these observations of gross pollution as well as other observations of increased productivity and/or changes in diversity downstream of point or diffuse sources of nutrient loading, many investigators have examined nutrient dynamics and the response of river communities to nutrient inputs.

Nutrient cycling in rivers involves the uptake and release of N and P by aquatic biota as well as the downstream transport of these elements. In most ecosystems, nutrients largely cycle in place (e.g., between the sediments and water in the case of lakes), and the cycle is often depicted as a circle whereby nutrients move from one compartment (or "pool") to another, eventually returning to the original pool. In rivers, however, this circular process is stretched into a spiral to account for downstream transport. Thus, biological uptake of nutrients during cellular processes, their release during tissue decomposition, and their downstream transport and subsequent uptake by organisms further downstream are referred to as "spiralling" (Webster and Patten 1979; Newbold et al. 1981; Paul and Duthie 1988; Mulholland et al. 1990). The downstream distance that a given phosphate or nitrate ion can travel when it is released in the water column until its uptake and incorporation in biota depends on the degree to which the growth of stream microbes is limited by nutrient availability. Rapid uptake and short transport distances are expected for rivers where plant growth is nutrient limited, whereas N and P will be transported downstream with only minor losses in rivers where factors other than nutrients determine plant abundance.

Besides the amount of nutrients added, the duration of nutrient loading is also important. Pulse events, in which nutrients are added for a limited or defined duration (e.g., a wastewater spill) and are diluted to non-toxic concentrations, often have little effect on an ecosystem in terms of enrichment, and recovery is rapid. Pulse events, however, can be extremely serious when concentrations are sufficiently great so as to be toxic. In contrast, sustained events (e.g., the addition of a continuous effluent discharge for months or years) are generally not toxic but can cause habitat alteration and changes in abundance and composition of riverine plants and animals, depending upon the river nutrient concentrations that result from the sustained event.

Whereas many rivers that have been sites of intensive agriculture and human settlement for many years have become eutrophic or even grossly polluted, rivers receiving moderate additions of nutrients from sewage or agricultural waste (i.e., the pollutant loading is not so severe as to cause de-oxygenation) have shown increased biological productivity (see Alberta Northern Rivers case study). Dose-response curves have been developed relating nutrient concentrations to benthic algae ("periphyton") growth. These show that growth saturation of periphyton occurs at very low concentrations, only 1 to 5 µg/L soluble reactive P (Bothwell 1985; Bothwell et al. 1989; Chambers 1996) and 10 to 15 µg/L N (Bothwell 1992). In large rivers, where the dominant primary producers are the phytoplankton, studies have also shown that nutrient addition promotes growth. For example, surveys conducted in southern Ontario and western Québec have found that phytoplankton biomass (expressed as chlorophyll *a* content) is positively correlated with the concentration of total P in the water (Basu and Pick 1996). Thus, small additions of nutrients can result in substantial increases in plant growth.

Increased primary productivity as a result of nutrient loading may, in turn, increase the growth and abundance of higher trophic levels. In the Kuparuk River, a small stream in Alaska, a large increase in periphyton biomass due to P addition increased the growth rate of the four most abundant large insects in the enriched section of the stream (Peterson et al. 1993). Along with the stimulation of invertebrate

Alberta's Northern Rivers: Impacts of Pulp Mills and Municipal Sewage

Expansion of the forestry industry in the boreal forest of western Canada during the past 20 years has raised concerns about the effect of pulp mill effluents on riverine nutrient regimes and the potential for eutrophication. Many pulp mills in western Canada were designed or recently upgraded to address concerns relating to acute toxicity caused by organic contaminants or low dissolved oxygen concentrations. In contrast, the consequences of nutrient loading have not been a major consideration for the management of pulp mill wastes.

The Northern River Basins Study (NRBS) was initiated in 1992 and ran until 1996. The aim of the study was to obtain further scientific information on the existing conditions in, and the effects of development on, the aquatic ecosystems of the Athabasca, Wapiti, Smoky and Peace rivers in northern Alberta (Figure 4.6). The ultimate goal was to provide a basis for wise management of the basins' natural resources. Of the eight major NRBS themes, one focused on assessing the effects of pulp mill and municipal effluents on the nutrient status and potential for eutrophication of the northern rivers. The Athabasca, Wapiti-Smoky and Peace rivers all arise in the mountains. Their basins are sparsely populated relative to their size and largely forested, although agricultural land comprises 17% the Wapiti-Smoky river basin. The waters of the four rivers are intrinsically nutrient-poor: annual median concentrations near the headwaters are < 5 µg/L total P and < 300 µg/L total N.

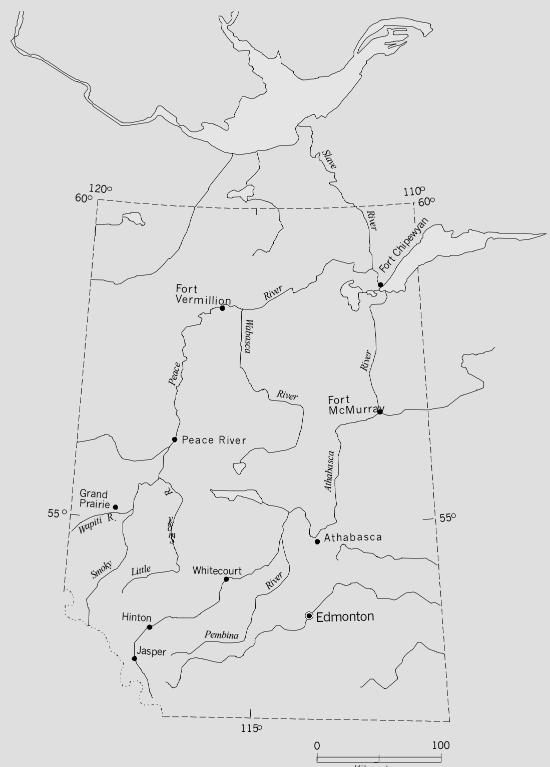

Figure 4.6. The Athabasca, Wapiti, Smoky and Peace Rivers in northern Alberta.

Research and monitoring studies were undertaken as part of the NRBS to characterize nutrient loading from all point and non-point sources, evaluate the impacts of nutrient loading on river chemistry, assess the response of riverine biota to nutrient loading from pulp mill and municipal effluents *in situ*, quantify nutrient responses of benthic biota, and investigate interactions between nutrients and contaminants in pulp mill effluent on food webs (Chambers 1996, Chambers et al. 2000). Results of these studies showed that elevated N and P concentrations occurred in the Athabasca River downstream of Jasper, Hinton, Whitecourt and Fort McMurray and on the Wapiti River downstream of Grande Prairie during fall, winter and spring. In the Athabasca River, 20% of all total phosphorus (TP) samples and 2% of all total nitrogen (TN) samples collected from 1980 to 1993 exceeded the Alberta Surface Water Quality Objective of 0.05 mg/L TP as P and 1.0 mg/L TN as N. Most of these exceedances occurred during summer and were likely due to high particulate concentrations associated with high river discharge. In the Wapiti River, 74% of TP samples and 19% of TN

Northern Rivers, Alberta concluded

samples collected near the mouth from 1991 to 1993 exceeded the Alberta Surface Water Quality Objectives compared with exceedances of only 12% for TP and 0% for TN upstream of Grande Prairie. This observation suggested that nutrients from the City of Grande Prairie and from Weyerhaeuser of Canada Ltd.'s pulp mill effluents contributed to non-compliance.

Continuously-discharging industrial and municipal sources were found to contribute 4 to 10% of the TN load and 6 to 16% of the TP load in the Athabasca River on an annual basis, with the contribution being higher during winter. Likewise, continuously discharging industrial and municipal sources contributed 20% of the TN and 22% of the TP load in the Wapiti River annually. For the Peace River main stem there was no evidence of nutrient impacts.

Elevated nutrient concentrations in the Athabasca and Wapiti rivers increased periphyton biomass and benthic invertebrate densities and, for the Athabasca River downstream of Hinton, increased the length and body weight of spoonhead sculpin (*Cottus ricei*), a small insectivorous fish species. The increase in periphyton biomass and benthic invertebrate densities downstream of effluent outfalls and, in the case of the benthic invertebrates, no loss of species suggest that the response to effluents is one of nutrient enrichment not toxicity. Studies conducted in artificial streams further showed that periphyton biomass and growth of several mayflies, stoneflies and caddisflies increased in response to nutrient or 1% effluent addition, with no significant difference between the two treatments. These results confirmed that the response to the current level of effluent loading was one of nutrient enrichment. There was no evidence of adverse effects to the ecosystem (Podemski and Culp 1996). Although detailed investigations of spawning grounds and early rearing habitat for fish in the Northern Rivers were not undertaken, it does not appear that dissolved oxygen problems caused by nutrient addition are adversely affecting fish populations at present.

These findings on the contribution of pulp mills to eutrophication of the Athabasca and Wapiti-Smoky rivers resulted in public recommendations to Ministers of Environment in the federal and Alberta provincial governments to eliminate or substantially reduce nutrient discharge to northern rivers, cap direct nutrient loads into specific reaches of the Athabasca and Wapiti-Smoky rivers, and reduce P in pulp mill effluents to minimal concentrations (Northern River Basins Study Board 1996). In the years since these recommendations were made, the Alberta Department of Environment has worked with kraft mill and municipal authorities to decrease nutrient loading to the most affected river (the Wapiti-Smoky system in Alberta) and has stated that any future industrial developments in this basin will be subjected to stringent discharge requirements so as to protect water quality. In addition, the federal government has required all pulp mills to undertake an environmental effects monitoring program that includes an assessment of changes in water quality and biota as a result of effluent inputs.

Patricia Chambers
Environment Canada

growth rates, growth of both young-of-the-year and adult grayling (*Thymallus arcticus*) increased. However, increased benthic invertebrate densities can, in turn, reduce periphyton biomass (Elwood et al. 1981; Winterbourn 1990; Yangdong and Lowe 1995). For example, Biggs and Lowe (1994) found P addition increased the production of invertebrate grazers in the Kakanui River, New Zealand but did not increase periphyton biomass. Removal of grazers resulted in a five-fold increase in periphyton biomass, indicating that a correspondingly more productive, macroinvertebrate community was assimilating increased periphyton production. In contrast with streams, zooplankton are an important part of the invertebrate community in large rivers. Unlike the tight linkage often observed between periphyton and benthic invertebrates in streams, zooplankton biomass in large rivers appears to be only weakly related to phytoplankton biomass (Basu and Pick 1996). This trend may occur because zooplankton are more susceptible to loss than faster-reproducing phytoplankton.

The Yamaska River, Québec: a Watershed under Intensive Agriculture

The Yamaska River basin is located in southwestern Québec and stretches from the Appalachian foothills near the U.S. border to the St. Lawrence Lowlands (Figure 4.7). Almost one-fourth of Québec's agriculture takes place in this basin, with hog and corn production being the two leading agricultural commodities. Some 210 000 hectares of land are cultivated (43% of the total area of the basin), mostly in the St. Lawrence Lowlands. Row-crops (mainly corn, but also soybean and vegetables) make up 58% of the cultivated area. Livestock numbers in 1996 were 311 000 animal units (where, for example, 1 animal unit equals 1 cow or 1 horse or 5 hogs between 20 and 100 kg or 125 chickens), including 168 000 hogs, and livestock density was 1.5 animal units per cultivated hectare. Agricultural intensification has increased sharply in the Yamaska basin over the past 25 years so that impacts of agricultural pressures on water resources are now among the highest in the province. Low summer flow (down to as little as 8 m^3/s) places further stress on the Yamaska River and its tributaries (Primeau et al. 1999).

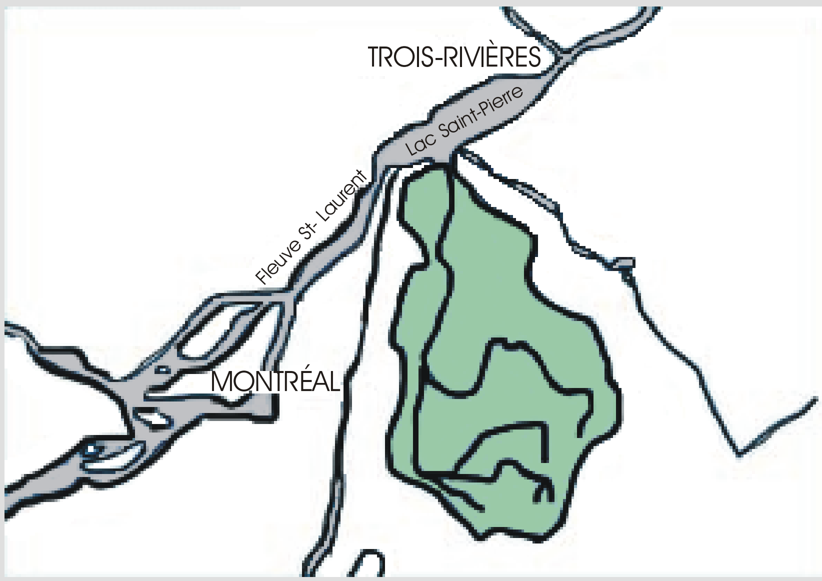

Figure 4.7. The Yamaska River basin, Québec. Adapted from MENV 2000.

Agricultural activities are the principal source of nutrients in the basin. Mineral fertilizers are a source of 13 000 tonnes of N and 3 800 tonnes of P annually, while livestock manure supplies an additional 23 000 tonnes of N and 6 700 tonnes of P. These inputs are very large compared to other sources. Overall, agricultural sources are estimated to contribute 68% of the N and 75% of the P discharged from the river into the St. Lawrence (Delisle et al. 1998). Available nutrients are well in excess of crop requirements in the basin. The Québec Ministry of the Environment estimated recently that organic and mineral fertilizers used in the Yamaska basin exceeded crop requirements for N and P by 144 and 285%, respectively. Excess N and P amount to 80 kg/ha and 27 kg/ha, respectively. Excess nutrients end up in ground and surface waters.

Water quality in the Yamaska River is considered poor by Québec standards. Phosphorus concentrations exceed the Québec guideline for the prevention of eutrophication (0.03 mg/L total P) almost everywhere in the basin. At the mouth of the river, median concentrations for total P (0.195 mg/L) and N (2.15 mg/L) are higher than in any other tributary of the St. Lawrence River (Primeau 1999). Levels of P, N, suspended solids and turbidity throughout the basin are among the highest in Québec and the river is considered to be severely eutrophic.

The most obvious environmental impacts of nutrient enrichment are excessive growth of rooted aquatic plants and algae. In the Yamaska, phytoplankton biomass (expressed as chlorophyll *a* concentration) ranges between 10 and 50 µg/L and increases with distance downstream of the headwaters. These values are very high and provide clear evidence of the advanced state of eutrophication of the river. Large floating mats of Lesser Duckweed (*Lemna minor*) and abundant macrophytes are also observed in reservoirs and backwater zones. The abundance of algae and aquatic plants impair the aesthetic value of the river and its potential for recreational activities.

The Yamaska River, Québec concluded

In addition to causing excessive aquatic plant growth and impairing human enjoyment of the river, eutrophication also affects other aquatic, and sometimes terrestrial, organisms. The degradation of organic matter (macrophytes and algae) leads to oxygen depletion with its adverse, and even lethal, effects on fish and benthic invertebrates. Fish kills have occurred in the Yamaska River in recent years, but exact causes (toxic algae, oxygen depletion, and industrial contaminants) have not been determined. High biological oxygen demand values are observed in the Yamaska River and have been correlated with row-crop acreage, thus suggesting an agricultural origin to oxygen depletion (Primeau 1999).

Larger municipalities in the basin (e.g., Granby, St-Hyacinthe, Cowansville, Farnham) rely upon the Yamaska River as a raw water source for drinking water. Algal blooms cause taste and odour problems, necessitating additional water treatment. In addition, high levels of ammonia have been found in river water during warm spells in recent winters, apparently as a result of runoff from agricultural lands. Adequate disinfection of water with high ammonia content can only be achieved through the use of larger quantities of chlorine, thereby resulting in safe but less palatable drinking water.

Water quality in the Yamaska River is considered the worst in Québec. All available data point to agricultural activities as the main cause of the current situation. Better nutrient management, especially of fertilizers (both commercial and manure) is a key factor for the rehabilitation of water and ecosystem quality in the basin. To that end, in 1988 the Québec Government launched a program supporting construction of manure storage tanks. Over 1 million m^3 of manure are now adequately stocked and better managed in the basin. In 1997, the adoption of *Regulations Respecting the Reduction of Pollution from Agricultural Sources* led to a new program supporting agro-environmental initiatives. Central to this strategy is the use of fertilization plans by farmers, whereby fertilizer applications are matched to crop requirements. Recently, the *Union des producteurs agricoles* completed a comprehensive agro-environmental assessment of Québec's agriculture sector, with a view to improving its environmental performance. Clearly, farmers are now more aware of environmental issues, particularly nutrient management, the first step towards establishment of sustainable agriculture in the Yamaska basin and elsewhere in Québec.

Jean Painchaud
Ministère de l'environnement du Québec

Increases in algal and invertebrate production in response to nutrient addition may, in turn, increase fish production. For example, fertilizer addition to the Keogh River, British Columbia resulted in a five to ten-fold increase in periphyton biomass, which translated to a 1.4 to two-fold increase in salmonid fry weights (Johnston et al. 1990). In the Bow River, Alberta, nutrients from sewage treatment plant discharges sustain an internationally known trout sport fishery in the 50-km reach downstream of Calgary (Culp et al. 1992). Hoyer and Canfield (1991) further showed that total fish standing crop was significantly correlated to total P concentrations for 79 rivers throughout North America.

Because nutrients are responsible for increased growth of riverine plants and changes in the abundance and taxonomic compositions of insects and fish then, in theory, removal or reduction of these nutrient inputs to polluted rivers should improve conditions. This premise is certainly the theory upon which lake remediation rests, namely that reduced nutrient loading (particularly P) will decrease the abundance of planktonic algae and improve water clarity. Compared to lakes and large rivers where water-column concentrations of P and N are good predictors of planktonic algae abundance (Dillon and Rigler 1974; Canfield 1983; Søballe and Kimmel 1987; Basu and Pick 1996), nutrient concentrations show varying degrees of success at predicting periphyton abundance in streams and moderate-size rivers (Biggs and Close 1989; Lohman et al. 1992; Scrimgeour and Chambers 1996; Dodds et al. 1997; Cattaneo et al. 1997; Carr and Chambers 1998; Chételat et al. 1999) and are even less successful at predicting the abundance of rooted aquatic plants (Kern-Hansen and Dawson 1978; Canfield and Hoyer 1988; Duarte and Canfield 1990). However, reductions in nutrient loading have improved water quality in some rivers. In the Thompson River, British Columbia, start-up of a bleached kraft mill at Kamloops in 1972 was associated with massive accumulation of benthic algae. Changes to the operation of the Kamloops municipal sewage and the mill wastewater treatment systems resulted in an approximate 35% reduction in total P loading between 1973-75 to 1989. This decline coincided with a 60% decline in periphyton biomass (Bothwell et al. 1992). Similarly, improvements to the City of Calgary's sewage treatment plant reduced total P, total ammonia and nitrate+nitrite loading to the Bow River by 80, 53 and 50%, respectively, between 1982 and 1988. This reduction coincided with a considerable decline in the biomass of rooted aquatic plants (Sosiak 2000).

In summary, studies on the effects of nutrient addition to river systems have generally found that in the case of moderate enrichment to rivers with intrinsically low nutrient concentrations, the productivity of river biota is increased, often without loss of species. Gross pollution involving de-oxygenation of the water results in reduced productivity by periphyton, benthic invertebrates and fish, and loss of species.

4.4. Wetlands

Wetlands are unique ecosystems that are distinguished on the basis of their vegetation, morphology, hydrology and chemistry. They are found in both freshwater and marine environments, and are among the world's most productive environments in terms of carbon incorporation and biomass accrual (Usher and Scarth 1990). Wetlands create habitats for plants and animals: it is estimated that 35% of Canada's rare and endangered animals are in someway dependent on wetlands (Mathias and Moyle 1992). In addition, they perform major ecological functions including provision of nursery or stopping grounds for fish and wildlife, flood control, nutrient retention, purification of aquifers, biogeochemical transformations of elements, and exchange of chemicals, nutrients and organic matter with adjacent ecosystems (Urban and Bayley 1986; Catalla 1993; Hook 1993). Fourteen percent of Canada is covered by wetlands; they are found along the shores of oceans, lakes and rivers, dotted across the Prairies and in countless, poorly drained depressions in the Canadian Shield (NWWG 1988).

The growth and stability of the vegetation in a wetland influences the composition and abundance of invertebrates and vertebrates and defines its biological productivity (Sjörs 1950; Wedeles et al. 1992). Although conversion of wetlands to urban and agricultural land has been the major cause of wetland loss (Moore et al. 1989), a less obvious cause of wetland decline is N addition (Morris 1991; WHO 1997a). Nitrogen often limits growth of wetland vegetation. Excessive inputs may accelerate eutrophication, therefore causing changes in water quality, habitat availability, and biodiversity. Although the effects of nutrient addition on wetlands are less well known than for other aquatic systems, two consequences of nutrient addition on wetland plants are typically observed: an increase in emergent biomass (Willis 1963; Wisheu et al. 1991) and a decrease in plant species diversity (Al-Mufti et al. 1977; Grime 1977; Vermeer and Berendse 1983; Wilson and Keddy 1988). Vegetation surveys and laboratory fertilization experiments suggest that nutrient enrichment of wetlands is a two-stage process. Under moderate nutrient loading, the number of species per unit area and biomass production increase. At higher nutrient loading, plant growth becomes limited due to competition for light (by shoots and leaves) and for space (by roots) (e.g., Willis 1963; Thurston 1969; Grime 1973; Al-Mufti et al. 1977). Plant diversity also gives way to dominance by single species due to invasion by aggressive N-loving, highly-tolerant, non-native vegetation such as purple loosestrife (*Lythrum* *salicaria*), water hyacinth (*Eichornia crassipes*), and giant salvinia (*Salvinia molesta*), or dominance by a monoculture of cattails (*Typha spp*.) or common reeds (*Phragmites spp*.) (Keddy 1988; Mitsch and Gosselink 1993). In forest wetlands, characteristic herbaceous species may be lost and replaced by herbaceous species not normally found in these habitats (Ehrenfeld 1983). These changes in vegetative biomass and species composition as well as associated abiotic changes (e.g., change in water balance due to increased evapotranspiration rates; Chalmers 1979; Howes et al. 1986) can influence the direction of wetland succession.

Changes in agricultural land management practices and sewage inputs have increased the N and P inputs to drainage basins, including wetlands. However, acid deposition is another significant source of N to wetlands (Morris 1991). Not only can the N in atmospheric deposition have direct effects on the abundance and composition of wetland vegetation, it can also change water and soil acidity which, in turn, may affect the availability of nutrients through processes such as phosphate adsorption, nitrification, denitrification and nitrogen fixation, and modify the speciation of metals (Gorham et al. 1984; Crowder 1991; Vaithiyanathan et al. 1996). In addition to direct effects of added N, these alterations may cause serious changes in the entire aquatic food web of wetlands.

Because the quantity of pollutants in water declines as it passes through a wetland, marshes have been considered for their potential to provide secondary treatment of urban and industrial stormwater, sludge and wastewater. However, these effluents also have high BOD concentrations that can lower dissolved oxygen concentrations in wetlands to undesirable levels (Teal and Teal 1969). In addition, nutrient loading from urban waste receiving only primary treatment can overwhelm a wetland and cause it to become choked with phytoplankton, thereby blocking light penetration to deeper vegetation (MacKinnon and Scott 1984). Wetlands are, however, well suited for nutrient removal as part of the "tertiary" or "polishing" stage of sewage treatment and stormwater runoff (MacKinnon and Scott 1984; Kadlec and Knight 1996).

Wetlands in freshwater and marine habitats differ on the basis of their vegetation, hydrology and chemistry. Because of these differences, the effects of nutrient addition will be considered separately for these two systems.

Freshwater Wetlands

Freshwater wetlands exhibit a continuum of variation in space and time relating to characteristics such as the quantity and source of ionic inputs, persistence of standing water, and vegetative composition. At either end of this continuum are ombrotrophic bogs (which are insulated from ground water by a thick layer of peat and receive most of their water and mineral salts from the atmosphere) and swamps and marshes (which have access to mineral-rich ground water). Swamps and marshes (also known as fens in Europe) are usually distinguished by the presence (swamps) or absence (marshes) of trees.

Ombrotrophic bogs are typically acidic. Because they are rain-nourished (i.e., they derive all their nutrients from the atmosphere), ombrotrophic bogs are particularly sensitive to airborne N loads (Urban and Eisenreich 1988). There are no reports as to the effects of N enrichment on ombrotrophic bogs in Canada; however, studies in the Netherlands and the United Kingdom have shown that the dominant species, *Sphagnum*, as well as other plant species favoured by low N concentrations, may decrease in abundance and be replaced by more nitrophilious plants (Lee et al. 1989; Greven 1992; Aaby 1994). Similarly, experimental application of more than 10 kg N (as NH_4/NH_3)/ha/year to Swedish ombrotrophic

Delta Marsh, Manitoba: a Prairie Wetland

Prairie wetlands are complex ecosystems that support a rich and abundant flora and fauna (NWWG 1988). This biodiversity is increasingly threatened by agricultural activities (e.g., nutrient addition, pesticides, plough-down of residues), the ecological effects of which are not fully understood. The fascination with wetlands lies in their rich and diverse wildlife as well as the uniqueness of the landscape (Griffiths and Townsend 1987). For migrating birds, especially waterfowl, wetlands are essential feeding and resting stopover grounds, as well as nesting and moulting grounds (NWWG 1988).

Delta Marsh stretches along the southern shore of Lake Manitoba. It is the largest (200 000 ha) of several marshes in the Lake Manitoba basin and forms one of the finest waterfowl breeding and harvesting areas on the Canadian prairies (MDMNR 1968). The marsh consists of a series of large and small interconnected bays and isolated ponds, separated from Lake Manitoba by a forested sand ridge (MacKenzie 1982). The four dominant emergent species in the Delta Marsh are: *Phragmites australis*; *Scolochloa festucacea*; *Typha glauca,* and *Scirpus lacustris* (Squires and Vander Valk 1992).

Although Delta Marsh borders fertile agricultural land, it is considered naturally eutrophic and not degraded by human activity. The marsh is used as a field station by the University of Manitoba to study the effects of nutrient additions on a prairie wetland. Nutrient additions have been found to enhance primary production in oligotrophic nutrient-poor wetlands in the Interlake Area of Manitoba (Murkin et al. 1992; Murkin et al. 1994; Gabor et al. 1994); however, the effects of nutrient enrichment of a eutrophic prairie marsh are not well understood. Experiments conducted by University of Manitoba scientists and others have shown that N and P addition in the range applied to small heavily-fertilized watersheds (i.e., 2.5 to 10 kg/ha N) can change the species composition of emergent vegetation (Neill 1990). Effects on other organisms (e.g., algae and aquatic invertebrates) are less clear cut, although it appears that increases in nutrient loading, up to a point, are actually favourable to waterfowl by increasing invertebrate production (Murkin et al. 1994). Further research is needed to determine the chemical processes involved in nutrient availability in these systems and the potential effect of fertilization on their productivity and overall function.

Elizabeth Roberts
Environment Canada

bogs lowered the density of sundew (*Drosera rotundifolia*), a characteristic bog species, due to enhanced competition for light with tall species (e.g., *Eriophorum* and *Andromeda*) which responded positively to the increased N inputs (Redbo-Torstennsson 1994; WHO 1997a). Because increased N loads strongly affect ombrotrophic bogs, due to their high N retention capacity and closed N cycling, WHO (1997a) estimated the critical atmospheric load of N to be 5 to 10 kg N/ha/year for ombrotrophic bogs. Exceedance of this critical atmospheric load would alter the chemical, biological or hydrological function of the wetland.

In contrast to bogs, fens are wetlands that are found in alkaline to slightly acidic habitats (NWWG 1988). Studies of mesotrophic fens in the Netherlands have shown an increase in tall graminoids (grass or *Carex* spp.) and a decrease in the diversity of subordinate plant species in response to N addition (Vermeer 1986; Verhoeven and Schmitz 1991; Koerselman and Verhoeven 1992). Based on these and other Dutch studies, the WHO (1997a) established a critical atmospheric load of 20 to 35 kg N/ha/year for fen systems. Marshes represent wetlands with open nutrient cycles and, as such, are often better equipped to handle the ill effects of nutrient additions (Morris 1991; WHO 1997a; see Delta Marsh case study). However, when the nutrient load, regardless of source, exceeds the filtering capacity of these wetlands, the marshes begin to die (see Cootes Paradise case study).

Cootes Paradise, Ontario: Urban Impacts on a Great Lake's Marsh

Land use conflicts that pit urban development against maintenance of natural ecosystem function are perhaps best illustrated by the case of wetlands and nowhere is this issue better illustrated than in southern Ontario. Today, the Canadian coastal marshes of the lower Great Lakes remain important resources for waterfowl, even though they have undergone drastic change over the last half century from draining, development, eutrophication and pollution (Mitsch 1992).

Cootes Paradise Marsh is a 250 ha wildlife sanctuary located at the western end of Hamilton Harbour, Lake Ontario. It is an important staging area for waterfowl as well as providing nursery and spawning habitat for sportfish, and serving as a local scientific and educational resource (Chow-Fraser and Lukasik 1995).

Over the last century, nutrient inputs from urban, rural and agricultural sources, water level fluctuations, sediment inputs from creeks, and the regular re-suspension of these sediments by wind and the feeding and spawning activities of carp have degraded Cootes Paradise (Chow-Fraser 1998). From 1919 to 1962, a municipal wastewater treatment plant (MWTP) in Dundas, Ontario discharged primary-treated sewage into the western end of the marsh. In 1963, the MWTP was upgraded to a secondary treatment facility (McLarty and Thachuk 1986) although P loading from the MWTP remained high until the early 1970s (approximately 45 kg P/d). In 1978, the plant was again upgraded to a tertiary treatment facility although sand filters to reduce further the loading of P into the marsh were not added until 1987 (Painter et al. 1991). The P loads from the MWTP have steadily decreased from greater than 45 kg/d in the early 1970s (Semkin et al. 1976) to well below 4 kg P/d in recent years (Chow-Fraser et al. 1998). In addition to the MWTP, nutrient loading (especially ammonia and soluble reactive P) has been linked to combined sewer overflows discharging untreated sewage and surface runoff during storm events (Chow-Fraser et al. 1996). Recently, retention tanks have been built to contain effluent from these overflows so that they no longer discharge directly into the marsh (Chow-Fraser et al. 1996). Finally, creeks flowing into the marsh have been contributors of large quantities of sediments and nutrients from the surrounding agricultural land (Chow-Fraser et al. 1996; Chow-Fraser et al. 1998).

The 1978 upgrade of the Dundas MWTP was responsible for dramatic reductions in the concentrations of N and P in the water column (Chow-Fraser et al. 1998; Table 4.3). Prior to the tertiary treatment upgrade, there was a very steep gradient of total phosphorus (TP), soluble reactive phosphorus (SRP), ammonia, and nitrate concentrations from the sewage plant outfall to the marsh outflow. Mean concentrations of TP and SRP entering the marsh at the sewage outfall were almost five and ten-fold higher, respectively, than those in the open water (5 vs. 1 mg P/L TP; 1.4 vs. 0.1 mg P/L SRP); total ammonia-N levels were more than ten times higher (13 vs. 1 mg NH_3^+-N/L) and total nitrate-N five times higher (4.5 vs. 1 mg NO_3^-N/L) (Painter et al. 1989; Chow-Fraser et al. 1998). In the eight years following upgrade to a tertiary facility, the sharp west-to-east gradient in TP concentrations has disappeared (Lougheed and Chow-Fraser 1998). During this period also, the treatment plant began aerating its sewage effluent to reduce the total ammonia-N loading to the marsh, but this additional process has increased nitrate-N loading. Consequently, nitrate concentrations have doubled since 1978 at the sites closest to the sewage outfall (9 to 11 mg NO_3-N/L) and are consistently ten times higher than at the open-water sites (0.5 to 1 mg NO_3-N/L). Moving the sewage outfall to a less restricted area with more mixing could alleviate these conditions.

A common characteristic of many degraded coastal marshes of the Great Lakes is the absence of submersed vegetation. Despite improvements in water quality, less than 10% of Cootes Paradise was covered with either submerged or emergent vegetation by the 1990s (Chow-Fraser et al. 1996) and the prominent members of the emergent community were introduced species, common reed (*Phragmites australis*) and purple loosestrife (*Lythrum salicaria*) (Chow-Fraser et al. 1998). High water levels in the 1940s and 1950s were the original cause for loss of emergent vegetation and resulting habitat changes; however, nutrient enrichment has played a significant role in the degradation of Cootes Paradise (Chow-Fraser et al. 1998).

Cootes Paradise concluded

Table 4.3. Water quality characteristics of Cootes Paradise Marsh in 1973 and 1994 in three stations (open water; vegetated; and Dundas MWTP outfall).

Parameter (mg/L)	Open water		Vegetated		MWTP outfall	
	1973	1994	1973	1994	1973	1994
Dissolved Oxygen - surface	10.5	9.4	NA	7.3	7.4	11.1
Total Phosphorus	2.22	0.13	NA	0.21	15.4	0.15
Total Dissolved Phosphorus	0.96	0.02	NA	0.06	10.7	0.07
Total Nitrate Nitrogen	0.70	0.58	NA	0.35	2.16	10.3
Total Ammonia Nitrogen	0.45	0.14	NA	0.16	8.27	0.27
Total Nitrogen	NA	3.22	NA	2.6	NA	12.6

Data from Bacchus (1974) and Lougheed and Chow-Fraser (1998).

Changes in the algal community of Cootes Paradise over the past four decades have mirrored changes in water quality and plant cover (Chow-Fraser 1998). Although the type of algae found in the marsh was not affected by degrading water quality, the quantity of algae collected at various sites in the marsh was affected (Chow-Fraser et al. 1998). For example, algal counts in an inlet receiving nutrient loading from a storm sewer and runoff from a nearby domestic landfill (Sims 1949) were higher than vegetated sites where poor light penetration and lower nutrient availability contributed to lower algal densities. Blue-green algae numbers increased in the 1970s (Bacchus 1974). However, the introduction of P removal at the Dundas MWTP in 1993 resulted in a reduction in blue-green algal abundance and increased diversity of green algae. There was also a return of *Chrysophyceae* (‘golden algae’) to the marsh, indicative of less eutrophic conditions (Wetzel 1983; Chow-Fraser et al. 1998). The most significant improvement in terms of the algal community was the development of P limitation in phytoplankton found in the open water.

Vegetation loss in the 1940s caused bottom sediments to become unconsolidated and unable to support a diverse assemblage of aquatic insect larvae (Chow-Fraser 1998). Hence, species diversity of aquatic insects declined dramatically over the past forty years from 57 genera (23 families and 6 orders) in 1948, to 9 genera (6 families and 3 orders) in 1978, to only 5 genera (3 families and 2 orders) in 1995 (Chow-Fraser et al. 1998). The diverse benthic community of the 1940s has now been replaced by a community consisting primarily of chironomid larvae, oligochaetes, and other worms associated with low-oxygen environments (Chow-Fraser et al. 1998). Benthic grazers have been kept in low abundance due to predation by benthivorous carp allowing epiphytic algae to proliferate further, contributing to light limitation of macrophytes.

Historical checklists indicate that Cootes Paradise once supported an important warmwater fishery (Whillans 1996). However, from 1948 to 1978, the abundance of sportfish declined probably in response to high diurnal fluxes in dissolved oxygen (< 2 mg O_2/L at night) in conjunction with the growth of the carp population (Chow-Fraser et al. 1998). The most abundant fish caught in the marsh in recent years have been cyprinids (common carp and hybrids), centrarchids (pumpkinseeds), and clupeids (gizzard shad and alewife), whereas the piscivores such as pike and bass have been exceedingly rare (Kay 1949; Chow-Fraser et al. 1998).

The Hamilton Harbour Remedial Action Plan is currently trying to restore Cootes Paradise Marsh (Rodgers et al. 1992; Chow-Fraser et al. 1996). Nutrient enrichment has been a significant contribution to the degradation of Cootes Paradise. In the absence of nutrient enrichment and other anthropogenic stressors (e.g., further deteriorations in water quality, encroachment by development, and introduction of exotic species), the emergent marsh might be able to recover in future decades.

Elizabeth Roberts
Environment Canada

Salt Marshes along the Bay of Fundy, Nova Scotia

Salt marshes occur, sometimes extensively, along the coast of the Bay of Fundy, Nova Scotia. These salt marshes are dominated by *Spartina alterniflora* and *Spartina patens*, which grow year-round making their productivity much greater than that of adjacent terrestrial vegetation (Teal and Teal 1969). The marshes in the Bay of Fundy are in many ways unique ecosystems defined by physical processes, of which tidal mixing is most prominent (NWWG 1988; Brylinsky et al. 1996).

Much of the research on *S. alterniflora* salt marshes has emphasized their contribution to marine coastal fisheries (Odum 1980; Pomeroy and Wiegert 1981). The tides that make high production of marsh vegetation possible also remove half the nutrients before the marsh's flora and fauna can use them (MacKinnon and Scott 1984; Keizer and Gordon 1985). In addition, *S. alterniflora* may act as a "pump", taking up P from the sediments and releasing it into tidal waters (Hatcher and Patriquin 1981). Thus, large amounts of nutrients and organic matter are released to adjacent estuaries, stimulating the coastal water food web. In this way, salt marshes enhance coastal fisheries, including shellfisheries (Teal and Teal 1969; Odum 1980).

The flora in some parts of Atlantic coastal plains is found nowhere else in the world (Fernald 1942; Roland 1976) and eutrophication by enriched runoff from land use disturbances could alter the species composition of shoreline vegetation (Ehrenfeld 1983). Coastal plain plants are weak competitors and could be displaced by taller, faster growing invasive plants that would move onto enriched shores (Wisheu and Keddy 1994). Wisheu et al. (1994) recommended, therefore, that in addition to securing habitat, a watershed management plan be developed to prevent nutrient loading from degrading coastal plain habitats.

Elizabeth Roberts
Environment Canada

Saltwater Wetlands

Saltwater wetlands or "salt marshes" are grass-dominated coastal ecosystems subjected to the inundation of saline tidal waters (Chapman 1974). The salt marsh is the most terrestrial extreme of the marine environment. Plants that live in this habitat are actually terrestrial species that cannot compete with other land plants but are able to tolerate saline conditions (Phleger 1971; Hatcher and Patriquin 1981). Salt marshes are characterized by complex interactions among oceanographic, biological, geological, chemical and hydrological processes (Glooschenko 1979). They are important in the global cycles of N and sulfur (NWWG 1988) by possessing open nutrient cycles and serving as large nutrient reservoirs. In Canada, salt marshes represent a relatively small part of the total wetland acreage (NWWG 1988), in part because of their need for salt water and tides, a narrowly defined physical requirement not commonly found in Canada (Chapman 1960; MacKinnon and Scott 1984).

Salt marshes perform important ecosystem functions including reduction of nutrient and organic export to adjacent coastal waters; spawning grounds for commercially important fish and invertebrates; nesting and staging areas for waterfowl and shorebirds; and areas of waste and toxic substance assimilation (NWWG 1988). In addition, the continuous cycling of nutrients between a salt marsh and the adjacent estuary replenishes the marsh with nutrients, promotes productive mollusc and crustacean growth in the estuary, and enhances offshore fisheries (Teal and Teal 1969).

Salt marshes, in contrast to freshwater wetlands, are strictly N limited (Valiela et al. 1975; Jefferies and Perkins 1977; Cargill and Jefferies 1984; Morris 1991). Information on the impacts of nutrients on saltwater wetlands is not as extensive as for their freshwater counterparts, possibly because the influence of tides that remove some of the nutrients from the systems. However, the effects of

Fraser River Estuary, British Columbia: Marshes versus Development

On the Pacific coast of Canada, estuaries and lagoons make up 10-20% of the shoreline (MacDonald 1977). Coastal marshes are restricted to the head of protected inlets and the deltas of major rivers, with the most extensive being the one that fringes the Fraser River Delta. Here, the chemistry of both the sediments and water is largely determined by the seasonal pattern in discharge and sediment load from the Fraser River (Hutchinson 1982; NWWG 1988).

The southwestern Fraser River Lowland extends from approximately Sea Island east to the Surrey/Langley border and covers most of the land south of the north arm of the Fraser River. This area is one of the most agriculturally productive areas in Canada (Pilon and Kerr 1984). Because many of the wetlands in this region have been drained for agriculture, urbanization, and industrial expansion (NWWG 1988), the wetlands are now largely confined to the 25 213 ha Fraser River Estuary (Ward 1992). Brackish estuarine marshes, characterized by sedges (*Carex lyngbyei*) and bulrushes (*Scirpus americanus*), occur along the delta front where salt and fresh water mix and represent about 93% of the marshes in the estuary. Salt marshes account for the remaining 7%. Productivity of both marsh types is high owing to high nutrient inputs (Bradfield and Porter 1982), especially N (Valiela and Teal 1979).

The amount, location, and nature of urban encroachment to wetlands are subjects of major concern within the Fraser River Lowland (Pilon and Kerr 1984; Ward 1992). Over the last few decades, activities such as dredging, draining and infilling have severely reduced wetland area (NWWG 1988). In addition to overt destruction, construction of dykes and river training structures have had indirect effects on the marshes due to changes in salinity, erosion and sediment redistribution, and nutrient supply (Kerr 1987). For example, activities such as building of causeways and jetties for ship terminals have changed sediment dispersal patterns in some marshes leading to more rapid rates of sedimentation locally and reduced inflow of freshwater (Clague and Luternauer 1982). Although much concern has focused on overt destruction of marsh habitat (Hutchinson 1982), contaminated urban and agricultural runoff, sewage discharge, industrial effluents, landfill leachate and air pollution deposition continue to degrade these wetlands (Harrison 1985; Kerr 1987; NWWG 1988; Stanley Associates Engineering Ltd 1992; Ward 1992; IRC 1994a, b). Perhaps the most detrimental of these sources in terms of high nutrient loads is sewage (Environment Canada and BCELP 1992). Municipal sewage systems discharging to the Fraser River collect and treat not only the domestic flow of over 80% of the approximate 1.5 million people living in the Fraser River Basin population, but also industrial and commercial wastewater (Dorcey and Griggs 1991; UMA 1993). In addition to nutrients, municipal wastewater discharges to the basin contain an array of potentially toxic compounds, including cyanide, chlorine, anionic surfactants, heavy metals, organic materials, phenols and sulphides (Environment Canada and BCELP 1992).

Purple loosestrife recently invaded the mashes of the Fraser River Estuary due, in part, to increased nutrient enrichment (Adams 1993; Grout et al. 1997). Concern has been raised that this plant may be out-competing native plants such as sedges. A shift in the productivity of native plants relative to purple loosestrife in the estuary's vegetation community may not only affect the flora but also the wetland food web due to seasonal differences in detrital supply (Grout et al. 1997). The timing and composition of detrital supplies to the Fraser River Estuary are important factors affecting invertebrate consumers (Sibert 1982). Because purple loosestrife decays more rapidly than sedges, most of the detritus from decomposing loosestrife may be released in the autumn only and not slowly over the whole year (Grout et al. 1997).

The Fraser River Estuary provides homes for the largest population of wintering waterfowl in Canada (NWWG 1988) and is one the most important stopping points for migrating birds on the Pacific Flyway (Cameron and Obee 1981; Leach 1982). It also supports one of the largest salmon runs in Canada (Dorcey et al. 1978; Hall and Yesaki 1987). Maintenance of the wetlands in the Fraser River Estuary will depend upon a balance between development, other human activities and nature.

Elizabeth Roberts
Environment Canada

increased anthropogenic nutrient inputs, especially N, can have serious consequences. The Bay of Fundy and the Fraser River Estuary on the east and west coasts of Canada, respectively, provide examples of some of these effects (see case studies).

4.5. Coastal Waters

In coastal waters, primary production is largely controlled by nutrient and light availability. The ability to absorb nutrients from the water column, as well as light availability, governs the shift in dominance from slow-growing seagrasses and large macroalgae to faster-growing macroalgae and phytoplankton and, finally, to motile phytoplankton (Duarte 1995). When light is plentiful and nutrients are limiting, the slow-growing seagrasses and large macroalgae are more competitive as they are capable of efficient internal nutrient cycling and, in the case of seagrasses, can access nutrient pools in the sediment (Duarte 1995). As the process of eutrophication continues and nutrient concentrations increase, the abundance of fast-growing macroalgae and phytoplankton increase and, because their biomass occurs close to the water surface, seagrasses are shaded reducing their ability to fix carbon. As the frequency of phytoplankton blooms increases, oxygen production and respiration become uncoupled (Duarte 1995). Increased microbial activity due to sedimentation of organic material causes oxygen demand to exceed oxygen production resulting in deep-water anoxia. Sediment anoxia impairs the ability of seagrasses to acquire N and accelerates seagrass mortality (Duarte 1995). With the loss of seagrasses, turbidity increases as the sediment is no longer stabilized, a situation that further favours motile phytoplankton that can move to the surface to maximize light exposure (Duarte 1995). Anoxia also enhances the release of nutrients from the sediments (i.e., internal nutrient loading), again favouring phytoplankton. This shift in the structure of the marine plant community is a self-accelerating cascade caused by the direct and indirect effects of increased internal nutrient loading and increased shading of benthic plants as the environment shifts from nutrient-limited to light-limited. In the marine environment, this ‘bottom-up’ control of the structure of the primary producer community by nutrients occurs at lower nutrient concentrations than in lakes (Dederen 1992).

The coastal zone is a physically dynamic ecosystem open to constant exchange of water with the open ocean. Circulation and tidal patterns in the near shore and estuarine currents near the mouths of rivers result in mixing of nutrient-poor surface water from the open ocean with the nutrient-rich water of the coastal zone (Persson 1991). The extent of mixing is often determined by salinity and temperature, which set up density gradients that limit blending of different water masses. Typically, as the less dense freshwater enters the coastal zone it creates a net seaward flow of low salinity water on the surface with a net landward flow of deep, denser, saline water (Harrison et al. 1994). This exchange of water is responsible for nutrient dilution along coasts. The extent of these exchanges largely determines the nature and extent of eutrophication in coastal water. In addition to salinity and temperature gradients, deep basins or fjords can also be isolated from water exchange because of physical barriers such as protruding sills (Persson 1991). Depending on the depth of the sill relative to the total basin depth, the deep water in these basins may only naturally re-oxygenate at intervals. The Georgia Basin and Halifax Harbour case studies provide examples of the complex interactions determining nutrient concentrations in coastal waters.

Nitrogen is generally the nutrient limiting primary production in north-temperate oceans, in contrast with freshwaters where P is typically the limiting nutrient (Howarth 1988; Vollenweider 1992). Evidence from the Northern Hemisphere indicates that over-enrichment of coastal waters has created a niche

Marine Aquaculture: Role in Coastal Eutrophication

Nutrient loading from finfish aquaculture operations has the potential to stimulate eutrophication of coastal waters as a result of excretion of dissolved or solid waste, and decomposition of unconsumed feed. The contribution of aquaculture to N and P loading has been identified for several marine operations in Canada. In an assessment of the contribution of salmon aquaculture, traditional fisheries, fish processing plants, a pulp mill and a municipal sewage treatment plant to nutrient loading of Letang Inlet (an embayment of the Bay of Fundy), salmon aquaculture was found to be the largest anthropogenic source of nutrients, contributing 56% of the total N and total P loads to the basin (Strain et al. 1995). These added nutrients could cause eutrophication and possibly trigger microalgal blooms in the vicinity of the cages (Wildish et al. 1990).

In a study of salmonid aquaculture in the Bay of Fundy, localized increases in ammonia were observed close to fish cages although ammonia concentrations for the entire inlet were not above background levels (Wildish et al. 1993). There was no evidence, however, of increased phytoplankton abundance in the vicinities of the cages, which would indicate a eutrophying effect by the salmon cages. This result was surprising as substantial epiphytic communities, including attached macroalgae, generally develop on the pens. In an assessment of the potential for eutrophication from coastal British Columbia fish farms, Black and Forbes (1997) reported that the total dissolved waste from fish farms represented less than 0.3% of the N load to the Strait of Georgia. Taylor et al. (1994) in a study of Sechelt Inlet, British Columbia from 1988 to 1990 concluded that salmon farms could not be shown to affect nutrient regimes or algal blooms in the inlet. Black and Forbes (1997) suggest that the aquaculture industry in coastal British Columbia is unlikely to have a significant widespread eutrophying effect at current production levels.

Although there appears to be no widespread enrichment effects caused by marine aquaculture, there is the potential for local effects immediately underneath cages (DFO 1997). Of the approximately 70% of nutrients typically lost to the environment from aquaculture operations, about 32% of N and 63% of P is lost in particulate form that ends up in the sediment (Holby and Hall 1991; Hall et al. 1992). This influx of particulate organic material can increase oxygen demand in the sediment, thereby increasing sediment nutrient release rates (Holby and Hall 1991; Hall et al. 1992, BCEAO 1997). On the Canadian East Coast, oxygen demand has been found to be four times higher under aquaculture pens than that at reference sites (Hargrave et al. 1993). Mild enrichment by organic sedimentation stimulates the abundance and biomass of the benthic community directly under the pens. If aquaculture pens are poorly sited, pollution sensitive species are lost and the benthic invertebrate community under the cages comes to be dominated by organic pollution taxa, such as the polychaete *Capitella capitata* and nematodes, interspersed among patches of anoxic sediment (BCEAO 1997; DFO 1997). In coastal operations in British Columbia, sediments 20 to 30m away from the pens show no apparent impacts of waste deposition (Cross 1990). Again, proper siting of the pens lessens the impact to the sediments.

Martha Guy and Patricia Chambers
Environment Canada

occupied by a diverse group of dinoflagellates and diatoms that like their counterparts in eutrophic lakes, blue-green algae, produce toxic chemicals (Burkholder et al. 1992; Taylor et al. 1994; see PEI case study, Section 5.3). Marine algae have been found responsible for at least four different illnesses in human consumers of molluscs as well as massive mortalities of fish, birds, and marine mammals (Paerl 1997). It is believed that blue-green algae do not dominate primary production in the ocean because of a decreased availability of iron and molybdenum, two micronutrients essential for nitrogen fixation by blue-green algae (Paerl 1997). In the Baltic and Mediterranean Seas, the frequency of blooms of these 'harmful' (i.e., toxic, hypoxia-inducing, food web-altering) algae has increased since the 1950s, as has atmospheric emissions of N associated with urban, industrial, and agricultural expansion in Western and Central Europe (Paerl and Whitall 1999). The occurrence of these 'harmful algal

Georgia Basin, British Columbia: the Land-Ocean Interface

The Georgia Basin Region is an extensive estuarine system between southern Vancouver Island and the mainland of British Columbia and Washington State (Figure 4.8). It extends from the northern end of the Strait of Georgia to the southern end of Puget Sound and is joined directly to the Pacific Ocean by the Juan de Fuca Strait (West et al. 1994). It represents an area of some of the most rapid population growth in North America. In 1996, approximately 4.5 million people lived in communities surrounding the basin dominated by the cities of Greater Vancouver and Seattle on the mainland and Victoria on Vancouver Island (BCSTATS 1998a; US Census Bureau 1998). By 2010, the population of the area is expected to grow by 28% with much of this growth occurring in unincorporated communities with little to no municipal or intensive regional government (West et al. 1994; BCSTATS 1998b). The potential clearly exists for increased nutrient runoff to coastal waters near these communities.

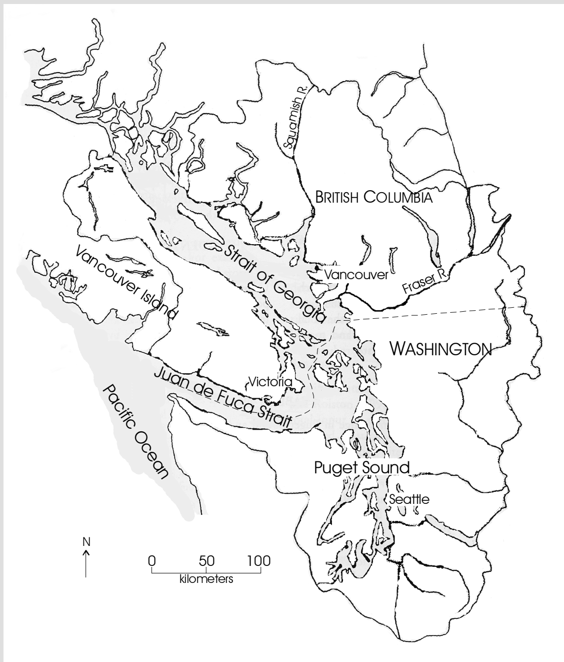

Figure 4.8. The Georgia Basin, British Columbia. The Georgia Basin includes the Straits of Juan de Fuca and Georgia and Puget Sound. Adapted from West et al. 1994.

A net seaward outflow in the upper portion of the water column and a net landward inflow in the lower portion of the water column characterize water circulation in the Georgia Basin (Thomson 1994). Deep inflowing waters supply the Strait of Juan de Fuca with 2 500 to 2 800 t N/d of total N (Harrison et al. 1994; Table 4.4). Depending upon interactions among the general circulation pattern, tides and winds, anywhere from 2 000 to 2 100 t N/d leaves the Juan de Fuca Strait in the seaward movement of surface waters and 500 to 1 000 t N/d is incorporated into biological matter or sediments. Except for a brief period from May to September, natural inputs of nutrients in the Georgia Basin are at concentrations in excess of plant requirements (i.e., nutrient-saturated conditions) (Harrison et al. 1983). This season coincides with the period of maximum algal growth and isolated bays and inlets are most susceptible to the potential impacts of human nutrient inputs. Land-based N inputs to the Georgia Basin from municipal MWTPs discharging directly to the basin plus Fraser River loads were estimated to be 115-130 t N/d (Harrison et al. 1994; Table 4.4). Although P concentrations are very low during the summer, it is rapidly regenerated and, hence, is not limiting (Harrison et al. 1983). Land-based inputs to the Georgia Basin proper are small compared to those cycled naturally as part of the general circulation pattern of the estuary, suggesting that on the broad-scale of the entire estuary, anthropogenic nutrients are not causing eutrophication.

Although natural N inputs to Georgia Basin far exceed anthropogenic inputs, cultural eutrophication has been observed in small, isolated bays and inlets where mixing is reduced. For example, minor increases in phytoplankton abundance and sporadic oxygen depletion have been observed in the bottom waters of Howe Sound, an embayment of Georgia Strait, and have been attributed to nutrient inputs from pulp and paper mills

Georgia Basin concluded

Table 4.4. Estimated N inputs to Georgia Basin. Adapted from Harrison et al. 1994.

Source	Estimated total N input (t N/d) Most Probable	Upper Limit
Inputs		
Sewage		
Greater Victoria, BC	3.3-3.5	5-6
Greater Vancouver, BC	20-22	
Seattle, WA	15-16	
Total sewage inputs (Victoria, Vancouver, Seattle and other municipalities)	65-70	100
Fraser River (excluding sewage)	50-60	
Total sewage + river	115-130	150-160
Aquaculture	6	
Oceanic deep-water inflow	2500-2800	
Export		
Oceanic surface-water export	2000-2100	

and mine tailings (Harrison et al. 1994). In contrast, it is not clear whether the high abundance of phytoplankton in the south end of Sechelt Inlet is due to the natural shallowness of the area allowing for increased mixing or a mild case of eutrophication due to salmon farms in the area (Harrison et al. 1994).

In recent years, concern has been raised about impact of the discharge of sewage from Victoria following only preliminary treatment (i.e., screening the effluent through 6-mm wire mesh), and, to a lesser extent, Vancouver's sewage effluent. In 1993, Victoria's sewage outfalls located at Clover Point (1 154 m offshore) and Macaulay Point (1 800 m offshore) together discharged approximately 98 000 m^3/d of sewage, representing an N load of 3.3 to 3.5 t N/d (Table 4.4; Taylor et al. 1995). Natural deep-water oceanic N inputs to the Juan de Fuca Strait in the vicinity of Victoria (2 500-2 800 t N/d) exceed sewage inputs by at least 400-fold (Table 4.4; Harrison et al. 1994). Thus, the Clover and Macaulay Point outfalls have little effect on surface water quality in terms of N concentration and hence phytoplankton abundance (Taylor et al. 1995). There is evidence, however, of negative effects of the effluent on benthic invertebrate communities around the outfalls, especially from contaminants (EVS Consultants 1992, Taylor et al. 1995). Similarly, Vancouver's 1992 N inputs from municipal wastewater were estimated to be 21.5 t N/d (Table 4.4; Harrison et al. 1994). Once the natural diffusion of the ocean's deep water into the interior of the Georgia Basin are considered, natural N inputs exceed those of Vancouver's sewage by about 100-fold (Table 4.5; Harrison et al. 1994). Following the1988 extension of the effluent pipe at Vancouver's Iona Island sewage treatment plant to 4 km into the Strait of Georgia in 1988, improvements were observed in Sturgeon Bank. There has been a shift from a benthic algal community dominated by blue-green and green algae to one dominated by diatoms. The density of benthic invertebrates has also increased, as have the oxygen concentrations in overlying waters (Environment Canada 1998a). In 1998, Vancouver upgraded the Annacis and Lulu Island sewage treatment plants to secondary treatment to minimize sewage impacts on the Fraser River and the Georgia Basin in the vicinity of Vancouver.

Currently, it appears eutrophication of the Georgia Basin is not widespread. This situation will likely remain as long as anthropogenic nutrient loading does not increase substantially and regulation of the Fraser River does not reduce flow to the point where estuarine circulation is impaired. Impairment of the natural hydrodynamics that flush the area on a daily basis would likely enhance eutrophication, as anthropogenic nutrient sources would not be washed from the basin.

Martha Guy
Environment Canada

Halifax Harbour, Nova Scotia: 200 years of Pollution

Halifax Harbour is a long, irregularly shaped, narrow estuary surrounded by Halifax Regional Municipality, which encompasses the former cities of Halifax and Dartmouth, the town of Bedford, and Halifax County Municipality (Figure 4.9). In 1996, the population of the area was about 343 000 (Halifax Regional Municipality 1998).

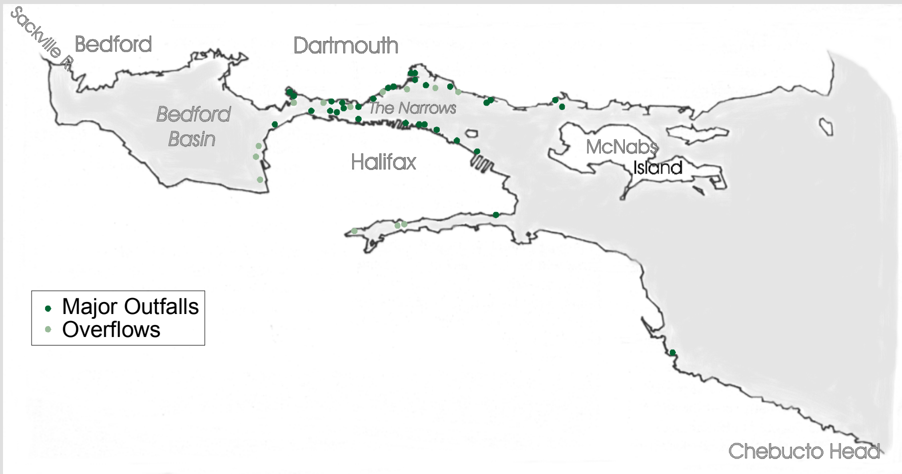

Figure 4.9. Halifax Harbour showing locations of major sewer outfalls and overflows. Adapted from EP-EARP 1993.

Water circulation in Halifax Harbour is typical of estuarine systems with freshwater, primarily from the Sackville River, mixing with surface seawater to create a two-layered flow system (FP-EARP 1993). The upper, less saline water layer (approximately 10m thick) flows toward the Scotian Shelf at rates up to 2.6 cm/s (Petrie and Yeats 1990). In the outer harbour, deep saltier water flows up the inlet toward the head of Bedford Basin at rates of 1-3 cm/s (Petrie and Yeats 1990). The strongest horizontal currents occur in the Narrows and in the McNabs Island area, whereas the most vigorous vertical mixing between the two layers occurs in the Narrows and downtown Harbour area (Petrie and Yeats 1990). The Harbour's two-layer mixing regime is modified by many factors including tides, winds, atmospheric pressure changes, and variations in the volume of freshwater input and rates of vertical mixing (FP-EARP 1993). A sill in the Narrows impedes water exchange within the deep (70 m) Bedford Basin, resulting in naturally higher nutrient concentrations in this area. Consequently, the bottom water of the Basin may become anoxic during years of strong summer stratification (FP-EARP 1993).

Europeans first settled the area around Halifax Harbour approximately 250 years ago and since then, raw sewage and industrial waste have been dumped into its waters. This activity has resulted in a build-up of organic material and metals in the bottom sediments. Currently, sewage is discharged at a rate of 2.1 m^3/s through 20 to 30 outfalls dispersed around the Harbour resulting in a daily load of 0.17 t NO_3-N and 1.6 t P (Table 4.5). This load is slightly larger than NO_3-N load but much larger than the P load from the Sackville River. Taken together the NO_3-N loads from these two inputs are smaller than that contributed by deep oceanic water: however, in the case of P, the Halifax sewage load is similar to the oceanic input (Table 4.5). Petrie and Yeats (1990) concluded sewage makes a negligible contribution to the NO_3-N budget of the harbour but a significant contribution to its P budget.

Halifax Harbour is not a pristine ecosystem. After two centuries of serving as receiving water for domestic and industrial waste, the environment has been contaminated. The complex nature of the system, however, makes it difficult to assess the impacts of added nutrients (DFO/DEMR/DOE 1989). Evidence of eutrophication exists: Bedford Basin experiences periodic fish kills in late summer, presumably due to anoxia from increased oxygen demand associated with decaying organic matter. However, the most recent fish kill in the Harbour (August 1993) was correlated with a dinoflagellate bloom, a class of algae not a normal component of phytoplankton succession in the Harbour. This unusual summer algal bloom may have been stimulated by elevated nutrients brought on by an extended period of calm weather near the end of a drought (P. Strain, DFO, personal communication).

Halifax Harbour concluded

Table 4.5. Estimated nitrogen and phosphorus inputs to Halifax Harbour. The range in oceanic inflows reflects differences in nutrient concentrations during low and high flows. Surface export was calculated for low flow only.

Source	**Estimated Nitrate input (t N/d)**	**Estimated total P input (t P/d)**
Inputs		
Sewage [1]	0.17	1.6
Total river inputs [2]	0.04	0.001
Oceanic inflow [1]	0.53 - 10	0.56 - 2.2
Export		
Surface export	0.78	0.81

1 Data from Petrie and Yeats (1990)
2 Data from J. Dalziel, Bedford Institute of Oceanography, unpublished data

Although these events suggest an excess of nutrients in the basin, they may also reflect natural fluxes of nutrients with the tide; however, the waste inputs into the Harbour are likely contributing to the severity, timing, and nature of the fish kills (P. Strain, DFO, personal communication).

Presently, Halifax Regional Municipality is moving toward arresting the discharge of raw sewage into Halifax Harbour. Plans to build a sewage system that would connect to a primary wastewater treatment plant, from which treated effluent would be discharged through one diffuser located west of McNabs Island stalled due to cost (FP-EARP 1993; P. Strain, DFO, personal communication). Concern exists, however, that any decrease in nutrient loading to the Harbour will decrease the organic content of the bottom sediments, thereby eliminating the strongly reducing environment that currently exists and allowing remobilization of heavy metals (DFO/DEMR/DOE 1989).

Martha Guy
Environment Canada

blooms' has resulted in closure of shell fisheries resulting in large economic impacts on coastal communities (Paerl and Whitall 1999). Although individuals of the harmful algae species are always present in low concentrations, the exact conditions that cause them to bloom is not clear although blooms do tend to occur following periods of intense rainfall, runoff and intense irradiance (Smayda 1997). For example, laboratory experiments with the diatom *Nitzschia pungens* showed that production of domoic acid, the toxin responsible for amnesiac shellfish poisoning from mussel consumption, was greatest under nutrient (other than nitrate) limitation (Taylor et al. 1994). Another theory currently holds that these blooms may result from changes in the ratio of silica to P or silica to N due to increased terrestrial freshwater runoff (Paerl 1997).

Shifts in the taxonomic composition of the vegetative community also affect the grazer community. Slow-growing seagrasses decompose slowly and experience only moderate losses to grazers. In contrast, phytoplankton production is transferred much faster to heterotrophs, either through increased grazing or decomposition (Duarte 1995). The loss of seagrasses in the coastal zones also means a loss of habitat for many fish and benthic organisms that are important grazers on the phytoplankton. In addition, both anoxia and hypoxia are becoming more prevalent in many estuaries and coastal seas. Increased anoxia in deep water decreases fish egg survival as well as decreasing the area of available habitat for reproduction (Kerr and Ryder 1992). This habitat loss ultimately results in a notable shift from a fish community dominated by the larger, benthic fish species to one composed primarily of smaller, pelagic fish species (Kerr and Ryder 1992).

In the last 20 years, the causes and extent of coastal eutrophication have been increasingly recognized as a global problem (Howarth 1988; Vollenweider 1992; NRC 1993a; United Nations Environment Program 1995; Paerl 1997). Nitrogen addition to coastal waters has increased substantially as a result of human activity: by 25% in the case of N loading from the St. Lawrence River to coastal waters (Howarth et al. 1996). Globally, atmospheric deposition is estimated to contribute approximately 40% of the total loading of new N per year to coastal areas with riverine discharges contributing another 30%, ground water contributing an additional 10% and biological nitrogen fixation adding 20% of new N (Paerl and Whitall 1999). Considering the entire coastline of the North Atlantic Ocean, non-point sources of N are approximately nine-fold greater than inputs from wastewater treatment plants (Howarth et al. 1996). Phosphorus loading has also increased to coastal waters and while non-point source inputs are a significant source of P, point sources may be high in many environments (Caraco 1995; Carpenter et al. 1998). On a more local scale, the expansion of the aquaculture industry has raised concerns about its contribution to eutrophication. Available evidence suggests the potential for local effects immediately underneath cages (see Marine Aquaculture case study).

4.6. Ground water

Nitrate contamination of ground water is usually due to over-application of fertilizer or manure on agricultural land, domestic waste from septic disposal systems, and losses of soil organic N produced by plough down of old meadows, legumes and other crop residues. However, under fertilization of crops can also result in nitrate leaching losses. These losses can be as large as those observed for over fertilized crops because crop growth is impaired leaving a sizeable amount of moisture in the soil. This extra moisture fosters microbial activity resulting in larger net N mineralization (Campbell et al. 1993). Nitrate is highly mobile in soil (Bohn et al. 1985) and, if not assimilated by plants, migrates readily into ground water. Once in ground water, the bacterial reduction of nitrate to nitrous oxide or nitrogen gas is the main potential mechanism for the removal of nitrate. Bacteria capable of denitrification are commonly found in many subsurface environments (Korom 1992). These bacteria require both anaerobic conditions and suitable electron donors such as organic carbon or reduced iron or sulfur for denitrification to occur. The amount of natural organic carbon in subsurface sediments is often very low (Mackay 1990); thus nitrates may persist in ground water due to the absence of a suitable energy source for denitrification. In contrast to nitrate, ammonium is adsorbed onto clay minerals and is not as prone to leaching; however, it is subject to bacterial oxidation to nitrate and may be readily mobilized in aerobic environments. Normally ammonium concentration in soils is very low. Phosphorus movement into ground water is retarded through soil adsorption and mineral precipitation reactions in the unsaturated zone (Wilhelm et al. 1994a; Harman et al. 1996).

Contamination of aquifers with nitrate from nitrogen fertilizers or spreading of animal manure occurs mostly in humid or irrigated areas where sandy soils overlie shallow aquifers (Henry and Meneley 1993). The risk depends on the N input to the land surface and the degree to which an aquifer is vulnerable to nitrate leaching and accumulation. In permeable soils that overlie unconfined, sand or gravel aquifers, leaching of nitrate is likely to occur when the soil moisture capacity exceeds field capacity. Aravena et al. (1993) noted an average concentration of 160 mg/L nitrate-N is typical of shallow ground water in unconfined aquifers overlain by heavily fertilized soils in southern Ontario. However, in clay and clay loam soils that drain slowly after a heavy rain or irrigation application, denitrification can be significant in reducing nitrate in the soils and can prevent leaching to ground water. For example, studies of manure application under dryland farming practices on loam to clay loam soils in central Alberta showed nitrate did accumulate and move down the soil profile under annual manure applications of up to 120 t/ha, but nitrate was not detected in the ground water at depths of 2.5 m or more after several years of study (Chang and Entz 1996, Olson et al. 1997a). Olson et al. (1997a) hypothesized that continued annual application of high rates of manure would cause an accumulation of soil nitrate, with excess nitrate leaching deeper into the soil profile and eventually reaching the water table. Under irrigated conditions, Chang and Entz (1996) found nitrate leached to ground water for annual manure applications of 60 t/ha. No one crop system or tillage method (zero versus conventional tillage) appeared to favour excess nitrate-N accumulation and potential leaching (Sommerfeldt and Chang 1985, Olson et al. 1997b;). A study of feedlots in Alberta found that although levels of N and P were increased in the soil, there was little evidence of extensive downward movement of the nutrients except under one feedlot in a slight depression (Sommerfeldt et al. 1973). Similarly, a lack of increase in soil nitrate concentration beneath a manure storage area in British Columbia was attributed to concentrations of free ammonia in the soil that were toxic to nitrifying bacteria (Zebarth et al. 1999). Extractable soil P concentrations were also higher under the manure storage area. However, instrumentation was not installed to monitor potential leaching of ammonium and P to the ground water at the study site.

Septic systems contribute the largest volume of effluent discharged to ground water in North America (Cherry and Rapaport 1994). Although there are no national statistics on sewage disposal systems in Canada, approximately 25% of the Canadian population is estimated to be served by septic disposal systems. Densities in some areas have increased to the extent that ground water degradation caused by septic systems has become an environmental issue (Harman et al. 1996). Nitrate concentrations at the water table beneath the tile beds of septic disposal systems are roughly 20 to 70 mg/L nitrate-N (Wilhelm et al. 1994b). Septic systems have relied on dilution and dispersion to reduce high concentrations of nitrates. In sand and gravel aquifers and fractured bedrock, low dispersion often maintains high nitrate concentrations for considerable distances (up to several hundred metres) from the drain field. The Cambridge, ON case study provides information about nutrient loading from a domestic septic system.

Nitrogen released from soil organic matter is also a significant source of nitrate for possible leaching to ground waters. Soil organic matter constitutes the main source of N in the soil system. A soil with 5% organic matter will contain about 5 600 kg/ha of N to a depth of 15 cm (Henry and Meneley 1993). Except for intensively managed grasslands (particularly grazed grasslands), perennial grass stands allow very little N movement below the root zone. However, when grass stands are ploughed to return to annual cropping, large leaching losses of nitrate are possible. The presence of bare fallow conditions can significantly increase the amount and rate of nitrate lost below the root zone and potentially add to ground-water nitrate supplies (Henry and Meneley 1993). Leaching, especially of nitrate, can also occur in relatively low-intensity dryland farming where summerfallowing is used. Generally under dryland agriculture, water and nitrate will move below the root zone in early spring when evaporation is low (Stewart 1970). In the Canadian prairies, the high moisture deficit has fostered wide use of summerfallow to ensure water reserves in the soil. During the fallow year in southern Saskatchewan sizeable amounts of nitrate can be leached from the soil, especially in rainy years. In fact, Campbell et al. (1984) estimated that in a year with 23% more precipitation than the long-term average during the growing season, at least 123 kg nitrate-N/ha were leached below 240 cm depth.

Domestic Septic System, Cambridge, Ontario

An intensive study of a contaminant plume from a single-family septic system near Cambridge, Ontario was begun in 1987 by the University of Waterloo and results for nutrient (nitrate and phosphate) contamination at the site have been reported in several papers (e.g. Robertson et al. 1991; Robertson and Cherry 1992; Cherry and Rapaport 1994; Shutter et al. 1994; Wilhelm et al.1994a; Robertson 1995). The site is located on a sand plain where 4 to 8 m of fine to coarse sand overlies a low permeability silt till. Household wastewater is discharged to a holding tank and tile bed that lie at a depth of 0.6 m near the base of the soil zone. The water table is 2 to 2.5 m below the ground surface.

Nitrogen concentrations in the Cambridge effluent average 32 mg/L of ammonium-N and 1.3 mg/L of nitrate-N when it leaves the septic tank (Wilhelm et al. 1994a), values considered typical for wastewater. Between the tile bed and the water table below the tile bed, the ammonium in the effluent is completely oxidized to nitrate in the oxygen-rich unsaturated sand below the tiles. Within the saturated zone, appropriate conditions (i.e. anaerobic environment, carbon source) for removal of nitrate by denitrification are not present; therefore, the nitrate plume persists in the ground water and is very mobile. Furthermore, dilution and dispersion of the nitrate in the plume are inhibited by a background concentration of about 28 mg N/L in the ground water due to the use of agricultural fertilizers on the overlying land. Compared to the N load from agricultural fertilizers around Cambridge, the contribution of nitrate from this septic system to the ground water is relatively small (Wilhelm et al. 1994a).

Phosphate-P levels in the Cambridge effluent averaged 6.3 mg/L for 19 samples collected from 1987 to 1994 and ranged from 1.4 to 14.2 mg/L (Robertson 1995), values also considered typical for domestic wastewater. Approximately 25% of phosphate is attenuated in the unsaturated zone. Attenuation in the unsaturated zone seems to be controlled by phosphate mineral solubility that is established relatively quickly after the effluent enters this zone. Detailed monitoring in the saturated zone has demonstrated the existence of a steadily expanding phosphate plume. Phosphate migration appears to be controlled by sorption processes which retard migration but which provide little further attenuation of the levels that persist. Phosphate is migrating at a rate of about 1 m/yr (Robertson 1995).

The system at Cambridge functions according to design specifications. The average ground-water velocity in the area of the tile beds has been estimated to range from 24 to 40 metres per year. A long narrow plume more than 130 m long and about 10 m wide is defined at the site. A sharp concentration gradient at the edges of the plume implies weak mixing and, therefore, dilution of the plume by the ground water. Shallow domestic wells that lie in the path of a septic system plume may be affected even at significant distance down-gradient from the source (Shutter et al. 1994).

Gary Grove
Environment Canada

Ground water/surface water interactions have not been extensively studied in Canada; therefore, potential impacts from discharge of contaminated ground water are not completely known. However, inputs of N-enriched groundwater to coastal and certain inland waters and P-enriched groundwater to inland waters may contribute to eutrophication. In a study of a septic system plume that had begun to discharge to a nearby river, Robertson et al. (1991) observed almost complete nitrate attenuation within the last two metres of the plume flowpath before discharge to the river. The attenuation was attributed to denitrification occurring within the organic rich riverbed sediments. Several septic system plumes in a barrier sand bar in Point Pelee National Park were investigated for potential interactions with Lake Erie and a large coastal freshwater marsh behind the sand bar (Crowe et al. 1998). Low but measurable concentrations of ammonium and phosphate were measured in the ground water at the edge of the marsh at these sites (Ptacek 1998; Ptacek et al. 1998). However, detailed studies of water levels and stable isotopic composition of ground water indicate relatively little lateral movement of ground water at some sites and weak to moderately strong seasonal flow reversals at others (Crowe et al. 1998, Huddart et al. 1999); therefore, the contribution of nutrients in the septic system effluent to eutrophication in the marsh ponds is neither entirely known nor necessarily significant. In Alberta, P from septic disposal systems is thought to comprise 4 to 6% of the total P budget of several lakes (Sosiak and Trew 1996; Mitchell 1999). However, in one extreme case, namely Jackfish Lake near Edmonton, P from septic disposal systems may represent up to 67% of the total P supply to the lake. This lake has a small watershed, many cottages and little internal loading.

4.7. Overview of Enrichment in Canadian Ecozones

To assess the geographic extent of eutrophication, information was compiled on water quality of Canadian lakes, river reaches and coastal waters. Review of these reports and papers and consultation with water quality managers showed that regional variation in natural water quality, soil types, climate, landuse, and human impacts resulted in very different impressions of nutrient-related water quality impairment. Even polling of experts at an OECD workshop in 1974 resulted in definitions of lake eutrophy ranging from 38 to 189 mg P/m^3 (mean = 84.4 mg P/m^3) and 861 to 4 081 mg N/m^3 (mean = 1 875 mg N/m^3; Janus and Vollenweider 1981). Most provinces and territories have P or algal guidelines against which to evaluate eutrophication of inland waters; however, there are currently no provincial or national criteria established for N concentrations to control excessive aquatic plant growth in coastal bays and estuaries. Our identification of nutrient-related water quality impairment was based on the trophic status provided by the investigator(s).

British Columbia contains portions of five ecozones; the Pacific Maritime, Montane Cordillera, Boreal Cordillera, Taiga Plains and Boreal Plains. In general, British Columbia enjoys excellent surface water quality with respect to nutrients for fresh and marine waters in all five ecozones. However, the Fraser River Basin, which covers almost one-quarter of the province and straddles the western edge of the Montane Cordillera and the eastern edge of the Pacific Maritime ecozones, shows signs of nutrient enrichment in its lower reaches (Figure 4.10). In this area, agricultural runoff and municipal effluents discharged to the Fraser River and its tributaries (e.g., the Sumas River) combine to create enrichment problems. It has been estimated that 90% of the Province’s municipal discharges occur in the Lower Fraser River Basin (Government of Canada 1996). There is little direct evidence of enrichment of coastal marine waters in British Columbia, due to high flushing by nutrient rich water from the open water (see Georgia Basin case study). Only poorly flushed bays show signs of eutrophication, such as low oxygen and high algal biomass.

In the Prairies ecozone of southern Alberta, Saskatchewan and Manitoba, most of the large rivers originate in the Rocky Mountains and flow east to Hudson Bay. As they flow east, the rivers serve critical functions such as power generation, water supply and recreation; all have been regulated. Water quality of the large mountain-fed rivers is generally good, except downstream of large cities and in areas of intensive agriculture (e.g., Old Man River, Alberta; Figure 4.10) (Government of Canada 1996). By comparison, rivers that arise in the Prairies (e.g., Souris River) are usually eutrophic as a result of both natural and agricultural inputs. Lakes in the Prairies ecozone tend to be small, shallow and, in the case of small waterbodies, ephemeral. Nitrogen and phosphorus concentrations tend to be naturally high; intensive agriculture in many watersheds has further elevated nutrient concentrations. A recent program of extensive water quality surveys and intensive research undertaken by the

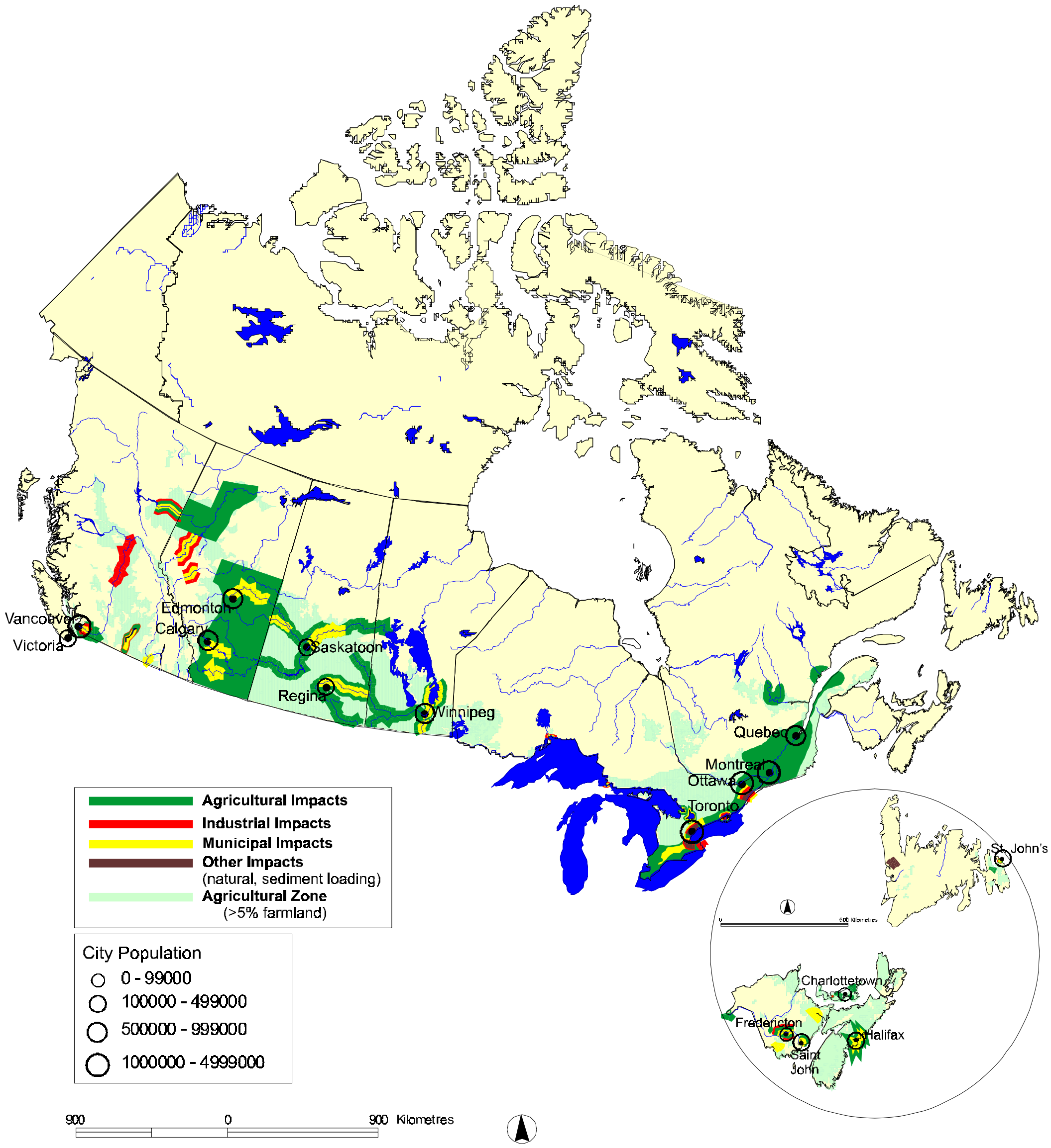

Figure 4.10. Documented sites of nutrient enrichment in Canada, 1998. Data reflects the both the extent of enrichment and the extent of monitoring undertaken to detect enrichment.

Data Sources:

BC	Government of Canada 1996; BCELP 1999; Harrison et al. 1994; French and Chambers 1995
AB	CAESA 1998; Environment Canada 1998c
SK	Environment Canada 1998c
MB	Environment Canada 1998c
ON	D. Boyd, Ontario Ministry of the Environment, personal communication; A. Gemza, Ontario Ministry of the Environment, personal communication
Great Lakes	Environment Canada 1999c; Government of Canada 1996
QC	Government of Canada 1996; Painchaud 1997
NB	Jerry Choate, NB Environment, personal communication
NS	Darrell Taylor, NS Dept. of the Environment, personal communication; Strain 1998
PEI	Bruce Raymond, PEI Technology and Environment, personal communication
NF	Gerry Collins, NF Dept. of Environment and Labour, personal communication

governments of Alberta and Canada showed that in Alberta, N and P often exceeded interim provincial water quality guidelines for the protection of aquatic life in streams in high and moderate intensity agricultural areas (CAESA 1998). The P guideline was also exceeded in small lakes in high intensity agricultural areas.

The Boreal Shield and Boreal Plains ecozones together cover nearly 2.7 million square kilometres of land and fresh water and include portions of seven Canadian provinces. The rivers and lakes in these ecozones account for 29% of Canada's freshwater surface area (Government of Canada 1996). Overall, water quality is very good. Nutrient concentrations in the eastern portion of the Boreal (i.e., the Boreal Shield) are naturally very low as the underlying rock throughout most of the ecozone is granite and provides little nutrients to overlying soils and water. In the Western Boreal Plains, nutrient concentrations are naturally somewhat higher due to thick glacial soils underlying the rivers and lakes (Cooke and Prepas 1998). Nutrient-caused water quality impairment in the Boreal ecozone is rare and when it does occur, relates to specific anthropogenic stressors (Figure 4.10). For example, rapid development in Ontario's Cottage Country (Haliburton and Muskoka lake districts) during the 1980s resulted in an increase in the concentration of P in many lakes. This increase was largely caused by increased runoff due to land clearing and construction. Since the construction boom has ended and cottage lots have been landscaped, total P values have decreased to values observed pre-development (Andy Gemza, Ontario Ministry of the Environment, personal communication). In 1997, only 20 out of 318 Ontario cottage lakes sampled by the Lakes Partner Program had bays or inlets with total P values greater than 21 µg/L P, the nutrient guideline used in the program to designate them as eutrophic (OME 1998). The majority of these lakes were in the Lake Ontario drainage basin. Similarly, investigations conducted under the auspices of the Northern River Basins Study identified zones of enrichment downstream of specific municipal or industrial outfalls to the Athabasca and Wapiti rivers of northern Alberta (Chambers 1996; Wrona et al. 1996; Chambers et al. 2000).

Agricultural runoff, municipal sewage and industrial wastewater have had a substantial impact on the loading of N and P to rivers and lakes in the Mixedwood Plains ecozone of southern Ontario and Québec. For example, Lake Simcoe and Rice Lake, two large lakes in southern Ontario, have both experienced nutrient impairment due to agricultural and municipal development in their basins (Figure 4.10). The effects of municipal and agriculture development are especially apparent in the Lake Erie basin where some of Canada's most intensive agricultural activity combines with intensive urbanization. Many of the rivers in southwestern Ontario (e.g., the Thames, Grand, Don and Humber Rivers) have total P concentrations that exceed Ontario's provincial water quality objective for rivers of 0.03 mg/L and are thus considered eutrophic (Basu and Pick 1996; D. Boyd, Ontario Ministry of the Environment, personal communication). Although the rivers themselves show varying degrees of excessive plant growth, the cumulative effects of non-point agricultural inputs and municipal and industrial point-source inputs can be seen along the shorelines of the Great Lakes (see Lake Erie case study).

Rivers and lakes in the agricultural basins of the St. Lawrence Lowlands in Québec also reveal the effects of high nutrient inputs. Recent water quality surveys in southern Québec showed that most rivers in the Lowlands had total P concentrations close to the provincial guideline of 0.03 mg/L TP and that agriculture was a major source (Painchaud 1997). In a study of the Yamaska River, 16% of the ammonium-N samples exceeded the guideline of 0.5 mg/L (see the Yamaska River case study). Time-series analysis of water quality data collected between 1988 and 1998 for rivers in southern Québec showed that total P and nitrate concentrations decreased during that period (Simard and Painchaud

2000). This downward trend resulted from the construction of over 500 wastewater treatment plants during the 1980s and 1990s, as well as a reduction in the use of agricultural fertilizers. Although loading from municipal and industrial sources has greatly decreased through the 1990s, agricultural loading remains important. The long-term accumulation of P in agricultural soils as well as the current growth of intensive agricultural operations, such as hog and corn farms, is a source of concern for water quality in the province. Urban sources are still evident in the St. Lawrence River in the form of increased ammonia levels downstream of areas where treated wastewater, storm sewer overflows and raw wastewater are discharged (Hudon and Sylvestre 1998).

In the case of the Great Lakes, concern during the 1960s about the degradation of water quality in the lower Great Lakes (Lakes Erie, Ontario and portions of Lakes Michigan and Huron) led to the signing of the Canada-United States Great Lakes Water Quality Agreement (GLWQA) in 1972. The aim of the GLWQA was to “restore and maintain the chemical, physical, and biological integrity of the waters of the Great Lakes Basin Ecosystem” (Government of Canada 1996). In 1985, the International Joint Commission Great Lakes Water Quality Board identified 42 Areas of Concern (AOCs) around the lakes, 12 of which are in Canadian waters and five are shared with the United States on connecting river systems (Environment Canada 1999c; Figure 4.10). Of the 12 Great Lakes Basin AOCs in Canada, eight have impaired water quality due to excess nutrient concentrations or undesirable algal growth. One of the eight sites is on Lake Superior, two are on Lake Huron, three are on Lake Ontario, and the remainder are on the St. Clair and St. Mary’s rivers. All but two are areas where excess nutrients enter harbours from tributaries draining agricultural land and then passing through intensive urban areas. The nutrients mix into and accumulate in the relatively static waters of the harbour. The 1987 revision of GLWQA formalized the establishment of Remedial Action Plans (RAPs) (Environment Canada 1999c). Restoration of beneficial uses within the Areas of Concern (AOCs) is the primary mission of RAPs, and is an essential step in restoring the integrity of the Great Lakes Basin ecosystem. The RAP program emphasizes adoption of a systematic and comprehensive ecosystem approach to restoring and protecting beneficial uses and the area's ability to support aquatic life (Environment Canada 1999c). This process has resulted in decreased nutrient loading from tributaries, although in some cases P concentrations remain high due to internal loading from harbour sediments and/or insufficient reduction in nutrient loads. Successes have been achieved through the RAP program with one AOC previously identified as having nutrient impairment, Collingwood Harbour, now considered restored (Environment Canada 1999c).

The Atlantic ecozones include the terrestrial Atlantic Maritime ecozone and the marine Atlantic and Northwest Atlantic ecozones and encompass the provinces of Nova Scotia, New Brunswick, and Prince Edward Island. This area of Canada has been subjected to intensive alteration in its almost 400 years of European settlement. For many of the large cities, untreated sewage has been discharged to coastal waters for over 250 years. Although this long history of untreated sewage discharge has caused a variety of problems, the complex nature of the coastal environments, high flushing rates and inputs of nitrogen-rich deep oceanic waters make the impacts of anthropogenic nutrients difficult to quantify. A recent assessment of water quality data collected from 34 Maritime inlets showed that the harbours can be arranged along a nutrient gradient; however, this gradient likely relates to the N and P content of the incoming ocean water rather than anthropogenic inputs (Strain 1998). Inland waters throughout the region have been affected by agriculture and industrial development and show varying degrees of eutrophication (e.g., Saint John River, NB; Boughten River, PEI; Black Brook, NB; Figure 4.10).

4.8. Conclusions

Changes in species abundance and dominance due to nutrient addition from human activities are a pervasive problem in Canadian ecosystems, particularly for those systems that are sites of intensive settlement, agriculture or industrial activity.

Fertilization studies involving single N applications have confirmed that N is the major limiting element for tree growth in many Canadian forest ecosystems. In recent decades, atmospheric deposition of N has been found to have steadily increased in parts of Canada and, because most of Canada's forests are N limited, this increased N deposition has likely increased the productivity of forest vegetation (Table 4.6). Controversy exists, however, as to whether the productivity gains resulting from increased N deposition along with a warming climate and elevated CO_2 levels in the atmosphere have offset the carbon loss caused by timber harvest, fire and insect-induced mortality. Atmospheric deposition of nitrate has also resulted in the forests surrounding Turkey Lake in central Ontario and Lac Laflamme in southern Québec being in an early stage of N saturation (Jeffries 1995).

Lakes in developed watersheds often have elevated nutrient concentrations, which increase the abundance of planktonic and sometimes filamentous algae and, in turn, change the abundance and composition of invertebrates and ultimately fish (Table 4.6). The increased organic matter production resulting from nutrient enrichment can also increase the abundance of bacteria, and their metabolic activity can lower oxygen concentrations in the water to the extent that fish survival is threatened. Although these effects will occur throughout a lake, they will be intensified near the nutrient source, e.g. near sewage or municipal wastewater discharges or near tributary inflows. Similar enrichment effects have been observed for rivers draining populated or agriculturally intensive lands. Here river volume, and thus the extent to which the pollutant is diluted, largely determine the severity of nutrient addition. Moreover, in rivers, effects of nutrient addition may be compounded when one input after another is added to the river before the effects of more upstream inputs are attenuated. This accumulation of nutrients is of particular concern for northward flowing rivers where settlement and industrial development tend to occur in the southern headwaters and pollutants can thus travel hundreds of kilometres through terrain that has otherwise experienced minimal human impacts. However, it should be noted that the changes in the food web that occur in response to nutrient addition are not always perceived to be negative. For example, nutrient-poor coastal lakes and rivers in British Columbia are purposely fertilized to enhance the food supply and thereby increase the size of juvenile and spawning salmon.

Marine and freshwater wetlands in Canada also show evidence of enrichment. Although the major cause of wetland loss is their conversion to agricultural or urban land, nutrient addition can cause changes in the composition of emergent vegetation and, thereby, affect the quality of water transferred to adjacent ponds, lakes and coastal waters (Table 4.6).

Coastal waters on both the Pacific and Atlantic shores of Canada have had a long history as receiving waters of untreated sewage. Although a number of environmental issues have been associated with marine discharge of untreated sewage, the complex nature of coastal environments, high flushing rates and inputs of nitrogen-rich deep oceanic waters have generally minimized any widespread effects of anthropogenic nutrients. Certain small bays and harbours where oceanic exchange is limited, do however, show evidence of eutrophication.

Table 4.6. Nutrient enrichment: a summary.

Potential enrichment effects on: Water/soil quality and characteristics	Plants	Animals	Human health and economy
Forests (Section 4.1)			
• N saturation is associated with changes in soil chemistry, such as loss of nutrient cations, an increase in soil acidity, and an increase in the availability of aluminum	• increased biomass of trees and other vegetation in forests • ultimately, a decline in forests if soil quality deteriorates		
Lakes (Section 4.2; Lake Erie, Qu'Appelle Lakes and Lac Heney case studies)			
• elevated nutrient concentrations • reduced water clarity • depletion of oxygen	• increased phytoplankton, filamentous algae and/or aquatic weeds • increased abundance of potentially toxic blue-green algae	• increased productivity of benthic organisms and fish at moderate nutrient levels • at high nutrient levels, reduced biodiversity of benthic organisms due to lowered oxygen; reduced survival of fish eggs due to oxygen depletion; changes in biodiversity of fish and benthic organisms.	• degraded shorelines and reduced recreational use due to nuisance algae/weeds; • loss or impairment of fisheries • increased costs for drinking water filtration; taste/ odour problems; blockage of water intakes • toxins produced by blue-green algae can poison livestock and people
Rivers (Section 4.3 and 5.2; Alberta's Northern rivers and Yamaska River case studies)			
• elevated nutrients concentrations • reduction in oxygen concentrations	• increased benthic algae, filamentous algae, planktonic algae and/or aquatic weeds	• increased productivity of higher trophic levels (e.g., insects and fish) at moderate nutrient levels • reduced productivity by benthic organisms and fish and loss of species at high nutrient levels.	• reduced recreational use due to nuisance algae/weeds; • increased costs for drinking water filtration taste/ odour problems; blockage of water intakes
Freshwater Wetlands (Section 4.4 and 5.2; Delta Marsh and Cootes Paradise case studies			
• elevated nutrient concentrations • water balance changes due to increased evapotranspiration rates caused by changes in plant biodiversity.	• bogs: decline in *Sphagnum* and replacement with more nitrophilous mosses; • fens: increase in tall graminoids (grasses, sedges) and loss of shorter plant species that are less able to compete for light; • marshes: increase in phytoplankton and emergent vegetation	• bogs, fens: changes in wildlife and waterfowl biodiversity if changes in plant biodiversity cause loss of habitat or food supply	

Table 4.6. Nutrient enrichment: a summary (concluded).

Potential enrichment effects on: Water/soil quality and characteristics	Plants	Animals	Human health and economy
Salt Marshes (Section 4.4; Bay of Fundy and Fraser River Estuary case studies)			
• elevated nutrient concentrations • water balance changes due to increased evapotranspiration rates caused by changes in plant biodiversity.	• increase in phytoplankton that block light penetration to deeper vegetation. • loss of plant diversity	• nutrients released from salt marshes to adjacent estuaries may stimulate coastal water food webs. • changes in wildlife and waterfowl biodiversity if changes in plant biodiversity cause loss of habitat or food supply	
Coastal waters (Section 4.5; Georgia Basin and Halifax Harbour case studies)			
• increased nutrient concentrations • deep-water anoxia due to increased organic loading • turbidity due to loss of seagrasses to stabilize bottom sediments	• increase in phytoplankton and macro-algae • increased abundance of algae that produce toxic chemicals. • reduction in seagrasses due to reduced light penetration and sediment anoxia	• loss of habitat for fish and benthic organisms	• four different illness in human consumers of molluscs; • large economic impacts on coastal communities as a result of closures of shellfish fisheries;.
Ground water (Section 4.6; Domestic Septic System case study)			
• increased nitrate and P concentrations in certain aquifers			• nitrate contamination of aquifers used as drinking water source

Elevated nutrient concentrations, particularly nitrate, are also found in ground waters (Table 4.6). Aquifers, which are shallow (<30 below the surface) and situated in very sandy, porous soil, are particularly vulnerable to nitrate contamination. Nitrate contamination of ground water is a concern to consumers of this drinking water source and may also contribute to the eutrophication of inland and coastal waters.

Nutrient addition and its consequent effects on biodiversity are found across Canada, wherever intensive agriculture, urbanization or industrialization occur. Areas of concern with respect to nutrient addition are the forests, lakes and rivers of the Lower Fraser Valley; the surface waters of the southern Prairie Provinces; the rivers draining into Lake Erie, Lake Ontario and the St. Lawrence River; and rivers draining agricultural watersheds in the Atlantic Provinces and the estuaries and coastal waters into which these rivers flow.