7.0 Emerging Issues

Highlights

- Phosphorus loads in Lake Erie are now about half the peak loads of the early 1970s. These reductions, in combination with the introduction of the zebra mussel, *Dreissena sp*., have improved water clarity in Lake Erie. However, concerns about deleterious changes to the amount and type of fish harvest have been raised.

- Urea was the chemical most commonly used at Canadian airports to de-ice runways and planes. Because its primary breakdown product is ammonia, which can be toxic to aquatic organisms, airports are now switching to potassium acetate and sodium formate which are less toxic although less effective as de-icing agents.

- In the summer, a phenomenon called “white haze” occurs in southern British Columbia when ammonia from manure volatilizes into the atmosphere and reacts with urban smog particles.

- Fire retardants are essentially fertilizers designed to enhance green plant growth in order to stop the spread of the fire.

- Livestock production has intensified in many parts of Canada. Overall, the number of farms with cows, pigs or poultry decreased by 59% between 1976 and 1996. In contrast the number of cows, pigs or poultry increased by 39%.

- The production of fin and shellfish by Canadian aquaculture operations is expected to double between 1997 and 2005.

In recent years, a number of specific issues related to nutrient levels and their environmental, human-health, or economic consequences have arisen. These range from concerns that reductions in phosphorus loading to the Great Lakes may have impaired the fishery, to the effects of aircraft and runway de-icers and fire retardants on aquatic ecosystem health, to economically-viable and environmentally-safe use of sewage biosolids and livestock manure. This chapter explores the scientific underpinnings and management considerations for these emerging issues.

7.1. Low Phosphorus Concentrations in the Lower Great Lakes

The *Canada-U.S. Great Lakes Water Quality Agreement* (GLWQA) signed in 1972 contained provisions to reduce obnoxious algae problems by reductions in the loading of P. Target P loads were achieved around 1986 mainly by construction of new MWTPs and the adoption of phosphorus precipitation techniques in older MWTPs. The target loads are about one half the peak loads of the early 1970s and lakewide P concentrations have responded proportionately. Recently, the appropriateness of the GLWQA loading targets has been questioned in light of the changing lake ecology brought about by zebra and quagga mussels (*Dreissena sp*.) that invaded the lower lakes beginning in 1988. Fluctuations and declines of some fish species have been noted by the commercial and sport fishery industries. Declines of some fish species are thought to be caused by excessive grazing pressure of the mussels at the bottom of the food chain (i.e. grazing on algae). At the same time, food webs are changing with the emergence of a mussel-goby-bass food chain. Unprecedented low P values have occurred in the east and central basins of Lake Erie since the *Dreissena* invasion. These concentrations were lowest in 1995 and increased in 1996 and 1997. Although water clarity has changed little offshore in Lake Erie after the mussel invasion (Charlton et al. 1998), there has been a noticeable increase in clarity in shallower nearshore areas (Howell et al. 1996) due to the mussels' filtration (grazing) effects. Phytoplankton abundance (measured as chlorophyll concentrations) do not seem to have changed much in the offshore waters of the central and east basins coincident with the *Dreissena* invasion compared to the decrease that occurred earlier due to nutrient reduction. This shift means that although lake productivity is lower than in the early 1970s, the majority of production decrease occurred before mussels arrived. In addition, there may be delayed effects of the achievement of the nutrient load targets that reinforce, or are even more important than, the effects of mussels. The mussels can cause pronounced depletion of algae in nearshore areas (Graham et al. 1996), and this may be where the early life stages of fish are most susceptible to grazing-induced lack of food. At this time, if and how mussels affect fish populations is the subject of ongoing research.

Another issue raised regarding P in the Lower Great Lakes is whether fish populations are at risk from low P. When the P minima occur offshore in summer in Lake Erie, concentrations of P are still higher than those in Lake Huron and this reflects the nutrient load and morphometry of Lake Erie. Lake Huron still has a fishery. Thus, there is probably no threat to the survival of fish in Lake Erie from present P and chlorophyll concentrations alone. The size and type of a sustainable fish harvest may be affected, however.

One solution for perceived low fish production suggested by some fishing interests is relaxation of P removal during winter months at MWTPs. The historical data indicate that nutrient loading reduction caused a decrease in total P of around 3 μg/L in the central and east basins of Lake Erie (Charlton et al. 1998). Therefore, assuming the lake functions as before, a return to nutrient load pollution levels of the 1960s would be needed to increase P by a similar amount. Much of the P load reduction, however, resulted from control of raw sewage and partially treated sewage. Only a portion of the difference between pre-and-post P control loads would be made available, therefore, through any simple shift in technology such as cessation of P precipitation at MWTPs. Most of the sewage P load now under control, which might be made available, is discharged in the west basin. Increasing west basin loads would have little effect on P in the central and east basins unless a return to gross pollution year-round was allowed. Concerns with this idea would be the operational practicality at sewage plants, effects on landowners, drinking water quality, beach quality, and the co-emission of toxic chemicals that may otherwise be removed along with P. In addition, more nutrient loading may have other deleterious effects by stimulating production of blue-green algae that now seem to be favoured in the west basin. Of course, the conditions of tributaries and the effects of more nutrients on shoreline properties already beset by noxious accumulations of *Cladophora sp.* algae would have to be taken into account if an attempt were made to fertilize the nearshore areas or even the lake as a whole. Fishing was apparently adequate in the period 1985 to 1990 when offshore phytoplankton abundance was almost as low as in recent years. Thus, it is doubtful that whole basin nutrient additions from sewage would be sufficient to influence fish production significantly. Scientific consensus is that a change in Great Lakes phosphorus management policy is not advisable at this time.

The present sewage loading limits in the Great Lakes are regulated on an effluent concentration basis. Thus, the load of P can gradually increase due to population growth. Fortunately, more diligent application of present P removal techniques can be applied in order to maintain or lower loads from sewage. Amelioration of local eutrophication problems in "areas of concern" (identified in the GLWQA) would cause a nutrient load reduction to the lake. Non-point sources are numerically very important now and these will decline if objectives to ameliorate conditions in tributaries and to prevent loss of valuable soil are successful. Non-point P loads are mainly poorly available to algae because the P is bound in soils, except immediately following rain when fields have just been fertilized. Thus, although the total P load to the lake may decrease further due to non-point controls, only a small part of that decrease would potentially affect lake productivity. The GLWQA sought to limit productivity of Lake Erie as a way of improving water quality, beach quality and fish habitat while preventing auto-fertilization of the lake from sediment phosphorus regeneration. Some decline in lake productivity would be expected but at this time we do not know whether this will be ameliorated or exacerbated by the food web shift to benthic populations.

7.2. Runway De-icers: The concern over the use of urea at Canadian airports

Every winter, major airports in the Northern Hemisphere are confronted with snow and freezing rain conditions, which play havoc with airline schedules and compromise human safety. To maintain safe conditions for travellers, airport operators apply chemicals to aircraft, taxiways and runway surfaces. The chemicals are selected for their ability to prevent formation of ice on aircraft and runways (anti-icing), or to melt ice and snow when formation has occurred (de-icing). For the past three decades, the chemical most commonly used at Canadian airports as both an anti-icer and a de-icer on runways is urea (Transport Canada 1990). Urea was selected because it does not corrode aircraft parts, stain paints, or affect the plastic of aircraft windows (Transport Canada 1994). Urea works by lowering the freezing point of water to -11.5ºC. The runway temperature, the presence of winds, the amount of precipitation on the airside pavement, the chance or occurrence of rainfall and the formation of precipitation all influence application rates. Generally, in anti-icing, a spreading rate of 15 g/m^2 is sufficient (Transport Canada 1972). De-icing quantities are determined by the thickness of the ice and are applied after as much snow and ice has been removed from the runway surface as is physically possible.

The use of urea as a runway ice control chemical, although effective, poses many concerns regarding environmental contamination. Results of Canadian and European studies indicated that 64 to 100% of the urea applied at airports may be discharged directly into the environment (Transport Canada 1990). Due to the frozen nature of most soil systems during periods of urea use, the most likely receptor of this runoff is surface water. There are two primary ways in which urea can disturb aquatic life and diminish water quality: toxicity and eutrophication. Urea has a 24-h LC_{100} value (i.e., the concentration of urea that kills 100% of the individual organisms to which it is administered) of 30 000 mg/L for creek chub in freshwater (Weiss 1986). Urea is also toxic to other aquatic life and micro-organisms at concentrations above 10 000 mg/L and to domestic animals at doses above 500 mg/kg (EPS 1985).

Once in the environment, urea [$CO(NH_2)_2$; 46.7 % N by weight] degrades to ammonia, nitrite, and nitrate. The degree of contamination to any aquatic system is determined mainly by the amount of urea used at the airport, the temperature of receiving waters and soils, and the pH of receiving waters and soils. High temperatures and high pH environments are more conducive to quick degradation of urea and its N components than are cold temperatures and low pH environments (Transport Canada 1990). A 1990 study by Transport Canada to determine whether application of urea on Canadian airport runways contributed to lake eutrophication, oxygen demand, aquatic toxicity, drinking water contamination or soil contamination found unacceptable levels of the breakdown products of urea (nitrate, ammonia, nitrate) at all of the studied airports. Since then, Canadian airports have

Table 7.1. Urea usage at selected Canadian airports (tonnes). Data source: 1974-1989 data from Transport Canada 1990; 1997-1999 unpublished airport data.

Airport	Average annual use	
	1974-1989 (t)	1997-1999 (t)
Calgary, AB	NR	16
Charlottetown, PEI	60	30
Edmonton, AB	45	No longer used
Fredericton, NB	NR	55
Gander, NF	NR	475
Halifax, NS	250	250
Montreal, QC (Dorval)	450	600
Montreal, QC (Mirabel)	900	600
Ottawa, ON	100	No longer used
Quebec City, QC	NR	200
Regina, SK	NR	No longer used
Saskatoon, SK	40	5
St. John, NB	NR	60
St. John's, NF	200	225
Thunder Bay, ON	NR	No longer used
Toronto, ON (Pearson)	300	No longer used
Victoria, BC	NR	20
Whitehorse, YK	NR	2
Windsor, ON	NR	2
Winnipeg, MB	75	No longer used
Yellowknife, NWT	NR	15

NR= Not reported

implemented a program of sampling and analyzing stormwater from open channels to ensure that airport effluent does not negatively affect the environment (Transport Canada 1994; DND et al. 1998).

The toxicity and eutrophication potential of urea has spurred a search for ways to mitigate possible deleterious effects. A number of methods for lowering urea's impact on the environment have been proposed and considered including blowing of snow over a larger area to increase the uptake of N by plants; selective harvesting of airside grass to maximize the amount of N removed from the area; construction of retention walls to slow down the escape of runoff and increase infiltration into the soil; ponding of urea-laden storm and melt water for summer watering of grass; use of a vacuum sweeper to collect stormwater for treatment; and treatment of urea stormwater by various methods. However, Transport Canada (1990) deemed all of these methods to be insufficient or impractical for controlling the environmental impacts of urea use at Canadian airports. Low toxicity alternatives have also been pursued. Three such products are calcium magnesium acetate, potassium acetate, and sodium formate. From the results of initial environmental studies (Transport Canada 1994), potassium acetate and sodium formate seem to be the most promising for use as runway anti/de -icers in Canada. Although urea remains operationally superior to all other de-icers, its degradation to ammonia creates serious environmental impacts. As a result, it has been banned at airports in several countries (Transport Canada 1994).

Urea use for each airport varies from year to year depending on the climatic conditions for each de-icing season. Urea is purchased by weight at the beginning of and during the de-icing season of October to May. Urea application records for individual de-icing events are not kept. Average urea use for a number of Canadian airports (1974-1979 and 1997-1999) are given in Table 7.1. In 1996, Lester B. Pearson International Airport was the only commercial airport that used exclusively non-urea runway de-icers (e.g., potassium acetate and sodium formate) (DND et al. 1998). However, a survey in August 1999 showed that airports in Edmonton, Ottawa, Regina, Thunder Bay, and Winnipeg have joined Pearson and switched to using sodium formate as a runway de-icer and potassium acetate as a runway anti-icer. Meanwhile, the Calgary Airport, the John G. Diefenbaker Airport in Saskatoon and the Whitehorse Airport now use mostly sodium formate as a runway de-icer mixed in with a little urea. Many of the airports still using urea indicated that they have performed assessments on the use of sodium formate and potassium acetate; however, cost of the replacements has deterred them from completely switching from urea. As for Canadian Forces Bases, only one Canadian base (Moose Jaw, SK) still uses urea as a runway de-icer; the others have switched to sodium formate (Lewis Cocks, DND, personal communication).

7.3. White haze over the southern region of the Lower Fraser Valley (adapted from Environment Canada 1999d)

For years, residents in rural regions of British Columbia's Lower Fraser River Valley were puzzled by the appearance of a thick band of white haze in the sky on calm, sunny days. Recent studies identified it as a rural version of urban smog associated with intensive agricultural production and, specifically, emissions from poultry and other livestock manures.

The haze, which occurs a couple of hundred metres above the ground, can persist for days trapped by a cap of warm air that seals off the valley like a lid and obscures the view of the surrounding mountains (Figure 7.1). Although its health effects are not yet fully known, the direct relationship between fine particles, respiratory disease and mortality has fuelled growing concern over this unusual phenomenon.

Manure has been a major concern in the Fraser Valley for some time because of its connection with rising N levels in local water supplies. Nitrogen budgets constructed for the Lower Fraser Valley revealed that a large quantity of N was entering the air as ammonia. Ground-level air sampling showed high concentrations of ammonium sulfate and ammonium nitrate in the air --evidence that the ammonia was likely combining with N and sulphur oxides from industrial pollutants and vehicle emissions. When the moisture in these compounds evaporates, the fine particulates that remain are visible as a milky haze. Some of these particles fall to the earth or are washed out in rain, so they can re-enter the cycle again and again.

Results from computer modelling suggest air parcels move from Vancouver through the Fraser Valley. As these air parcels pass over the Abbotsford area, an area of intense poultry farming, they pick up high levels of ammonia. White haze has not yet been reported elsewhere in Canada, although it is also a common phenomenon in farming regions of Colorado. Current efforts to transport manure out of the area to nitrogen poor regions of the province will help reduce agricultural ammonia emissions in this area. As well, local research efforts have investigated a change in poultry diets to reduce the N content of manure, amending turkey litter with alum to reduce ammonia emissions, or using it as a nutrient additive in cattle feed. A field study planned for the late summer of 2001 will attempt to

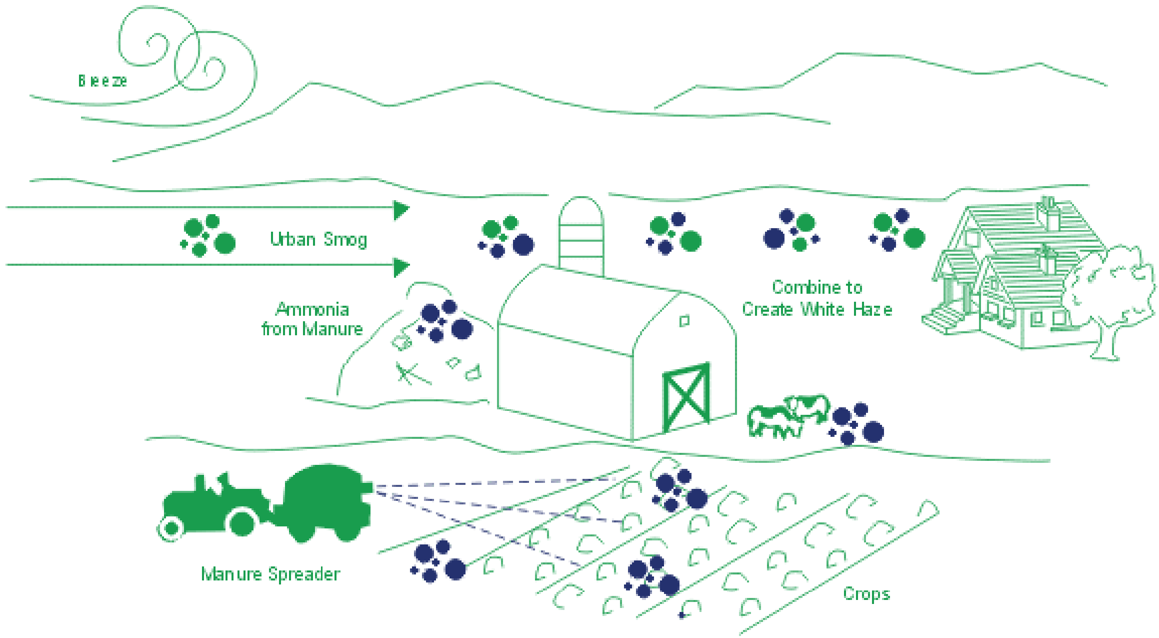

Figure 7.1. Ammonia from manure volatilizes into the atmosphere and reacts with particles of urban smog to create white haze.

measure the white haze in the Lower Fraser Valley. These, as well as existing data on white haze, will be used to develop airshed management plans and encourage stricter controls on N and sulphur oxide emissions in this rapidly growing region of the country.

7.4. The effects of nutrient enrichment on biodiversity

Biodiversity refers to "the variability among living organisms from all sources, including, among others, terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part" (Glowka et al. 1994). Biodiversity can be examined at three levels of biological organization: ecosystem, species, and gene. Ecosystem diversity refers to the number, variety, and extent of ecosystems either globally or within a given geographic area. It is important because genetic and species diversity tend to be greater where there is a variety of ecosystems to which species can adapt and where there has been enough time for complex ecosystems to develop. Species diversity refers to the variety of species on Earth or within a given area or ecosystem. It is essential to the production, consumption, decomposition, recycling, and other processes that take place in ecosystems. Finally, genetic diversity refers to the extent of heritable variation among members and populations of a given species. It allows species to adapt to changes in their environment (Government of Canada 1996). Earth's biodiversity is increasingly threatened by human activities that cause loss of natural ecosystems and extinction of species (e.g., Ehrlich and Ehrlich 1981; Freedman et al. 1996). Nutrient addition to native ecosystems is one of many threats to biodiversity.

Human alteration of the N cycle has accelerated losses of biological diversity especially among plants adapted to low-nitrogen soils (Vitousek et al. 1997). Limited supplies of biologically available N are typical of most natural terrestrial and marine ecosystems, and many native plant species are adapted to function best under this constraint. New supplies of N added to these ecosystems can cause a dramatic shift in the dominant species and a reduction in overall species diversity as the few plant species adapted to take advantage of high N out compete their neighbours. For example, experimental application of N fertilizers to grasslands and heathlands in Europe and North America has dramatically reduced the biodiversity of plant species, in turn potentially affecting other ecological processes (Vitousek et al. 1997). Similar phenomena are seen in freshwater ecosystems where P (rather than N) addition is usually responsible for eutrophication and loss of diversity.

The stresses from nutrient enrichment tend to drive aquatic ecosystems to less resilient, less complex, simpler states. In short, the composition of aquatic ecosystems and natural ecological processes are compromised:

- Moderate enrichment of **lakes** tends to result in increases in phytoplankton and rooted aquatic plants, coupled with changes in aquatic plant species composition. There are also increases in the abundance and changes in the composition of aquatic animals, such as invertebrates and fish. With gross enrichment, lake bottoms become smothered, resulting in reduced biodiversity of benthic organisms, as well as losses of fish species as a result of reduced survival of fish eggs due to oxygen depletion.
- In **rivers and streams**, moderate enrichment leads to increases in periphyton biomass and rooted aquatic plants, along with increases in productivity for higher trophic levels, such as insects and fish. With gross enrichment, there is reduced productivity by periphyton; reduced productivity by benthic invertebrates and fish; and a loss of some species.
- In **wetlands**, the number of species per unit area and biomass production increase under moderate nutrient loading. At higher nutrient loading, plant growth becomes limited due to competition for light (by shoots and leaves) and for space (by roots). Plant diversity also gives way to dominance by single species due to invasion by aggressive nitrogen-loving, highly tolerant, non-native vegetation. For example, in freshwater bogs, *Sphagnum* declines in abundance and is replaced with more nitrophilous mosses, whereas in fens, there are increases in tall graminoids (grasses and sedges) and decreases in subordinate plant species that are less able to compete for light. Marshes display increases in phytoplankton and emergent vegetation. In salt marshes, waters become choked with phytoplankton, blocking light penetration to the deeper vegetation. Thus, the direction of wetland succession is altered by changes in vegetative biomass and species composition. There may be shifts in invertebrate species composition, which can lead to changes in the quality and quantity of waterfowl food. Amphibians (frogs, toads, and salamanders) may be subject to nitrate and nitrite toxicity. Large amounts of nutrients released from salt marshes to adjacent estuaries may stimulate coastal water food webs.
- **Coastal waters** show evidence of increases in fast-growing phytoplankton and macro-algae, and reduced light to seagrasses, diminishing their ability to photosynthesize. Sediment anoxia impairs the ability of seagrasses to acquire N and accelerates seagrass mortality. Turbidity increases as sediments are no longer stabilized, further favouring motile phytoplankton that can move to the surface to maximize exposure to light; anoxia also enhances release of nutrients from the sediments (internal nutrient loading), favouring phytoplankton. Habitat is lost for many fish and benthic organisms. Marine algae have been found responsible for massive mortalities of fish, bird, and marine mammals.

Coupled with the stresses of increased nutrients is the fact that industrial emissions of nitrogen oxides further stress the biodiversity of species and ecosystems as they give rise to acid precipitation. Results include soil and water acidification, reductions in availability of nutrients for plants and animals for both terrestrial and aquatic ecosystems, and death of aquatic organisms (Government of Canada 1991). In conclusion, maintaining diversity helps ensure that the Earth will continue to perform the natural ecological processes upon which all life depends.

7.5. Fire retardants: Forest saviours or destroyers?

An important challenge currently facing forest managers in Canada is the development of ecosystem management plans that preserve biodiversity, allow multiple use of forested lands, and promote sustainable use. Forest fires play an important role in maintaining ecosystem health as they reset the successional stage of a forest. However, fires also endanger lives and cause economic loss to the forestry industry and property owners. Canada uses a variety of chemicals to aid in the suppression of wildfires, including long- and short-term fire retardants, foams, and wetting agents. These fire-fighting practices may have significant ecological consequences. For example, chemicals used to suppress wildfires may be applied to environmentally sensitive areas containing endangered, threatened, or economically important plant and animal species (Hamilton et al. 1994). Although it is not feasible to adopt non-suppression policies for all wildfires, concern exists about local impacts of retardants due to their potential toxicity or nutrient enrichment effects (Duchesne 1994).

Long-term fire retardants are formulations that continue to reduce or inhibit combustion after the water in the chemical-water mix has evaporated (Labat-Anderson Inc. 1996). They are usually applied aerially to a 25-30-m strip around the perimeter of a fire as the fire spreads, changes direction or approaches a town (Albert Simard, NRCan, personal communication; NARTC 1993). The retardant mixture consists of about 85% water, 10% fertilizer (typically ammonium phosphate or ammonium nitrate), and 5% minor ingredients (colour, thickener, corrosion inhibitors, stabilizers, and bactericides) (George 1995). Thus, long-term fire retardants are basically fertilizers. The fertilizers are present to encourage growth of vegetation both in the short-term, to make it more resistant to the fire, and over the long term following the fire (Johnson and Sanders 1977). The application of long-term fire retardants adjacent to and onto surface waters has resulted in four major environmental concerns: ammonia toxicity to sport fish; toxicity caused by the breakdown products from individual ingredients in the retardant; decline in species number and diversity of stream organisms; and eutrophication (Labat-Anderson Inc. 1996).

The greatest environmental threat posed by long-term fire retardants is ecosystem enrichment. Although it is preferable that aerial drops of fire retardant into or along any body of water be avoided, the fire location or the nature of the terrain may result in some retardant being dropped into lakes or streams. The possibility of eutrophication is greatest when retardant is dropped directly onto a stream or lake, particularly if the N content of the receiving water is very low. The eutrophication risk posed by long-term fire retardants to aquatic systems depends therefore upon the volume of retardant dropped onto a water body, the volume of the water body, and the flow or water exchange in the stream or lake. The risk of eutrophication caused by runoff from vegetated strips treated with fire retardant is considered to be small as most of the P is retained by the soil in a non-leachable form (Norris et al. 1978). Moreover, by the next year, most of the nutrients have been consumed by the forest vegetation (Albert Simard, NRCan, personal communication).

Long-term fire retardants usually pose a small toxicity risk in the environment. For aquatic ecosystems, the toxicity risk depends upon the ability of the receiving water to dilute the fire retardant. In general, long-term retardants are practically nontoxic to fish, with 96-h LC_{50} values ranging from 90 mg/L for Coho salmon yolk-sac fry to greater than 10 000 mg/L for rainbow trout eggs (Johnson and Sanders 1977; Poulton et al. 1993). Long-term retardants are slightly toxic to aquatic invertebrate species, with 96-h LC_{50} values for scud ranging from 45 to 62 mg/L (Johnson and Sanders 1977;

Monsanto 1991). Long-term fire retardants do not appear to pose an acute hazard to adult birds, mammals, earthworms, or humans (Smith 1987; Chemonics 1991; Monsanto 1991; Stillmeadow 1991; Poulton et al. 1993; Vyas et al. 1994). However, the fertilizer contained in the retardant may, under very specific conditions, cause nitrate poisoning to animals consuming hay or other forage crops contaminated by the retardant (George 1995; Labat-Anderson Inc. 1996). Little is known about the potential effects of these formulations on vertebrate behaviour and population dynamics. There is also little information on the potential toxicity of long-term fire retardants to terrestrial plants. It is possible that aerosols of ammonium sulfate may cause injury, including yellowing, tissue death and wilting, of plant leaves (Bradstock et al. 1987).

Currently, it appears that long-term fire retardants used to control Canadian forest fires do not pose a threat to the flora and fauna when used appropriately. However, further research is needed to address the impacts of these fertilizing chemicals when they are applied to areas that are naturally nutrient poor or nutrient rich, because the addition of the retardants could cause a dramatic shift in trophic status, even if the effect is only temporary.

7.6. Agriculture Intensification

The last century has seen great development in many agricultural technologies such as high yielding crop varieties, chemical fertilizers, pesticides, irrigation and mechanization (Matson et al. 1997). These developments have resulted in agricultural operations becoming increasingly specialized so that most farms now fall into one of two classes: (1) high density livestock operations, which often lack an adequate land base for environmentally-safe manure application, and (2) intensive cash-crop farms, which require high applications of chemical fertilizer. Where mixed farms were able to efficiently recycle animal manure by applying it to agricultural fields, the geographic separation between intensive livestock operations and cash-crop farms has resulted in poor use of manure as a fertilizer in some locales. This inequitable distribution of resources has raised concerns about the effect of P and N from manure and chemical fertilizers on air and water quality (Matson et al. 1997, Gleig and MacDonald 1998).

Manure is a low-density nutrient source and thus is not economical to transport long distances (Gleig and MacDonald 1998). Between 1976 to 1996, livestock operations in Canada became fewer and larger. For example, the number of hogs produced increased by 91% whereas the number of farms with pigs decreased by 69% over the same twenty-year period (Figure 7.2; Statistics Canada 1997c). Similarly, chicken numbers increased by 17% whereas chicken farms decreased by 72% while the number of cattle increased by 8% whereas beef farms decreased by 37% (Figure 7.2; Statistics Canada 1997c). For intensive livestock operations, manure poses a serious management problem as it must be stored for at least part of the year and it should be field-applied at rates that match the nutritive needs of the crop. In Canada, approximately 1 000 million tonnes of manure (wet weight) are produced annually (Larney 1992; Statistics Canada 1997c). Without proper storage, handling and field application of animal wastes, odour complaints, eutrophication of surface water, and ground water contamination become major concerns (Environment Canada 1998b; McCulloch et al. 1998). Mismanagement of manure may cause fish kills (see Chapter 5.2), contribute to the proliferation of toxic blue-green algal blooms, nitrate contamination of ground water, and eutrophication of surface waters.

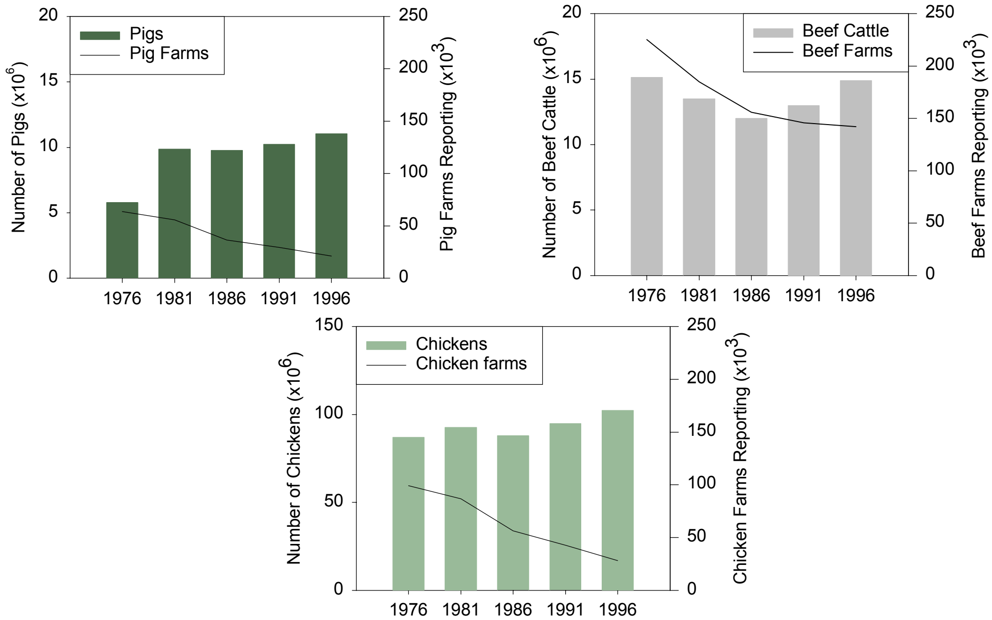

Figure 7.2. The changes in the numbers of pigs, beef cattle, and chickens and their associated farm numbers in Canada from 1976 to 1996. (Data from Statistics Canada 1997c).

In the high intensity livestock areas of Canada, centred in Ontario, Québec and Alberta, there are more nutrients available from manure than are required by crops (Bailey and Buckley 1998; Gleig and MacDonald 1998). Traditionally, manure application rates are calculated based on the N needs of the crop. This method of calculation, however, does not account for the manure's P content, which will exceed crop requirements if manure is applied on the basis of its N content. At current application rates, P levels in the soil are increasing and so is the risk of P migration and thus contamination of surface and ground water especially in the humid regions of Canada (Bailey and Buckley 1998; Gleig and MacDonald 1998). In Québec, British Columbia and Ontario, high soil P levels in areas around intense livestock production are becoming an environmental problem (Bailey and Buckley 1998).

If high intensity livestock areas have a surplus of available nutrients from manure, the opposite is true in the intensive cash cropping areas of Canada. Here, chemical fertilizers are required to meet the nutritional demands of the crop. The inequitable distribution of livestock and crops therefore limits the use of manure, not only as a fertilizer but also as a rich source of organic matter. Organic matter in the soil increases the nutrient and water holding capacity of the soil, influences soil temperature and nutrient mineralization, and provides a home for a diverse animal, bacterial and fungal community (Matson et al. 1997). The loss of the organic soil layer created by the soil biota has resulted in an increased reliance on irrigation, mechanical tillage and chemical fertilizers, all of which represent economic inputs for the producer.

Another feature of the intensification of farming is an overall reduction in plant diversity. Although managing a monoculture appears easier from a producer's perspective, this reduction in plant species creates an ecosystem favouring insect pests and plant pathogens. Nitrogen fertilization has also been found to increase the number of crop pathogens and populations of sap-feeding insects such as aphids, leafhoppers and planthoppers (Matson et al. 1997). Greater crop losses due to pests lead to an increased requirement for the use of pesticides. Recent studies have also shown that fertilizer drift promotes the growth of unwanted introduced weeds in the herbaceous layer of non-target habitats adjacent to cultivated fields (Boutin and Jobin 1998; Kleiljn and Verbeek 2000), therefore only exacerbating the problem of weed invasion in cultivated fields.

Over the last 50 years, improved agricultural technology has vastly increased annual crop yields in Canada. These improvements, however, have increased the impact of agriculture on the environment, especially with respect to water quality. The agricultural community has responded with the development of best management plans that attempt to balance the continued food production needs for Canada while attempting to minimize negative effects to the environment. In Canada, manure storage management issues have been addressed through regulations, guidelines, and codes of practice under existing legislation (e.g., British Columbia, Saskatchewan, Alberta, and Manitoba) and through assistance to producers to improve manure storage and management (e.g., Québec) (Government of Canada 1996). Similarly, guidelines to help producers calculate manure and fertilizer application rates based on crop requirements and existing soil conditions determined through soil testing are produced by each province in an effort to avoid problems associated with over or under fertilizing crops. New technologies have also alleviated some manure management problems. For example, techniques such as liquid manure injection systems that incorporate manure into the soil have reduced nutrient losses. Composting of animal manure has resulted in a more stable product not as susceptible to nutrient loss and environmental contamination. Advances in biotechnology are also increasing nutrient retention by animals and thus reducing wastage.

7.7. Municipal Biosolids Disposal

Sewage sludge is a nutrient rich, organic by-product of municipal wastewater treatment. Its creation during wastewater treatment is unavoidable and as effluent quality objectives continue to increase, so does the amount of sludge produced. Biosolids are the portion of the sewage sludge that has been stabilized through digestion to meet public health regulations for application to land (OMEE & OMAFRA 1996, WEF 1998).

Biosolids are highly valued as a soil amendment and nutrient source for agriculture (Campbell and Webber 1997; Evans 1998; Mackenzie 1998). However, because biosolids are derived from household and industrial waste, concern exists that application of biosolids to arable land will result in accumulation of heavy metals or organic contaminants in the soil and, ultimately, in crops or livestock. Biosolids vary widely in their heavy metal and organic contaminant content, depending on the mix of industries contributing waste to a MWTP. At present, the concentration of heavy metals in biosolids is the primary factor used to determine land application rates (CBCL Ltd. 1996; OMEE & OMAFRA 1996; AEP 1997; Campbell and Webber 1997; BCELP 1998a; SERM 1998). Organic contaminants found in biosolids include chemicals such as polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), phthalate esters, dioxins, and furans. Identifying and treating the mixture of organic chemicals in sludge is a difficult, time consuming and expensive task, and thus organic contaminant content has not been used to determine land application rates. Organic contaminants, however, were not found to accumulate in Ontario agricultural soils treated with biosolids at rates greatly exceeding recommended values (Webber 2000). The introduction and enforcement of industrial surcharge programs aimed at controlling pollution at the source and the recognition of the economic benefits of waste stream recovery and reuse have resulted in a considerable decrease in heavy metal concentrations in biosolids during the 1980s (Campbell and Webber 1997). This trend suggests that as technology continues to better detect and treat contaminants in sewage sludge, contaminant concentrations will continue to decline.

Currently, the volume of biosolids produced in Canada is increasing as more Canadians are connected to sewers and sewage treatment. In addition, higher levels of wastewater treatment mean greater biosolids production. Limited storage space for biosolids and increasing tipping fees at landfills have necessitated new management strategies for biosolids. Until recently, incineration was the preferred treatment technology as it minimized the quantity of material to be disposed. However, public concern about air emissions and ash disposal has caused this management solution to fall out of favour even though the technology required to clean air emissions is available, albeit expensive (Campbell and Webber 1997). Land application is currently the preferred management option for biosolids as it is cost effective, especially for smaller operations, and there is little environmental risk if application guidelines are carefully followed. Composting and fertilizer production also take advantage of the nutrients and organic compounds in the biosolids while reducing the volume of material produced. Composting operations, however, often fail due to odour complaints and a lack of market for the product. In fertilizer production, the biosolids are dried, or pellitized, and combined with other materials to be marketed as an organic fertilizer (Campbell and Webber 1997). Other emerging management technologies for biosolids include thermal conversion into liquid fuel (oil) for use as a recoverable energy source, utilization of biosolids ash in building materials, and technologies such as aqueous phase oxidation, alkaline stabilization and chemical or biological stabilization to immobilize or eliminate toxic compounds (Campbell and Webber 1997).

7.8. Aquaculture Expansion

Over the last 20 years aquaculture in Canada has developed into an industry representing about 1 750 producers and a wholesale value of $338 million, accounting for approximately 1.5% of total global aquaculture production in 1997. Although Canada is a small player in the global aquaculture industry, Canada's small to medium operations are important sources of employment to the largely rural areas in which they are situated. Since 1991, price declines, a moratorium on new sites in British Columbia, and an outbreak of an infectious disease in New Brunswick salmon have reduced the projected rapid expansion in the Canadian aquaculture sector. Aquacultural production is predicted, however, to double in Canada between 1998 and 2005 (DFO 1999). Such growth must occur within a consistent framework of sound environmental management practices designed to maintain a healthy and productive aquatic environment.

Aquaculture operations interact with the environment in a multifaceted manner. Although a number of concerns have been voiced about potentially deleterious impacts arising from aquaculture operations, from a water quality standpoint, the greatest impact results from the addition of food pellets to the cages. Food formulations that maximize N and P conversion into fish growth and thus minimize food waste to the environment are serving to decrease this impact. Nevertheless, in situations, such as Lac Heney, Québec, where water circulation is insufficient to dilute nutrient inputs adequately, nutrient loading from aquaculture can permanently alter the food web (see Lac Heney case study, Chapter 4.2). As well, oxygen depletion due to the microbial breakdown of accumulated food waste under aquaculture cages is often responsible for altering the benthic invertebrate community. In the case of shellfish aquaculture in Canada, the concern with respect to nutrients is not losses to the surrounding water, as shellfish operations in Canada are not supplied with additional food. Rather, increasing anthropogenic nutrient loads in marine environments appear to be increasing the magnitude and occurrence of hazardous algal blooms. A hazardous algal bloom in the vicinity of a shellfish aquaculture operation necessitates its closure as filter-feeding shellfish accumulate the toxins in their flesh making them unfit for human consumption.

In October 1999, British Columbia lifted its four-year moratorium on aquaculture expansion, an action expected to be followed by a large growth in Canadian aquaculture (BCEAO 1997). Still fraught with opposition from environmental groups, the expansion of aquaculture is being touted as a viable alternative to the wild salmon fishery, creating jobs for unemployed fishers. However, aquaculture operations, if not run properly, have the potential to affect biological and physical environments adversely. Potential effects will vary widely and depend on the physical setting and scale of the operation, site maintenance, feeding and the extent of competing uses for the environment in the area. The challenge, then, is to ensure that proper management measures are in place as Canada's aquaculture operations expand.

7.9. Municipal Wastewater Treatment Plants Siting

Effluents from MWTPs are often discharged to rivers and lakes in concentrations that can be damaging. Effluents are a concern because several of their constituents such as P and ammonia are many times more concentrated than is desired in the ambient environment. For example, in Lake Ontario the ambient desired P concentration is 0.010 mg/L but MWTP discharge concentrations are typically 50 to 100 times higher (0.5 to 1.0 mg/L). Thus, the receiving water is used to dilute the effluent, resulting in a zone of dilution near the discharge location. Dilution of effluents in receiving waters is an important factor in minimizing the costs of wastewater treatment. However, even if the total quantity of a particular pollutant that is discharged is acceptable, the size of the dilution zone and the concentration of the pollutant and environmental degradation in the dilution zone may still be issues. Size and local effects of the dilution zone depend on volume and concentration of the effluent, the efficacy of the diffuser (if one is used), the dispersion characteristics (water movement and turbulence) of the receiving water, and location of the discharge.

For rivers, the choice of a discharge site depends on environmental as well as economic factors. The location, upstream versus downstream as well as the distance across the river, may be chosen to minimize local effects. Discharges close to riverbanks may hug the riverbank and proceed many kilometres downstream with only gradual mixing or dilution. Discharges in the middle of rivers can mix across the flow in two directions and thus can be diluted somewhat closer to the discharge point. Discharge diffusers can cause rapid mixing of the effluent in ambient water with associated concentration decrease.

In lakes, the length of the discharge pipe is also an economic and environmental consideration. Lengthier pipes cost more but offer less environmental damage (assuming the total amount of constituents such as P is appropriate). When mixing zones impinge on shallow water or shorelines, the high nutrient concentrations in treated sewage can cause algal blooms and accumulation of noxious attached algae such as *Cladophora*. The longer the pipe the deeper the water in which an effluent can be discharged. The deeper the water, the more lake water moves past the discharge site to dilute the effluent. Buoyant plumes can be entrained into more water when the site is deeper. Thus, the extent of high concentrations in mixing zones in surface water is minimized with longer and deeper pipes. In addition, discharges far offshore minimize impacts on water intakes. Again the further offshore the discharge, the less likelihood there is of shoreline effects.

Volumes of discharges are gradually rising as population grows. Thus, increasing volumes of effluent are being discharged in the same locations as were established decades ago. Most discharges in large lakes are within 1 to 2 km of shore. As the effluent volume increases, so does the size of the mixing zone. The impact of the mixing zone can be managed somewhat with more and more advanced sewage treatment. Eventually, however, there will be a trade off between more and more costly treatment versus more and more costly pipelines that extend further offshore. In North America, even very well treated sewage is extremely active in stimulating algal growth. The natural dispersion and assimilation abilities of lakes depend on depth and distance offshore. Thus, as volumes of sewage grow, more consideration will have to be given to discharge location if deleterious nearshore effects are to be avoided.

7.10. Lake Winnipeg: Eutrophication of a Manitoban Great Lake

Lake Winnipeg is located in central Manitoba. It is the 10th largest (23,750 km^2) freshwater lake in the world and its drainage basin is the second largest in Canada. The basin spans four provinces (Alberta, Saskatchewan, Manitoba and Ontario) and three US states, and includes three major rivers (Red, Winnipeg and Saskatchewan). Lake Winnipeg is the central lake in a chain of large lakes extending across Canada from Great Bear Lake and Great Slave Lake in the Northwest Territories to Lake Athabasca situated on the Alberta/Saskatchewan border, to Lake Manitoba and then to the Great Lakes on the Ontario-USA border.

Lake Winnipeg has considerable economic, aesthetic and cultural value to Manitobans. The landed value of the pickerel and whitefish fishery exceeds $25 million annually and supports more than 700 fishers and their communities, many of whom are aboriginal. Lake Winnipeg also serves as a reservoir for Manitoba Hydro and contributes to the generation of export power, valued in excess of $350 million annually. Lake Winnipeg is also the source of drinking water for many aboriginal communities located along the Nelson River.

The tributaries draining into Lake Winnipeg are naturally variable in their nutrient and silt load, reflecting both the parent geology and human activity in their basins. Tributaries arising to the east of the lake drain mostly the boreal shield ecozone and are characterized by high water yields and low nutrient and silt content. In contrast, waters from the south and west drain sedimentary parent material and are characterized by low water yields and high nutrient and silt concentrations.

Changes in human activity in the basin over the past 30 years have increased stress on the Lake Winnipeg ecosystem. It appears there has been a gradual increase in N and P likely due to changes in agricultural practices, including expansion of the livestock and food processing sectors, and increases in human population. Urban populations have continued to grow with several major cities still having no P removal from wastewaters. The lake has also been affected by increased demand and value for hydro-electric generation, which has led to hydraulic management of Lake Winnipeg as a reservoir. This management action may be changing the lake's nutrient carrying capacity through increased light penetration and the retention and recycling of N and P (M. Stainton, DFO, Winnipeg).

Only a few studies have been conducted on Lake Winnipeg. Bajkov (1934) in the 1920s and Brunskill (1973; Brunskill et al. 1980) in 1969 undertook the only comprehensive studies dealing with spatial and temporal variability in the physical, chemical and biotic components of the lake. Five smaller studies have been undertaken since 1969 by staff from Fisheries and Oceans Canada and the Province of Manitoba (Manitoba Department of Mines, Resources and Environmental Management 1974; Todd et al. 1996; Stewart et al. 1998; M. Stainton, DFO/Winnipeg, unpublished data). Together, these data indicate that increased human activity in the basin has caused Lake Winnipeg to become more eutrophic. Although the effects of the variety of human activities operating in the Lake Winnipeg basin are not fully understood, the deterioration in water quality is likely caused by a combination of factors. These include increased P loading from the watershed, increased transparency of the lake water due to reduced sediment input from tributaries dammed for hydroelectric power generation, and increased retention and recycling of nutrients in the lake due to changes to hydraulic cycles stemming from lake impoundment. A much better understanding of changes in nutrient loading from the watershed to the lake and changes to internal nutrient processing is required to manage inputs and effect clean-up of this great lake.