Ecosystem Health

Science-Based Solutions

Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems

Report No. 1-8

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Scientific Supporting Document

Canadian Guidance Framework for the Management of Phosphorus in Freshwater Systems

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NOTE TO READERS

The *Ecosystem Health: Science-based Solutions* series is dedicated to the dissemination of scientific knowledge, information and tools for monitoring, assessing, and reporting on ecosystem health to support Canadian in making sound decisions. Documents published in this series include the scientific basis, methods, approaches and frameworks for environmental guidelines and their implementation; monitoring, assessing, and rehabilitating environmental quality in Canada; and, indicator development, environmental reporting and data management. Issues in this series are published *ad libitum*.

This particular issue provides a general overview of the current understanding of phosphorus in surface waters and presents details on the Canadian Guidance Framework for the management of phosphorus in freshwater ecosystems in Canada. For additional information regarding this document, please contact:

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ABSTRACT

This document presents the Canadian Guidance Framework for the management of phosphorus in freshwater systems. In Canada, the trophic status of water bodies that are impacted or unimpacted by anthropogenic releases of phosphorus range from oligotrophic to hyper-eutrophic. Typically unimpacted sites support relatively diverse and abundant communities of aquatic organisms that are self-sustaining and support various water uses. However, elevated loads of phosphorus can have many adverse effects on aquatic ecosystems. One of the most important consequences is the increased growth of algae and aquatic macrophytes. The senescence and decomposition of these organisms create oxygen deficit and can result in fish kills. In some freshwater systems, blooms of cyanobacteria contribute to a wide range of problems including summer fish kills, foul odours, tainted drinking water, and release of toxins that can kill livestock and may pose a serious health risk to humans.

The protocol to develop Canadian Water Quality Guidelines is intended to deal specifically with toxic substances. Although elemental phosphorus can be toxic, it is rare in nature, and therefore rarely of concern. Because aquatic communities are well adapted to the ambient conditions that occur within lakes and rivers in which they reside, it is neither feasible nor desirable to establish a single numerical water quality guideline value for phosphorus in Canada. Furthermore, management goals in different regions of Canada may vary. It is generally considered that the trophic status of water bodies should not be altered to ensure that the designated uses are not adversely affected.

Some of the effects of phosphorus are aesthetic, and so its management requires an element of subjectivity. Therefore it is more difficult to derive ‘safe’ levels of phosphorus than it is for toxic pollutants; what is considered nuisance plant growth to some may be considered desirable to others. This requires a need for guidelines that are scientifically defendable and which can be incorporated into management practices. At international, national, and local levels nutrient enrichment has been tackled through a wide range of approaches. Basing criteria on the best scientific data will minimize any conflict surrounding desirable phosphorus levels and aesthetic impairment and maximize the potential benefits to controlling nutrients.

Methods employed, both nationally and internationally, have been identified for the development of phosphorus criteria. Of the numerous methods, Australia and New Zealand have developed criteria or ‘trigger values’ that are risk based, scientifically defensible, consider the effects of other contributing factors, and are easily incorporated into management practices. The development of the Canadian guidelines for phosphorous follows a similar direction. A framework-based approach that includes elements of the Australian model has been developed. The proposed approach accommodates the non-toxic endpoints associated with phosphorus and can be incorporated into existing management strategies (Figure 1.1, Chapter 1). The framework offers a tiered approach in which (i) phosphorus concentrations should not exceed predefined ‘trigger ranges’; and (ii) phosphorus concentrations should not

increase more than 50% over the baseline (reference) levels. The trigger ranges are based on the range of phosphorus concentrations in water that define the reference trophic status for a site. If the upper limit of the range is exceeded, or is likely to be exceeded, further assessment is required. When assessment suggests the likelihood that phosphorus levels will result in an undesired change in the local system, a management decision must be made.

RÉSUMÉ

Le présent document expose en détail le Cadre d'orientation canadien pour le phosphore dans les réseaux d'eau douce. Au Canada, les plans d'eau, qu'ils soient touchés ou non par des rejets de phosphore, couvrent un large spectre quant à la situation trophique, depuis l'oligotrophie jusqu'à l'hyper-eutrophie. Ceux qui sont généralement épargnés par de tels rejets soutiennent des communautés relativement diverses et abondantes d'organismes aquatiques qui sont autosuffisantes et permettent différentes utilisations de l'eau. Toutefois, les charges élevées en phosphore peuvent avoir beaucoup d'incidences négatives sur les écosystèmes aquatiques. Un des principaux effets néfastes est la prolifération d'algues et de macrophytes, dont la sénescence et la décomposition créent un déficit en oxygène et peuvent causer la mort de poissons. Dans certains réseaux d'eau douce, des proliférations de cyanobactéries contribuent à l'existence de nombreux problèmes différents, dont la mort de poissons en été, de mauvaises odeurs, l'altération de la qualité de l'eau potable et la libération de toxines qui peuvent entraîner la mort d'animaux d'élevage et présenter un grand risque pour la santé humaine.

Le protocole d'élaboration des Recommandations canadiennes pour la qualité des eaux vise spécifiquement les substances toxiques. Le phosphore élémentaire, bien qu'il puisse être toxique, est rare dans la nature; il suscite donc rarement des inquiétudes. Étant donné que les communautés aquatiques sont bien adaptées aux conditions ambiantes des lacs et des cours d'eau où elles se trouvent, il n'est ni possible ni souhaitable d'établir un seuil numérique unique de concentration de phosphore au Canada pour le maintien de la qualité de l'eau. Sans compter que les objectifs de gestion de l'eau peuvent varier d'une région à l'autre. Il est généralement admis qu'il faut éviter de modifier la situation trophique des plans d'eau afin de prévenir les incidences négatives sur les utilisations désignées.

Comme certains des effets du phosphore sont de nature esthétique, la gestion de cette substance comporte un élément de subjectivité. Pour cette raison, il est plus difficile de déterminer quelles sont les teneurs « sûres » en phosphore par rapport à celles de polluants toxiques : ce que certains considèrent comme une croissance végétale nuisible peut être souhaitable pour d'autres. Il faut donc produire des recommandations fondées scientifiquement qui peuvent être intégrées dans des pratiques de gestion. On a adopté une foule d'approches différentes aux paliers international, national et local pour s'attaquer à l'apport excessif de matière nutritives dans les plans d'eau. Si les critères reposent sur les meilleures données scientifiques disponibles, on réduira au minimum le risque de conflits liés à la teneur en phosphore souhaitable et à la diminution du caractère esthétique, tout en maximisant les possibilités pour la maîtrise des éléments nutritifs.

On a inventorié les nombreuses méthodes employées, tant au Canada qu'à l'étranger, pour l'élaboration de critères relatifs à la teneur de l'eau en phosphore. Ainsi, l'Australie et la Nouvelle-Zélande ont établi des critères ou valeurs-seuils

fondés sur le risque justifiables scientifiquement qui prennent en compte les effets d'autres facteurs et sont faciles à intégrer à des pratiques de gestion. L'élaboration des recommandations canadiennes relatives au phosphore ont un fondement comparable. Une approche reposant sur un cadre qui intègre des éléments du modèle australien a été élaborée. Elle permet l'établissement de paramètres d'évaluation non liés à la toxicité pour le phosphore et peut être incorporée aux stratégies de gestion existantes (figure 1.1, chapitre 1). Ce cadre offre une approche par étapes à l'intérieur de laquelle les concentrations de phosphore ne devraient pas i) dépasser des « gammes-seuils » prédéterminées et ii) augmenter de plus de 50 % par rapport aux concentrations de base. Ces gammes-seuils sont basées sur la fourchette de concentrations de phosphore dans l'eau qui définit la situation trophique de référence à un site donné. S'il existe une probabilité que la limite supérieure de la fourchette soit dépassée ou si celle-ci est effectivement dépassée, une évaluation poussée s'impose. Lorsque cette évaluation semble indiquer que les concentrations de phosphore produiront vraisemblablement une modification non souhaitée du réseau d'eau douce local, il faut prendre une décision de gestion.

GLOSSARY

θ	thermal
Abiotic	non-living things
AFDM	ash free dry mass
Algae	prokaryotic and eukaryotic photosynthetic organisms, never with roots stems or leaves
Allochthonous	particles produced outside of the system of interest
$AlPO_4 \cdot H_2O$	aluminium phosphate hydrate (variscite)
Anthropogenic	resulting from, or influenced by mans activities
APA	alkaline phosphatase activity
Autotrophic	able to elaborate all its chemical constituents from simple, inorganic compounds
Benthic	living at the soil-water interface
Biomass	organic matter harvested as a source of energy/ total dry mass of an animal or plant population
Bioturbation	movement of sediment particles by organisms living in or within the sediment
BOD	biological oxygen demand
$Ca_5(PO_4)_3OH$	hydroxyapatite
$CaCO_3$	calcium carbonate
CCME	Canadian Council of Ministers of the Environment
CH_4	methane
Chemisorbed	irreversibly adsorbed in which the absorbed surface is held on the subsurface by chemical forces
Chl *a*	chlorophyll *a*
Chlorococcale plankton	unicellular plankton which may form colonies of uninucleate, or multinucleate, cells, which never divide vegetatively
Chrysophycean plankton	golden-brown algae, a class of eukaryotic algae
Cladocerans	small mostly freshwater crustaceans of the order Cladocera
CO_2	carbon dioxide
CO_3^{2-}	carbonate
CWQG	Canadian Water Quality Guidelines
Cyanobacteria	blue-green algae
Cyanophyta	blue-green algae
Desmids	unicellular green algae
Diagenesis	process affecting the top meters of sediment, including dewatering, bioturbation, bacterial decomposition of organic matter, dissolution of carbonates and redox reactions
Dimictic	waterbody having two mixing periods, typically spring and fall
Dinoflagellate plankton	mesokaryotic algae, motile cells have two inserted

	flagella
DIP	dissolved inorganic phosphorus
DO	dissolved oxygen
DOC	dissolved organic carbon
DOP	dissolved organic phosphorus
Ecoregion	an environmental area characterized by specific land uses, soil types, surface form, and potential natural vegetation
Ecozone	large terrestrial ecosystem unit that contains distinctive sets of nonliving and living resources that are ecologically related as a system
EEC	European Economic Community
EIS	Environmental Impact Statement
E^o	Energy
EMAP	Environmental Monitoring and Assessment Program of the USEPA
EPA	Environmental Protection Agency
Epilimnion	warm upper layer of water in a lake
EU	European Union
Euphotic zone	photic zone – zone where light penetration is sufficient for photosynthesis
Eutrophic	in which hypolimnion becomes depleted of oxygen in the summer by the decay of organic matter
Eutrophication	an excessive growth of organisms and oxygen depletion which is caused by the enrichment of water by nutrients such as nitrogen and phosphorus
Fe (II)	ferrous
Fe (III)	ferric
$Fe(OH)_3$	ferric hydroxide
$Fe(OH)_3P$	phosphorus adsorbed ferric hydroxide
$FePO_4$	ferric phosphate
$FePO_4 \cdot H_2O$	iron phosphate hydrate (strengite)
GRCA	Grand River Conservation Authority
H_2S	hydrogen sulphide
HCO_3^-	bicarbonate
HPO_4^{2-}	hydrogen phosphate
Humic acid	complex organic acids occurring in soils and formed by the decomposition of dead vegetable matter
Hypolimnion	the cold lower level of water
Lacustrine	related to a lake
LCM	Lakeshore Capacity Model
Lentic	associated with standing water
Littoral	the shallower water of lakes where light reaches the bottom and where rooted plants grow
LSEMS	Lake Simcoe Environmental Management Strategy
Lysis	decomposition or splitting of cells

Macroinvertebrates	benthic fauna without backbones whose adult form is typically larger than 500 μm
Macronutrients	element required in relatively large quantities
Macrophyte	large aquatic plants
Mesotrophic	moderately productive, relating to moderate fertility of a lake in terms of its algal biomass
Metastable	state which is apparently stable, often because of the slowness with which equilibrium is reached
Micronutrients	trace element required in relatively small quantities by living organisms
Microorganisms	organisms too small to be visible without a microscope
NH_4^+	ammonium
NO_3^-	nitrate
NZ	New Zealand
O_2	oxygen
OECD	Organization for Economic Co-operation and Development
Oligotrophic	body of water in which primary production is low
OMOE(E)	Ontario Ministry of the Environment (and Energy)
Oxic	containing oxygen
Periphyton	microfloral growth upon substrata of freshwaters
Phytoplankton	floating photosynthetic plants or animals
PIBI	Periphyton Index of Biotic Integrity
Planktivores	organisms which feed on plankton
Plankton	floating plants or animals
PO_4^{3-}	orthophosphate ion
POP	particulate organic phosphorus
Profundal zone	area beneath the limnetic zone and extends to the bottom (were light is insufficient for photosynthesis)
P_S	soluble phosphorus (SRP)
P_T	total phosphorus (TP)
RA	relative abundance
Redox	oxidation-reduction
Saprobien	method used to assess the condition of water resources, based on nutrient content
Seston	all particulate material that can be retained by nets or on reasonably fine (> 0.5 μm) filter
Silviculture	planting and care of forests
SiO_2	silicon dioxide
SO_4^{2-}	sulphate
SRP	soluble reactive phosphorus
Thermocline	in lakes, region of rapidly changing temperature found between the epilimnion and hypolimnion
Trophic	pertaining to nutrition
Trophogenic zone	water column depth in lake where photosynthetic

	production predominates
UK	United Kingdom
UN	United Nations
USEPA	United States Environmental Protection Agency
WQG	water quality guidelines
WQI	water quality index
WQO	water quality objectives
Zooplankton	floating and drifting animal life

1. INTRODUCTION

1.1 Background

Elevated levels of phosphorus, in conjunction with nitrogen, can affect aquatic ecosystems in many ways, much of which can be considered negative. One of the most important consequences is increased growth of algae and aquatic macrophytes and distinct shifts in species composition. The senescence and decomposition of these organisms, as well as nocturnal oxygen consumption by community respiration, create shortages in dissolved oxygen (DO) resulting in fish kills. In some freshwater systems, blooms of cyanobacteria in particular are a prominent symptom of eutrophication (Carpenter et al. 1998a). These blooms contribute to a wide range of water quality problems including summer fish kills, foul odours, and tainted drinking water. Furthermore, certain cyanobacteria produce and release toxins that can kill livestock and may pose a serious health threat to humans.

Owing to these environmental concerns, phosphorus is on the list of priority pollutants for the Canadian Council of Ministers of the Environment (CCME). Currently, no national environmental quality guidelines for phosphorus exist, although individual provinces may have guidelines or objectives. The Protocol for the Derivation of Guidelines for the Protection of Aquatic Life (CCME 1991) is intended to deal specifically with toxic substances and provide numerical limits or narrative statements based on the most current, scientifically defensible toxicological data available. Phosphorus does not fit this model because it is non-toxic to aquatic organisms at levels and forms present in the environment; however, secondary effects, such as eutrophication are a serious concern. Therefore, attempting to set phosphorus guidelines by the Canadian Water Quality Guidelines (CWQG) protocol is difficult due to the subjective nature of what constitutes impairment. Some of the effects of phosphorus are aesthetic and, therefore, its management requires consideration of societal values. As such, guidelines were not derived; rather a guidance framework was developed that is consistent with CCME guideline principles but that permits site-specific management of phosphorus.

1.2 Development of the Framework for the Management of Phosphorus in Freshwater Systems

As with traditional guidelines, a thorough review of the scientific literature on phosphorus was conducted as part of developing this guidance framework. A number of approaches used by other jurisdictions for setting phosphorus guidelines were examined and are described in Chapter 3. A framework-based approach that includes elements of the Australian model (NWQMS 1999) is proposed. The proposed approach, outlined in Chapter 4, accommodates the non-toxic endpoints associated with phosphorus and can be incorporated into existing management strategies (Figure 1.1). As a first step, management goals and objectives (e.g., maintain, enhance, or restore) for the water system are defined. Next, the condition or water quality and desired uses of the system are considered, and an appropriate reference, or baseline, condition is defined. The framework provides a tiered approach in which water bodies are marked for further assessment by comparing their trophic status to ‘trigger ranges’ that have been predefined (Table 1.1). The trigger ranges are selected according to the level of phosphorus observed in the reference or baseline condition of a lake or river. Should the upper limit of these ranges be exceeded, or are likely to be exceeded, further assessment is recommended to better characterize the type and magnitude of impact. When the trigger range is not exceeded, there is still the risk of observable changes in the community structure; therefore a fail-safe has been added to the framework. Changes greater than 50% of the baseline may result in observable changes in the macrophyte and periphyton communities; therefore if the increase from the baseline is greater than 50%, regardless of whether it lies within the trigger range, further assessment is recommended.

Chapter 5 provides a summary of some of the recommended assessment options currently in use in Canada, the US, and other parts of the world. The approaches have been divided into three groups: (1) the development of a water quality index (WQI); (2) multivariate approach, where an array of physical, chemical and biological variables can be analyzed statistically; and (3) the adaptation of existing predictive models. These approaches are not mutually exclusive, and are merely provided as guidance; the development of tools suited for specific ecosystems are encouraged. When an approach suggests the likelihood that phosphorus levels will result in an undesired change in a specific system, a management decision must be made (Chapter 6). Management decision is a critical step in the framework that links back to the narrative water quality goals of the program; to ensure management strategies are having the desired effect, it is important that some form of monitoring program be adopted.

Table 1.1 Total phosphorus trigger ranges for Canadian lakes and rivers.

Trophic Status	Canadian Trigger Ranges Total phosphorus ($\mu g \cdot L^{-1}$)
Ultra-oligotrophic	< 4
Oligotrophic	4-10
Mesotrophic	10-20
Meso-eutrophic	20-35
Eutrophic	35-100
Hyper-eutrophic	> 100

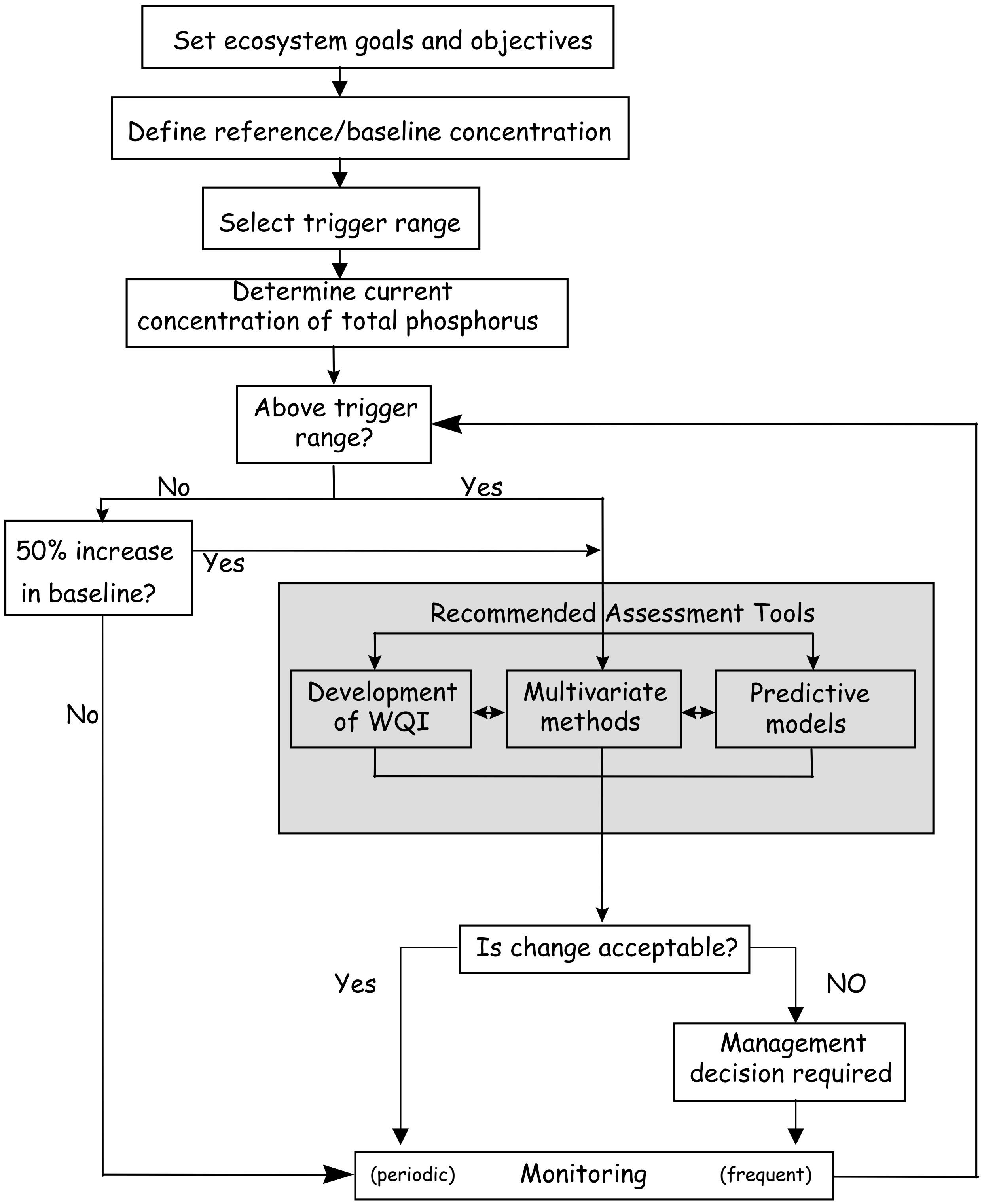

Figure 1.1 Canadian Guidance Framework for the management of phosphorus in freshwater systems.

1.3 Specific Concerns

1.3.1 Establishing a reference condition

Establishing the reference condition of a given water body is the most important step in the framework because it will determine which trigger range will be used for comparison. It will likely be the most difficult part of the framework. In some cases, historical data may be available, but in most cases there will be a need to estimate a baseline concentration. Many techniques have been used to estimate baseline phosphorus concentrations. Where site-specific historical data are not available or can not be obtained by other methods (e.g., modelling, paleoecology), the "best available condition" is recommended for use in the framework. It involves selecting relatively clean sites with similar chemical, physical, and biological characteristics and grouping these water bodies into one reference condition, to which the study lakes and rivers may be compared.

It is recognized that the determination of baseline conditions in framework may be resource intensive. For this reason we have provided several options for determining reference concentrations, ranging from use of existing (historical) data to derivation and application of predictive models to hindcast pre-development phosphorus concentrations. Many jurisdictions (e.g., British Columbia and Ontario) that are actively managing phosphorus, already have established reference conditions that could be used in the framework. In addition, reference conditions will be relatively simple to determine in areas with little or no development. In geographical areas where there is a high density of water bodies, a single reference condition may be defined for the entire area.

1.3.2 OECD trophic status values

We are recommending that the Organisation for Economic Co-operation Development (OECD) (Vollenweider and Kerekes 1982) trophic status values be used as trigger ranges because they are internationally accepted (Table 1.1). These ranges are applicable to Canadian waters because they are widely accepted in the scientific community as standards. However, we have subdivided the OECD meso-eutrophic category (10-35 $\mu g \cdot L^{-1}$) into mesotrophic (10-20 $\mu g \cdot L^{-1}$) and meso-eutrophic (20-35 $\mu g \cdot L^{-1}$). This subdivision was deemed necessary because considerable variability in community composition and biomass exist in Canadian waters over the OECD range of 10-35 $\mu g \cdot L^{-1}$.

These trigger ranges are not only applicable for lakes. Typically, rivers can sustain higher loads of phosphorus than lakes without observable changes in community composition and biomass, as phosphorus is often flushed from the system before it can be utilized. Ultimately, the receiving environment is the most sensitive component to changes in phosphorus loads and as such dictates phosphorus concentrations in river water. Therefore, we recommend that the modified OECD trophic classifications be applied to rivers as well as lakes. We acknowledge that the United States Environmental Protection Agency (USEPA) adopted trophic classifications for rivers

(Dodds et al. 1998) that are higher than the OECD values for lakes; these values may be appropriate for selected sites (e.g., large river systems, sites that have high baseline concentrations).

1.3.3 Increase of phosphorus over the baseline

The framework proposes that if the current concentration of phosphorus in a given lake is greater than the upper limit of its trigger range, or 50% higher than its baseline, further assessment is recommended. A number of approaches, including the multivariate approach, the water quality index, and paleolimnological and mass balance models are evaluated for setting the management goals for phosphorus or other parameters indicative of enrichment. The 50% increase from baseline was based on Pre-Cambrian lakes in the Canadian Shield (OMOE 1997) and lakes in the UK (Johnes et al. 1994).

The 50% increase limit is reasonable for Canadian conditions. In general, a 50% increase over the baseline will result in observable changes in community composition and biomass. However, Clarke and Hutchinson (1992) found that a 50% increase in lakes with an already high baseline concentration of phosphorus (>12 $\mu g \cdot L^{-1}$) may not be protected. Therefore, a 50% increase may be too large to protect lakes where the baseline concentration is greater than 12 $\mu g \cdot L^{-1}$. In the absence of empirical data to recommend an alternative for these lakes, this increase limit is deemed preferable to no limit.

Although studies have predominantly focused on lakes, the limit is appropriate for application to conditions in both lakes and rivers. In large lakes, the 50% increase should be applied to the most sensitive area(s) (e.g., river mouth, point sources, or littoral zones) rather than averaged over the whole lake. In this situation, baseline phosphorus concentrations should also be established for these areas. In order to be protective of the receiving environments within the watershed, we are recommending that this 50% increase also apply to river systems.

1.3.4 Validating the framework

A number of different case studies are discussed in this document, each of which were used to highlight a different aspect of the framework. First, data from Lake Simcoe is used to demonstrate how an existing lake management system fits into the framework (Chapter 7). Second, mathematical models, specifically the export coefficient model, a method that has been used across Canada, is used. The model's usefulness in modelling nutrients has been demonstrated with data from northern river basins study (Chambers and Dale 1997; Chambers et al. 2001); on an urban watershed in Waterloo County (Winter and Duthie 2000); and as part of the Lake Simcoe environmental management strategy (Isems 1995). Third, data from a diamond mine was applied to the framework *a priori* to determine whether phosphorus could be a potential problem (Chapter 8). In this example, phosphorus was not considered during the preliminary impact assessment; however, it was later discovered that elevated levels of phosphorus indirectly resulted in anoxic conditions.

2. PHOSPHORUS IN THE AQUATIC ENVIRONMENT

2.1 Introduction

Phosphate, nitrate, and silica are generally considered the most critical nutrients for autotrophic production in freshwaters. The principle nutrients are usually nitrogen (N) and phosphorus (P), and increases in these nutrients can often result in an increase in primary production. Water entering a lake will generally be relatively richer in nitrogen than phosphorus because nitrogen is naturally more abundant than phosphorus, and because phosphorus compounds tend to be more insoluble than nitrogen ones (Moss 1998). In nutrient poor freshwater, inorganic phosphate can often be, although not always (Grimm 1987), the principle factor limiting the growth.

2.2 Physical and Chemical Properties of Phosphorus

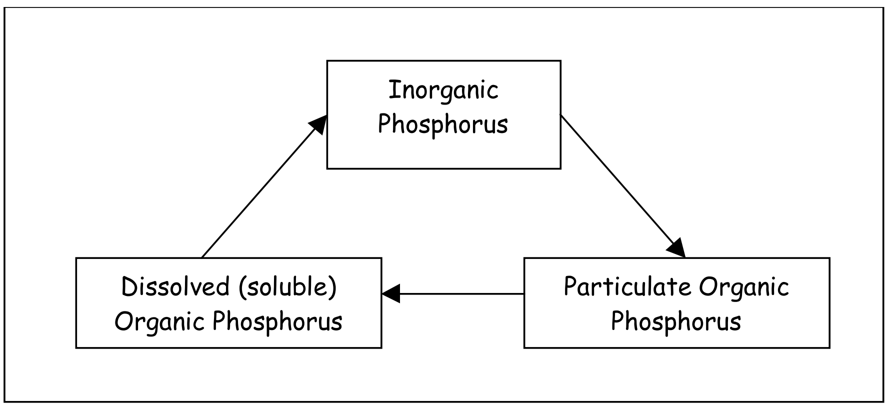

Figure 2.1 Schematic of the three forms of phosphorus occurring in aquatic systems and their cycling.

Phosphorus (P; CAS No. 7723-14-0, atomic mass 30.97376) is a highly reactive, multivalent, non-metal of the nitrogen group that is never found free in nature. Phosphate rock, which contains the mineral apatite, an impure tri-calcium phosphate, is an important source of phosphorus. In aquatic system phosphorus occurs in three forms (Figure 2.1). For their nutrition, aquatic plants require inorganic phosphate, typically in the form of orthophosphate ions (PO_4^{3-}). This is the most significant form of inorganic phosphorus, and is the only form of soluble inorganic phosphorus directly utilized by aquatic biota. This form of phosphate is transferred to consumers and decomposers as organic phosphate. Phosphate concentrations tend to increase with increases in total phosphorus (TP), but the proportion of phosphate declines with increasing TP (Hudson et al. 2000). A major proportion of phosphorus (as much as 95%) in fresh water occurs as organic phosphates, cellular constituents of organisms, and within or adsorbed to inorganic and dead particulate organic material. This is subsequently made available for recycling via mineralization and decomposition (Wetzel 2001). Hudson et al. (1999)

measured phosphorus regeneration directly and observed that cycling efficiency does not vary with phosphorus concentration and that the phosphorus supply for lake plankton comes primarily from plankton community. Phosphate is extremely reactive and interacts, especially under oxidizing conditions, with many cations to form relatively insoluble compounds that precipitate out of the water. The availability of phosphate is also reduced by adsorption to inorganic colloids and particulate compounds (such as clays, carbohydrates, and hydroxides).

Exchange of phosphorus between its various forms is often rapid, and involves numerous pathways. At least two major fractions make up the organic phosphorus of the open water seston:

i. A rapidly cycled fraction which is exchanged with soluble forms. In this fraction, phosphorus is transferred rapidly from the particulate phase to low-molecular-weight compounds;

ii. A fraction of dissolved organic and colloidal phosphorus material that is released and cycled more slowly.

2.3 Sources and Fate

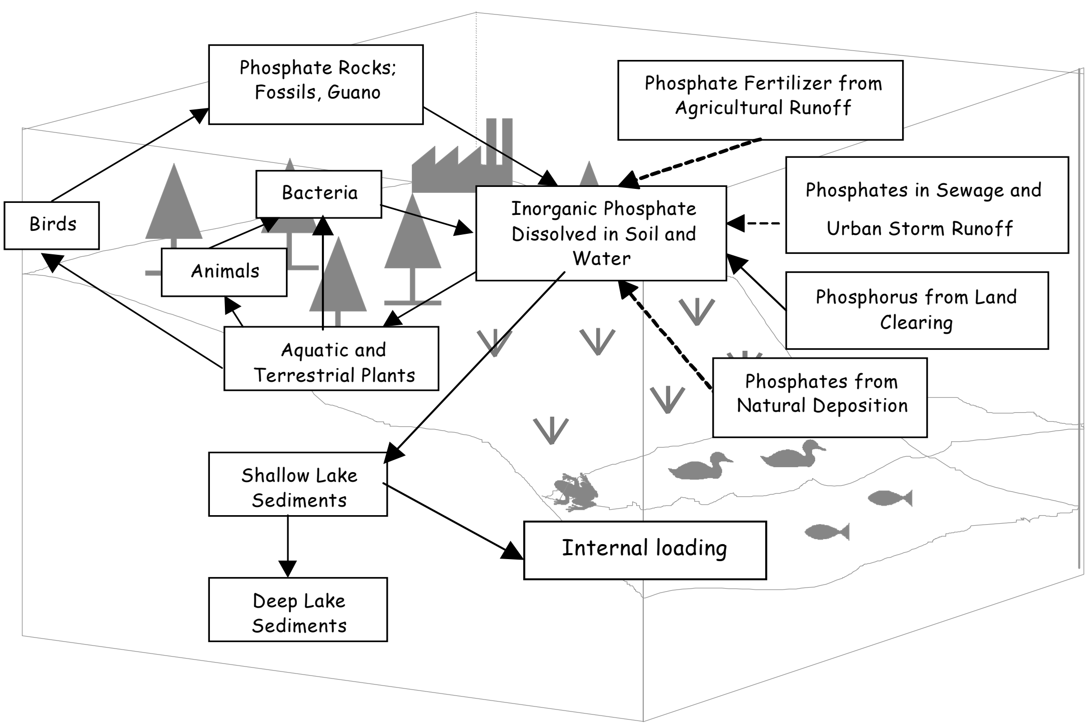

Figure 2.2 The phosphorus cycle.

Sedimentation of particles may result in constant losses of phosphorus from the trophogenic zone. As a result, new phosphorus supplies must enter the ecosystem in order to maintain or increase productivity. Phosphorus enters fresh water from atmospheric precipitation and from groundwater and surface runoff, and there are numerous sources of input of nutrients into aquatic systems (Figure 2.2); these can be grouped into:

i. Atmospheric, such as direct inputs of rain, aerosols, and dust;

ii. Point sources, including sewage treatment plants, drainage ditches, and trade effluents;

iii. Non-point sources, such as storm water, agricultural, and land clearing runoff;

iv. Non-point sources from within the water course, like washout from riverbanks and sediments and internal loading from sediments.

The loading rates of phosphorus vary greatly with patterns of land use, geology, morphology of the drainage basin, soil productivity, human activities, pollution, and other factors (Table 2.1). When phosphorus is added to waters that have low productivity, the usual response is a rapid increase in algal productivity.

Phosphorus plays a major role in biological metabolism, and when compared to other macronutrients required by biota, it is the least abundant and commonly the first element to limit biological productivity. In the majority of lakes, phosphorus is normally the limiting nutrient because the amount of biologically available phosphorus is small in relation to the quantity required for algal growth. However, in some areas (prairie lakes and rivers) nitrogen is increasingly being found to be the most important nutrient.

Table 2.1 Export coefficients ($kg \cdot km^{-2} \cdot a^{-1}$) for total phosphorus and total nitrogen for specific land uses in the Athabasca and Wapiti-Smoky River basins (source: Chambers and Dale 1997).

Land use	Total Phosphorus	Total Nitrogen
Forest	10	135
Cropland	25	150
Pasture land	50	300
Atmospheric loading	20	400

(*N.B. considerable increase in the export of phosphate occurs when forest is converted to cropland use and cropland is converted to pasture land)*

A major proportion of polluting nutrients entering rivers and lakes comes from sewage treatment plants, in untreated sewage or from farming activities. Sources might be discrete, such as a specific sewage out-fall, or diffuse, such as from farmland within the catchment. Natural processes within the watershed can also contribute significant amounts of phosphorus.

In forest habitats, the impact of forest harvesting and wildfire on streams and lakes water quality has been well studies in Canada. Post logging and post fire monitoring in Alberta, Ontario, and Québec showed an increase in the export of nutrients in water systems in comparison to undisturbed watersheds (Bayley et al. 1992; Carignan et al. 2000; Lamontagne et al. 2000; McEachern et al. 2000). The increased leaching from those events have resulted in the variation of the aquatic food web composition of the water systems (Carignan and Steedman 2000; Nicholls et al. 2003).

Nutrients from urban sources may be derived from domestic sewage, industrial wastes and storm drainage. Industrial sources of nutrients may be locally important, depending on the type of industry, the volume of effluent and the amount of treatment it receives.

Rural sources of nutrients include those from agriculture, forest management and rural dwellings, of which the first is the most universally important. Rural dwellings generally dispose of their sewage into septic tanks, which might cause local pollution. Nutrients are lost from farmland in three ways:

i. By direct surface runoff of fertilizers and manure;

ii. Through leaching of soluble plant nutrients by drainage water percolating through the soil; and

iii. By the erosion of surface soils or by the movement of fine soil particles into subsoil drainage systems.

2.3.1 Surface waters

In unpolluted rivers dissolved inorganic phosphorus averages about 10 $\mu g \cdot L^{-1}$ wordwide, with total dissolved inorganic phosphorus averaging about 25 $\mu g \cdot L^{-1}$ (Wetzel 2001). The concentrations often increase by a factor of two-to four-fold during and following increases in discharge from heavy rainfall events or in the early stages of snow melt. Dissolved inorganic phosphorus levels of 50-100 $\mu g \cdot L^{-1}$ from agricultural runoff to over 1000 $\mu g \cdot L^{-1}$ from municipal sewage sources have been reported (Meybeck 1982). While nutrients are transported downstream they are cycled among biota and abiotic components of the river ecosystems. The transportation mechanism resembles spirals and has been termed nutrient spiraling (Wetzel 2001). Although upstream movements of nutrients can occur in backflows from eddies, fish migration, and flight of adult aquatic insets, net flexes are downstream.

Rivers differ greatly from lakes in their response to nutrient enrichment depending on a number of physical features, especially substrate type, depth, water clarity, velocity, shading, etc. The phosphorus comes from the river bottom, banks, and flood plains during the storm event (Verhoff et al. 1982). There are two views of chemical transport during storms: (i) the continuous flow theory, which envisions the chemical to be washed from land, through the river system, and into the receiving water body during a storm event; and (ii) the discontinuous theory which proposes that the chemical is moved from the land through the stream by a series of flood waves. The first wave would carry material from the land into the riverbed. The second wave would resuspend the chemical, carry it to some distance, and redeposit further downstream. This process continues until the chemical reaches the receiving water body. Verhoff et al. (1982) showed that the peak in TP concentration almost always precedes the flow peak of the river. TP concentration also declines to its low flow level before flow returns to its normal range of steady flow rates. Evidence from Verhoff's (1982) study showed that the transport of TP occurs via a mechanism by which phosphorus is picked up at one point in the stream and deposited in another. Given this, the distance of transport becomes an important issue.

Long-term evaluations of phosphorus dynamics in rivers are rare. Examination of an average phosphorus budget for a forested second-order stream ecosystem of New Hampshire identified that dissolves and fine particulate phosphorus (63%) dominated phosphorus inputs, whereas falling and windblown litter comprised 23% of the total input (Wetzel 2001). In this ecosystem, subsurface inflows (10%) and precipitation (5%) contributed relatively small sources of phosphorus input. No annual net retention of phosphorus occurred in this stream, however, over shorter time periods, inputs exceeded exports. The accumulated phosphorus was exported in large pulses during precipitation-mediated episodes of high-stream discharge.

Lakes have contact with their drainage basins via both epilimnion and hypolimnion. Phosphorus enters with in-flowing water and leaves with out-flowing water and by sedimentation. It has been demonstrated that phosphorus in the epilimnion is extremely mobile (Rigler 1964). Rigler (1956) showed that in Toussaint Lake, a small acidic lake in Ontario, with a well developed littoral zone of rooted plants, the littoral region was the most important contributor to the turnover of phosphorus in the epilimnion. Phosphorus was lost to this compartment 10-times more rapidly than through the outlet. Return of phosphorus from the littoral zone during the summer was about 20% higher than loss. From this and other work, it is clear that the littoral flora play an important role in the dynamics of the phosphorus cycle; however, there are conflicting conclusions on the importance of the littoral flora to phosphorus circulation in lakes. In experimental pond systems, filamentous algae of the simulated littoral zone were a major site of phosphorus uptake before it reached the epilimnion (Chamberlain 1968). As the filamentous algae decayed, phosphorus was released to the open water. In this relationship, the equilibrium exchange system between phosphorus input to or release from the littoral flora and from or to the epilimnetic water varies with the amounts of phosphorus input from the drainage basin. Also, phosphorus uptake by and input from littoral macrophytes and attached algae will vary with the physical constraints of littoral development determined by basin morphology.

It is important to note that the primary productivity of rivers is generally much less responsive to increased nutrient loadings than lakes and that nutrient availability is generally higher in rivers than lakes (Wetzel 2001). Nutrient losses in lakes are greater and large quantities are stored in the parts of the ecosystem that are not readily available to the producers. In streams where nitrogen may become limiting to algal growth with increased phosphorus loading, it is likely that denitrification will be less effective in reducing stream nitrogen in low order streams because of shorter contact time with anoxic environments.

TP concentrations in non-polluted natural waters extend over a very wide range from less than 1 $\mu g \cdot L^{-1}$ in very unproductive waters, to greater than 200 $\mu g \cdot L^{-1}$ in highly eutrophic waters; however, most uncontaminated freshwaters contain between 10 and 50 $\mu g \cdot L^{-1}$ of TP (Wetzel 2001). Variation is high and can be related to regional geology; phosphorus levels of freshwaters are generally lowest in mountainous regions of bedrock geomorphology (such as in the Boreal and Pre-Cambrian Shields), and increase in lowland waters derived from sedimentary rock deposits (e.g., the Boreal plains of Alberta). Lakes rich in organic matter, and bogs, tend to exhibit high TP concentrations.

Total and soluble phosphorus shows little variation with increasing depth in oligotrophic lakes. Eutrophic lakes commonly show a marked increase in phosphorus in the lower hypolimnion. Much of this increase is from soluble phosphorus near the sediment-water interface.

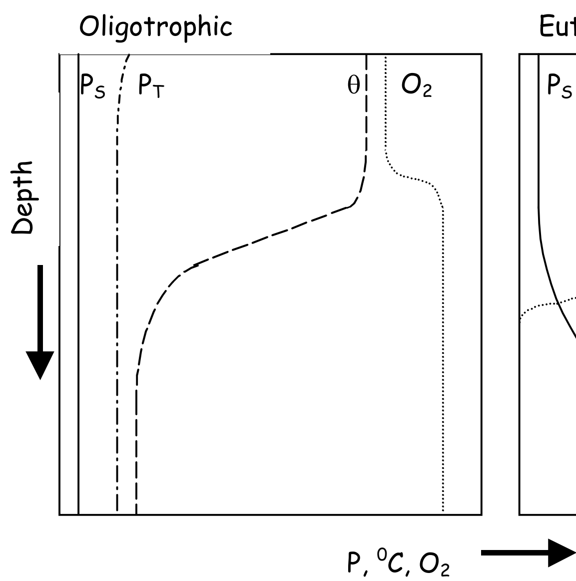

Figure 2.3 Generalized vertical distribution of soluble (P_S) and total (P_T) phosphorus in stratified lakes of low and high productivity. Oxygen (O_2) and thermal (θ) profiles are also provided (source: Wetzel 2001).

2.3.2 Exchange at the sediment/water interface

Precipitation and sorption of nutrients onto sediments are physico-chemical processes that influence the concentrations of some ions. Phosphorus and nitrogen behave differently in sediments. Sedimentation of particulate phosphorus represents a slow but continuous loss from the epilimnion. Abiotic transformations can have a strong influence on phosphorus, and a lesser influence on ammonium-nitrogen, whereas nitrate-nitrogen apparently is affected little (Mason 1991). The nitrate ion is fairly mobile because of the predominantly negative charge on sediment particles and is readily leached, if it is not taken up by plants. By contrast, phosphate is precipitated as insoluble iron, calcium or aluminium phosphate and then released slowly. The solubility of nitrate means that direct agricultural runoff is a major contributor to nitrate loadings on fresh waters. Loss of phosphate by leaching from agricultural land is less than that for nitrogen, and its input to fresh waters is largely from sediment erosion. Arable farming increases the natural rate of erosion of land because the soil is often bare for several months of the year. Much of this phosphate is tightly bound to the soil particles and is not immediately biologically available when it reaches fresh waters. The solubility of phosphate is enhanced when the soil particles become incorporated into anaerobic mud. In waters with a heavy precipitation of carbonates, phosphates and many micronutrients form insoluble compounds that precipitate to the bottom sediments.

Exchange of phosphorus across the sediment/water interface is regulated by oxidation-reduction (redox) interactions, which are dependent on oxygen supply, mineral solubility and sorptive mechanisms (Stumm and Morgan 1996), the metabolic activities of bacteria and fungi, and turbulence from physical and biotic activities (Wetzel 2001). The primary mechanism by which organic phosphorus is converted to inorganic phosphate in the sediment is through bacterial mediated breakdown of organic matter. Such mechanisms also create the reducing conditions required for the release of phosphorus into the water (Wetzel 2001).

As the oxygen content of the water near the sediment interface declines, the oxidized microzone barrier of the sediment weakens. The release of phosphorus, iron, and manganese increases markedly as the redox potential decreases. With the reduction of ferric hydroxides and complexes, ferrous iron and adsorbed phosphate are mobilized into the water column. The same general reactions are observed in the hypolimnetic water just overlying the sediments of eutrophic lakes.

In all but the upper few centimetres of sediment, exchange is slow and controlled by low diffusion rates. If the water above the sediment is oxygenated, an oxidized microzone is formed below the sediment/water interface (0-5 mm), under which the sediments usually become extremely reducing. This microzone effectively prevents phosphorus (which becomes soluble under reducing conditions) from migrating upward into the water column. In productive lakes, the oxidized microzone is lost, as the hypolimnion becomes anoxic. The release of phosphate and ferrous iron into the water occurs readily when reducing conditions reach a redox potential of about +200 mV (Wetzel 2001).

Precipitation of phosphorus as iron and aluminium phosphates is important only at relatively high levels of dissolved phosphorus (about 25-100 $\mu g \cdot L^{-1}$) (Stumm and Morgan 1996) and hence is unlikely to be substantial in relatively unpolluted waters. At low oligotrophic levels of dissolved phosphorus (<10 $\mu g \cdot L^{-1}$), sorption of organic and inorganic phosphorus compounds by sediments occurs rapidly, particularly in fine grained sediments, and may effectively regulate water concentrations of dissolved phosphorus (Meyer 1979).

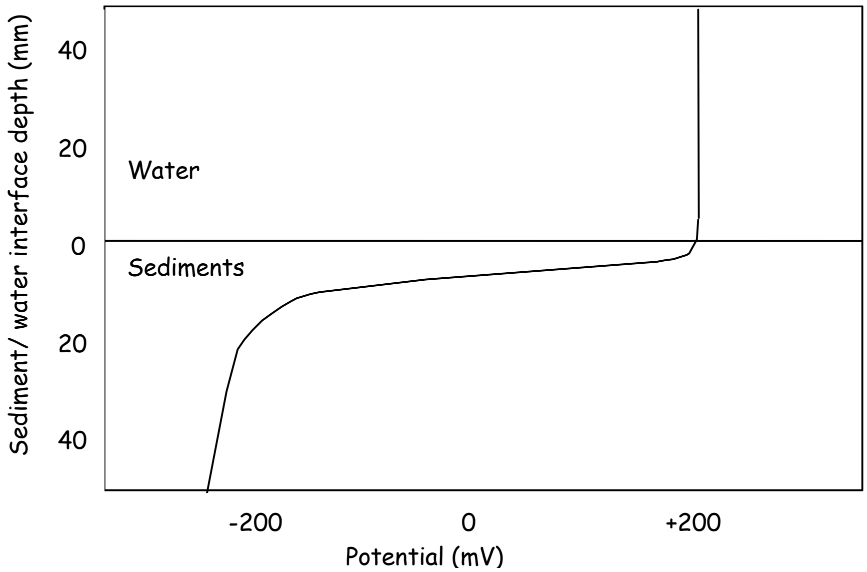

Figure 2.4 Diagrammatic profile of composite electrode potential across the sediment-water interface (source: Wetzel 2001).

Soluble phosphorus can accumulate in large quantities in anaerobic hypolimnion. Autumn circulation results in the rapid oxidization of ferrous iron and the precipitation of much of the phosphate as ferric phosphate. The movement of phosphorus from the sediment interstitial water can be accelerated by physical turbulence and by biota (bioturbation). Rooted aquatic macrophytes often obtain phosphorus from the sediment and can release large amounts into the water during growth and senescence and death. High populations of benthic biota can also increase the exchange of phosphorus across the sediment/water interface as a result of burrowing activities, etc.

Mass balance models can provide a reasonably accurate estimation of permissible phosphorus loading needed to achieve a certain level of productivity. In certain shallow lakes with greater than average turbulence, large littoral areas and small anaerobic hypolimnia, reduced productivity does not always occur as rapidly in response to decreased loading as predicted from the models. In these lakes, phosphorus release from the sediment ("internal sources") is much greater than values (10–30% of total loading) for deeper, more stratified lakes.

2.3.3 Sediment interactions

Lake sediments contain much higher concentrations of phosphorus than water. Under aerobic conditions, the exchange equilibria are largely unidirectional toward the sediments; however, under anaerobic conditions, inorganic exchange at the sediment-water interface is strongly influenced by redox conditions (Wetzel 2001). The quantitative effect of sediments on lake-water composition depends to a large extent on the volume/surface area ratio (Salomons et al. 1987). For example, in shallow lakes with high surface areas, the sediment influence is expected to be much stronger compared to deep lakes.

The diagenetic chemical reactions occurring within the sediments consist of abiotic and biogenic reactions (Stumm and Morgan 1996). Because of these reactions, sediments exert a significant effect on the overlying waters. The function of sediments in phosphorus dynamics is important if a complete understanding of the eutrophication process is to be achieved. Most minerals are transported as solutes in the water column, although phosphorus, hydrophobic organics and many trace metals are transported mainly as particulates. Experiments with phosphorus isotopes have shown that phosphorus is rapidly removed from the water column (Allen 1995), and the exchange of phosphorus between sediments and the overlying water is a major component of the phosphorus cycle in natural waters. The effectiveness of a net sink of phosphorus to the sediments and rapidity of regeneration of phosphorus back to the water column depends on an array of chemical, physical, and metabolic factors. Traditionally, aerobic oligotrophic lake sediments have been considered as a sink for phosphorus. Sediments in eutrophic waters, where the phosphate release rate is high when the sediments turn anaerobic (Lijklema and Hieltjes 1982), have been regarded as a source of phosphorus for the water column (Mortimer 1971).

There is little correlation between the amount of phosphorus in the sediment and productivity of the overlying water; phosphorus concentrations can be several orders of magnitude higher than the levels observed in the water (Wetzel 2001). Some important factors affecting the phosphorus concentrations in the sediment include:

i. The ability of the sediments to retain phosphorus (i.e., Fe content, TOC, particle size, etc.);

ii. The conditions of the overlying water (e.g., aerobic state); and

iii. The biota within the sediments that can alter the exchange equilibria and thus effect phosphorus transport back to the water.

In rivers, rapid variations in concentrations of dissolved nutrients as well as suspended solids, suspended bacteria, and biological oxygen demand can occur. Work carried out by Casey and Farr (1982) showed that during periods of artificially increased flow, substantial variations (greater than under normal flow) were observed in suspended solids, bacterial counts, BOD, dissolved organic carbon (DOC), nitrate, reactive phosphorus, dissolved silicon, and potassium. Little or no variation was detected for calcium and sodium concentrations. Variations in water chemistry due to such spates appeared to originate within the stream channel. Increased levels of dissolved phosphorus were observed during some natural periods of increased flow that could not be accounted for by allochthonous inputs. It was concluded that release of sediment interstitial water, enriched in dissolved phosphorus, was the most likely cause. This suggested that decaying organic matter, decomposition, and solution of phosphate-containing minerals can occur within organically rich and anaerobic sediments.

Sediment phosphorus release (internal loading) can be an important source of phosphorus and can maintain high water column phosphorus concentrations, even in the absence of significant external loading (Marsden 1989; Holz and Hoagland 1999).

Increased decomposition rates in deep waters of eutrophic lakes during productive summers causes anoxia which destroys the oxidized microzone and reduces iron to the ferrous form. This results in the release of phosphate that was formally immobilized as a complex with ferric hydroxide in the sediments. Turnover of the hypolimnion in fall restores the oxic conditions to the sediment-water interface, and allows the ferric hydroxide complexes to sequester sedimentary phosphorus again.

2.3.4 Ground water

Under natural conditions, phosphorus concentrations in ground water are generally low; average concentrations of about 20 $\mu g \cdot L^{-1}$ (Wetzel 2001). This is mainly because of the relatively insoluble nature of phosphorus containing minerals and the scavenging of surrounding phosphate by biota and soil particles. The fate of phosphorus in sewage effluent prior to and after entering the groundwater can vary greatly depending on the local hydrogeological conditions. Similarly, ground water in areas containing septic systems, particularly those near lake or river locations, can contribute high phosphorus loadings. Approximately 25% of Canadian are served by the septic disposal systems and these systems are sources of contamination to ground water, and ultimately to surface water (Chambers et al. 2001). The degree to which phosphorus is retained in a septic field depends on the age of the field and soil characteristics, including adsorption capacity, natural drainage and permeability. Phosphate migration in the saturated zone appears to be controlled primarily by sorption processes that significantly retard migration (Robertson 1995; Harman et al. 1996). However, as the adsorption capacity becomes saturated, migration of phosphate in the ground water zone may occur; low but perceptible rates of phosphate movement have been observed at several sites (Robertson 1995). An understanding of phosphorus movement in groundwater would assist decision makers in applying phosphorus loading models, particularly in areas where development is restricted by phosphorus loading.

2.4 Parameters Affecting Phosphorus Availability

2.4.1 Seasonal variation

Variation in discharge on both seasonal and annual time scales influences whether nutrients are stored or exported. Subsurface flow and storage zone mechanisms constitute an important aspect of hydrological influence over nutrient dynamics. Because abiotic and biotic retention mechanisms operate principally at or within the streambed, especially in smaller rivers and streams, hydrologic processes that favour exchange between the surface and interstitial waters will enhance the retention of recycling nutrients.

Phosphorus turnover rates can be very rapid in summer periods of high demand and relatively low inputs, but become as much as two orders of magnitude slower during winter periods. Phosphorus turnover rates are faster under oligotrophic conditions of greater phosphorus deficiency. A generalization of the major phosphorus fluxes is provided in Figure 2.5.

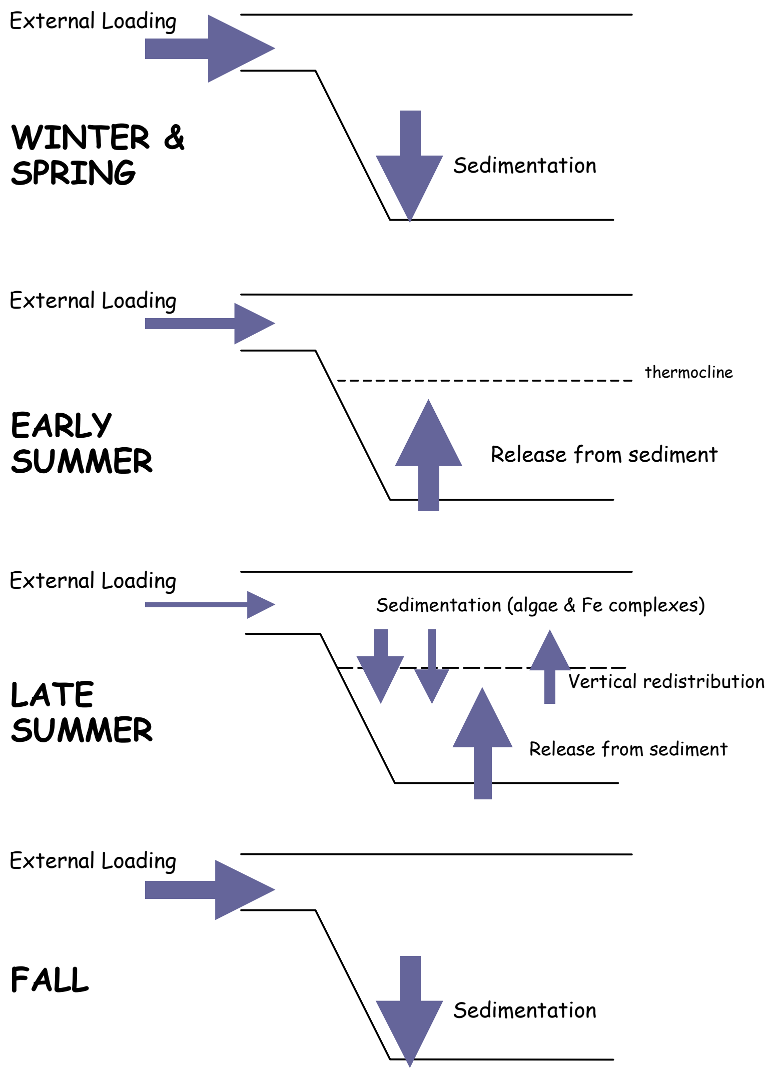

Figure 2.5 Generalization of the major seasonal phosphorus fluxes. The relative importance of exchange is indicated by the thickness of the arrows (source: Kennedy et al. 1986).

During winter and spring (Figure 2.5), phosphorus loading is high and sedimentation results in the establishment of longitudinal concentration gradients. In early summer, flow and external phosphorus loads decrease. The development of anoxia above shallow upstream sediments leads to the release of phosphorus and iron and to establish marked vertical gradients. Phosphorus transport across the thermocline may account for phosphorus increase in the euphotic zone (Kennedy et al. 1986).

Phosphorus inputs from the river directly to the epilimnion during the summer are minimal. Losses of phosphorus occur by particulate settling and by precipitation of ferric hydroxide/ phosphate complexes. Major phosphorus fluxes during the summer occur between the sediment and the overlying water, and between the epilimnion and hypolimnion. In late summer, destratification begins in the shallow upstream areas and proceeds downstream, accompanied by the precipitation of iron and phosphorus. External phosphorus loads increase with increasing runoff. The major phosphorus flux in the fall again is from the water column to the sediment.

There is a brief period of time (early summer) when the sediments of lakes have a high release rate; this occurs during the transition from aerobic to anaerobic conditions in the top layer. This dynamic behaviour is typical for sediments in which iron plays an important role in binding phosphate. Due to the dissolution in the anaerobic zone and precipitation of iron at the interface with the aerobic layer, a zone, enriched with phosphate adsorbed on freshly precipitated iron, migrates gradually towards the sediment-water interface when sediments turn anaerobic during the growing season. The pH plays an important role in the adsorption when this shifting zone reaches the interface, where the pH is much higher, a high flux of phosphate can result (Lijklema and Hieltjes 1982).

2.4.2 Hydrology

Hydrologic regime influences phosphorus uptake by its effect on the standing stock biomass and productivity of the biological community, and thus the amount of material stored in biological tissue. Similarly, storms can influence phosphorus uptake by affecting the standing stock, contribution of leaf litter and other organic matter. Hydrology also interacts with in-stream retention devices and geomorphological features of river channels to determine the relative importance of transport and storage to phosphorus dynamics. Retention and uptake is favoured by low flow, a high ratio of streambed area to channel volume, retention devices such as debris dams and beaver ponds, and permeable substrates that allow substantial interstitial flow. Through-flow of inputs and export of stored materials are favoured by the opposite conditions.

2.4.3 Ion exchange

Degradation of organic matter and the transformation of iron hydroxides to iron sulphides can cause a reduction in the total adsorption capacity. A prime medium for sorption of inorganic components by sediment is the metastable iron and manganese oxides, which have a high degree of isomorphic substitution and are a substrate for both inorganic and organic constituents (Damiani et al. 1987). Fe/Mn oxyhydrates are present in all parts of the hydrological cycle. Fe/Mn oxides are formed in stratified lakes and estuaries, and at the sediment/water interface. Differentiation under changing redox conditions is the main determinant of the relative accumulation of trace elements in Fe/Mn concentrations in both marine and freshwaters.

The mineral or chemical composition of phosphate compounds tends to be different in different environments. Variscite ($AlPO_4 \cdot 2H_2O$) and strengite ($FePO_4 \cdot 2H_2O$) appear to be more common in soils and freshwater sediments, whereas apatite ($Ca_{10}(PO_4)_6(OH)_2$) prevails in certain marine sediments (Stumm and Morgan 1996). In areas of high organic productivity in sediments of eutrophic lakes and shallow areas of the ocean, calcium phosphate minerals are deposited. The main sink of phosphates in the oceans and in many lakes consists of Fe(III) oxides on the surface of which phosphates become chemisorbed (Stumm and Morgan 1996). Ferric iron sequesters phosphate in an oxidized microzone at the sediment surface in an oligotrophic lake.

Remineralization of organic nitrogen and phosphorus results in increases in ammonia and phosphate below the sediment/water interface. The increase in alkalinity is also a result of the decomposition of organic matter either directly (as organic nitrogen is remineralized to NH_4^+) or indirectly (as calcite dissolves in response to the release of CO_2 associated with remineralization of organic carbon) (Figure 2.6).

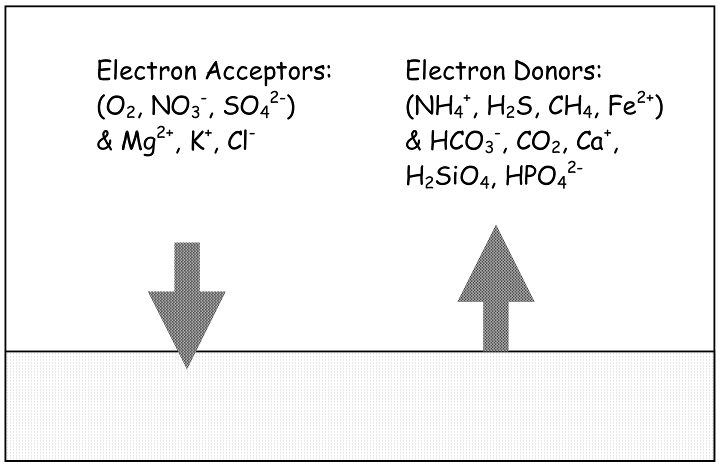

Figure 2.6 Direction of fluxes expected for dissolved constituents between sediment pore waters and the overlying waters (source: Stumm and Morgan 1996).

Precipitation of phosphorus as iron and aluminium phosphates is important only at relatively high levels of dissolved phosphorus (in the region of 25-100 µg·L^{-1}) and hence is unlikely to be substantial in relatively unpolluted areas (Allen 1995). At low levels of dissolved phosphorus, the sorption of organic and inorganic phosphorus compounds by sediments occurs rapidly, particularly in fine-grained sediments, and may effectively regulate stream water concentrations of dissolved phosphorus.

It is often assumed that the formation of calcium-bound phosphate is an extremely slow process; however, this is only true for high concentrations of phosphorus; evidence from lakes and rivers suggests that if suspended $CaCO_3$ is present, the reaction occurs in a few days. It has been shown in many Dutch lakes that calcium concentration and pH control the maximal solubility of phosphate (Golterman 1982). In hard waters the calcium ion is just as important as iron as a phosphate controlling mechanism.

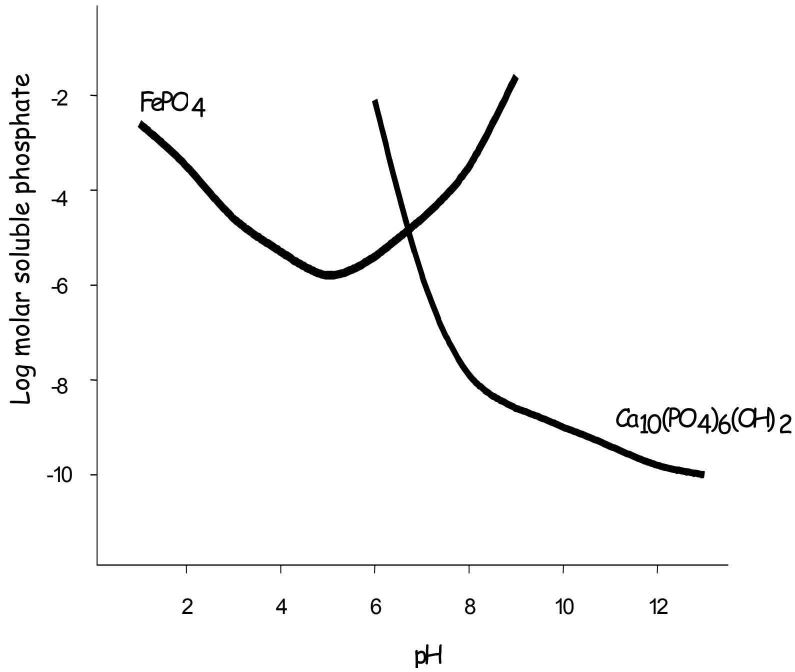

Figure 2.7 Solubility diagram for phosphate phases (source: Golterman 1982).

The curve shown in Figure 2.7 demonstrates that $FePO_4$ and $Ca_{10}(PO_4)_6(OH)_2$ cannot coexist under normal lacustrine conditions. Lower pH values may occur in the overlying water of shallow lake sediments, and iron may control the phosphate concentrations there. However, as algal growth takes place in the water, in the most eutrophic shallow lakes, the calcium ion concentration and pH will control the solubility of phosphate surrounding the algae. It is therefore essential that, for hard water eutrophic shallow lakes, the solubility product of apatite is set as a threshold for the permitted phosphate concentration (Golterman 1982).

2.4.4 pH

Changing pH can affect the complexing of metal ions with phosphates. Increasing pH leads to the formation of calcium carbonate, which co-precipitates phosphate with carbonates (Wetzel 2001). Elevation of the pH of waters containing typical concentrations of calcium lead to apatite formation, and in general, high phosphate adsorption by clays is favoured by low pH levels (approximately 5 to 6).

The distribution of the several acid and base species of orthophosphates and condensed phosphates in solution is governed by pH. Strengite and variscite are stable solid phases if phosphate is precipitated out in the low-pH range. In neutral pH range, metastable hydroxyphosphate aluminium(III) or iron(III) precipitates can be formed (Stumm and Morgan 1996).

Many species of algae exhibit optimal growth and uptake of phosphorus within a distinct range of pH. The pH may alter the rates of phosphate absorption through direct effects on the activity of enzymes, on the permeability of the cell membrane, or by changing the degree of phosphate ionization. Uptake rates of phosphorus have been correlated directly with the presence of numerous ions and compounds of the water, such as potassium, micronutrients, and organic compounds.

2.4.5 Oxygen-redox interactions

Oxygen content at the sediment/water microzone is influenced primarily by the metabolism of bacteria, fungi, and planktonic invertebrates that migrate to the interface, and by sessile benthic invertebrates. Sediment oxygen demand is high and governed by the intensity of microbial and respiratory metabolism, slow rates of diffusion and from the fact that inorganic elements (e.g., Fe^{2+}) accumulate in reduced form when released into the sediment from decomposing biota. Microbial degradation of dead particulate organic matter that settles into the hypolimnion and onto the sediments is primarily a consumptive oxygen process in the deepwater areas of lakes. The rate of oxygen depletion is governed by the rates of organic loading to the hypolimnion and the lake or reservoir morphology.

At the sediment surface, a difference of a few millimetres in oxygen penetration is a critical factor regulating exchange between the sediment and water. The oxidized layer forms an efficient trap for iron and manganese, as well as phosphate, thereby greatly reducing the transport of minerals into the water and scavenging minerals such as phosphate from the water. The release of phosphorus, iron, and manganese increases markedly as redox potential decreases. Manganese is reduced and mobilized at a higher redox potential than iron.

The concentration of dissolved iron increases in lakes following the onset of anoxia in early June, and reaches a maximum in late August. Soluble reactive phosphorus (SRP) concentrations also increase steadily during this period. If sediment phosphorus was associated with hydrous iron oxides, as suggested by Mortimer (1971), ferric Fe(III) reduction following the onset of anoxia would result in concurrent increases in phosphorus and ferrous Fe(II), as appears to be the case in Lake DeGray, south central Arkansas, USA (Kennedy et al. 1986). The rapid reduction in the concentrations of iron and phosphorus immediately after destratification and reoxygenation of the bottom waters suggests precipitation of phosphorus with hydrous iron oxides.

Since the formation of hydrous iron oxides can occur relatively rapidly, oxidation and precipitation would prevent the introduction of significant quantities of hypolimnetic iron to the surface water during deepening and eventual loss of the thermocline in late summer (Stumm and Morgan 1996). Phosphorus, not immediately removed by co-precipitation with iron, would be redistributed to surface waters (Figure 2.8).

Kennedy et al. (1986) showed in DeGray Lake that anoxic sediments appear to be a significant source of phosphorus in the upper portion of the lake in the summer (Figure 2.9). Concurrent increases in soluble iron and SRP immediately above the sediments strengthened the hypothesis that anoxic sediments play an important role in phosphorus dynamics.

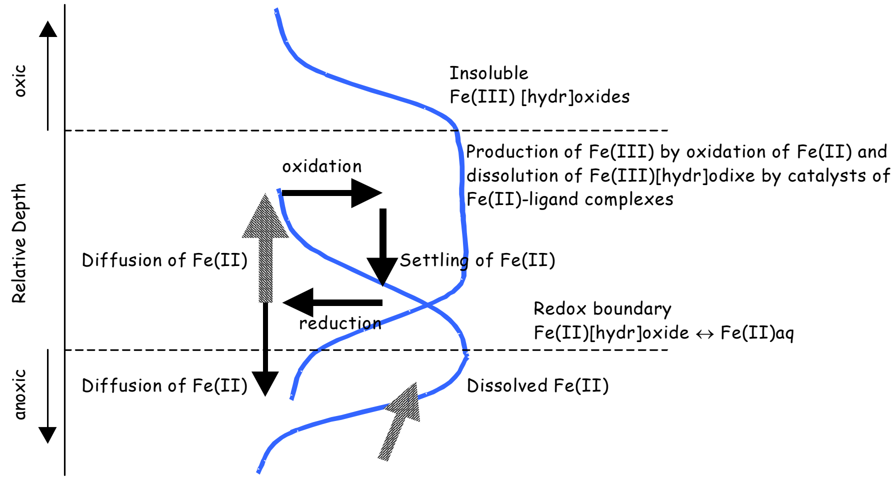

Figure 2.8 Redox conditions in natural waters: transformation of Fe(II, III) at an oxic-anoxic boundary in the water or sediment column (source: Stumm and Morgan 1996).

A relation between profundal fauna (especially Chironomidae) and the degree of productivity (trophic status) in lakes has been established (Mortimer 1941). Hutchinson et al. (1939) suggested that redox potential is an important determining factor, and demonstrated a relationship between the chironomid population in a series of lakes and the redox potential of the bottom water during thermal stratification.

2.4.6 Variables that determine trophic status

Like all ecosystems, standing waters are complex and characterized by a huge number of physical, chemical, and biological variables. The location and size of the catchment of the water-filled basin determine the flow-through of water and its retention time in the basin. In turn, this determines the potential for plankton production (Johnes et al. 1994). In general, regional chemical characteristics of surface waters are closely related to the soil characteristics of their drainage basins (Vollenweider 1968). Soils reflect the regional geological and climatic characteristics. Surface drainage is often a major contributor of phosphorus to streams and lakes. The quantities of phosphorus entering surface drainage vary with the amount of phosphorus in soils, topography, vegetative cover, quality and duration of runoff flow, land use, and pollution.

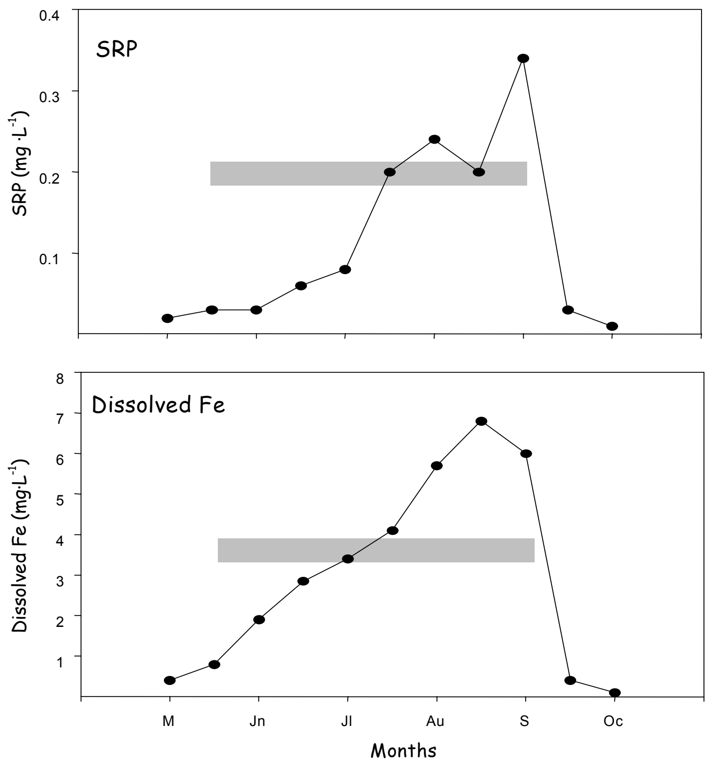

Figure 2.9 Changes in soluble reactive phosphorus (SRP) and dissolved iron. Shaded area represents period of anoxia (source: Kennedy et al. 1986).

The phosphorus content of precipitation and of particulate material fallout is highly variable, and results in variable phosphorus loading to lakes. In general, the contribution of phosphorus from precipitation is less than that of nitrogen. The major source of phosphorus in precipitation is from dust generated over the land from soil erosion, and from urban and industrial contamination of the atmosphere. Although the phosphorus content of precipitation is generally low, it may constitute a significant part of the total loading (Wetzel 2001). Concentrations and loads of phosphorus in precipitation do not correlate with land use as consistently as does nitrogen (Castillo et al. 2000). In catchments of Chesapeake Bay (USA), phosphorous concentrations did not correlate with land use, but correlated with sediment load, and the phosphorus export differed among watersheds, most likely due to geological differences (Jordan et al. 1997).

Basin size determines, among other things, the mixing regime and whether the water mass will stratify, the ratio of littoral:profundal habitats, the degree of wind disturbance of the bottom, the variety of littoral habitats, and hence the potential for survival of particular plant and animal species (Hutchinson 1967). Geology and topography of the catchment will determine the chemistry of the water that will drain from it, the land use to which it may be put, and therefore modifications that may be made to the geochemistry by land use (Moss 1988). Water chemistry will influence the nature and productivity of the biota, through the provision of nutrients (Schindler 1978). There are also a multitude of biotic factors which have a major effect on the composition of lake communities.

Perhaps the most important determinant of lake state is the nature of its catchment, which in turn is determined by its geographical location and geology, soil, and land-use of the catchment. The catchment determines the supply of nutrients to a lake and its ultimate productivity. Internal sources of such nutrients in the lake, which may become increasingly important as a lake shallows, are also ultimately supplied to the greatest degree by the catchment. The size of the catchment in relation to the size of the lake basin also determines the retention time and the effectiveness of the supplied nutrients in determining production (Dillon and Rigler 1974; Vollenweider 1975).

Vollenweider (1975) models are based on very simple input parameters on the relationship between phosphorus load and water renewal time of lakes, and resulting algal biomass, are good examples of predictive ecology for management. However, it is important to point out that considerable variability exists when data are plotted on a linear scale. This variability can be explained when examined on a regional to local basis because some lakes apparently offer optimal conditions for their algal communities, while others may support less than half the biomass at the same phosphorus load. There are limits to growth, however, and the predictive outcome of phosphorus loads decreases as the load increases (Andersen 1997). As the phosphorus load increases, there is a decrease in the efficiency of energy transfers and system stability tends to decrease. At some point, a waterbody will show a decrease or decoupling between elevated nutrient supplies and productivity at higher trophic levels. Often, a highly predictive need in this regard must be addressed at a site-specific level but much useful predictive management can be done at regional scales. Beyond the large-scale predictive relationships of the Vollenweider models, one must look to foodweb dynamics and its relationship to nutrient cycling and ecosystem stability for an improved understanding of phosphorus dynamics.

2.4.6.1 Chlorophyll

In most lakes there is a direct relationship between the concentration of the growth limiting nutrient and the maximum crop of phytoplankton (Horne and Goldman 1994). Simple empirical TP – chlorophyll models (Dillon and Rigler 1974) have been used to predict changes in chlorophyll concentrations (algal biomass) as a result of changes in TP concentrations, and a strong correlation has been shown between chlorophyll, TP and total nitrogen TN (TN) (Dillon and Rigler 1974; Canfield 1983; Brown et al. 2000; Vollenweider and Kerekes 1982). The ability to predict the change in phytoplankton from changes in TP is a useful approach, even though it is an empirical relationship that must be established for an individual lake or group of lakes. The theory of TP-chlorophyll relationship assumes that all phosphorus is recycled (Horne and Goldman 1994). The TP-chlorophyll relationship can be extended to predict the effect of the watershed on phytoplankton blooms. If phosphorus is the growth-limiting nutrient, the amount of TP entering the lake each year should be related to the phytoplankton standing crop and the relationship is termed as a *phosphorus loading cure*. However, the use of chlorophyll as a reliable predictor of phosphorus concentration should be viewed with the caveat that other work (Brown et al. 2000) has indicated a significant amount of variation in the yield of chlorophyll per unit TP.

2.4.6.2 Nitrogen

A shortage of nitrogen will often favour the development of nitrogen-fixing blue-green algae. Although TN seems to be an important factor influencing the yield of chlorophyll per unit TP (Canfield 1983), studies suggest that other environmental factors such as suspended solids (Hoyer and Jones 1983), aquatic plants (Canfield et al. 1984), flushing rates, light availability (Soballe and Kimmel 1987), and grazing effects (Shapiro 1979; Pace 1984) also need to be considered.

2.4.6.3 Nitrogen:Phosphorus ratio

The most important nutrient factors causing the shift from lesser to a more productive state are phosphorus and nitrogen. Typically organic matter of aquatic algae and macrophytes contains phosphorus, nitrogen, and carbon in approximately the ratios 1P:7N:40C per 100 dry weight or per 500 wet weight (Vallentyne 1974). In nutrient-poor freshwaters, inorganic phosphorus is often the principle factor limiting growth of algae and other primary producers. Nitrate-nitrogen tends to become limiting when phosphorus is plentiful and when the atomic ratio of N:P fall below a theoretical level of 16:1. In practice the shift from phosphorus to nitrogen limitation often occurs over a wider range (10-30:1) (Allen 1995). Bulgakov and Levich (1999) showed that high N:P weight ratios (20-50:1) can favour the development of *Chlorococcales*, while a reduction of the N:P ratio to values of 5 to 10 frequently leads to a community dominated by cyanophyta. Their model predicted that the relative abundance of different phytoplankton species depended only on the relative amounts of nitrogen and phosphorus in the environment, so that the optimal N:P ratio for a given species is equal to the ratio of its minimum cell requirements for these elements. In phosphorus-limited systems, an increase in phosphorus will result in an increase in productivity (Mason 1991). If nitrogen is limiting, some cyanobacteria (blue-green algae or cyanophytes) which are capable of directly fixing elemental nitrogen will proliferate, provided phosphorus is not limiting. Certain cyanobacteria typically increase under nitrogen limitation, sometimes to nuisance levels. The recent CCME (2002) draft document provides a detail discussion on the role of N:P ratios in regulating primary production in freshwater ecosystems.

2.4.6.4 Dissolved organic carbon

Dissolved organic carbon (DOC), especially humic materials that cause shading, may have diverse and powerful effects on ecosystem metabolism (Kalff 2002). DOC potentially limits primary production by shading (Jones 1992). Colored oligotrophic waters high in DOC have higher TP vs. chlorophyll ratio than that of clear waters, because a large proportion of the TP associated with DOC is not available to phytoplankton (Kerekes 1980; 1981). Furthermore, DOC may interact chemically with iron and phosphorus to limit nutrients available to plants (Carpenter et al. 1998b; Jackson and Hecky 1980). This suggests that in dystrophic waters (e.g., lakes in Nova

Scotia), eutrophication issues and application of the framework should be carefully evaluated.

2.5 General Effects of Eutrophication

Eutrophication causes marked changes in biota (Figure 2.10). The effects that have been observed in systems undergoing nutrient enrichment, and the problems to humans associated with these effects are summarized below, and in Table 2.2 (Mason 1991):

Effects on Aquatic Ecosystems

i. Decrease in biodiversity and changes in dominant biota;

ii. Decline in ecologically sensitive species and increase in tolerant species;

iii. Increase in plant and animal biomass;

iv. Increase in turbidity;

v. Increase in organic matter, leading to high sedimentation;

vi. Anoxic conditions may develop.

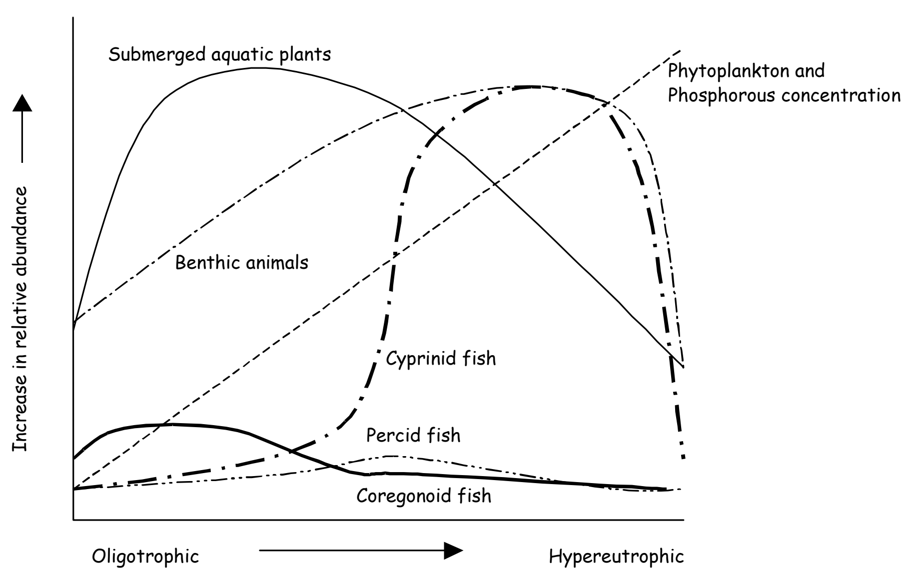

Figure 2.10 General changes in northern temperate lakes as they become eutrophic (source: Moss 1988).

Table 2.2 Summary of the problems associated with eutrophication of lakes, reservoirs, and impoundments (modified from GEMS 1990).

Quality of life concerns	Caused or indirectly dependent on: Algal blooms and species composition	Excessive macrophyte and littoral algal growth	Altered thermal conditions	Mineral turbidity	Low dissolved solids
Water quality impairment					
• Taste and odour, colour, filtration, flocculation, sedimentation and other treatment difficulties	Very frequent	Occasional	Occasional	Occasional	
• Hypolimnetic oxygen depletion; Fe, Mn, CO_2, NH^+_4, CH_4, H_2S, etc.	Frequent	Occasional	Occasional		
• Corrosion problems in pipes and other man-made structures	Frequent	Occasional		Occasional	Occasional
Recreational impairment					
• Poor aesthetics	Frequent	Occasional		Occasional	
• Hazard to bathers		Frequent			
• Increased health hazards	Occasional	Occasional	Occasional		
Fisheries impairment					
• Fish mortality	Occasional		Occasional		
• Undesirable fish stocks	Frequent	Frequent	Occasional	Occasional	
Ecological impairment	Very frequent	Frequent	Frequent	Occasional	Occasional
Aging and reduced holding capacity and flow					
• Silting	Occasional	Occasional		Frequent	
• Pipe and screen clogging		Occasional			

Quality of Life Concerns

i. Treatment of potable water may be difficult and costly;

ii. The supply may have an unacceptable taste or odour problem;

iii. The water may be injurious to health;

iv. The aesthetic/ recreational value of the water may decrease;

v. Macrophyte growth may impede water flow and navigation;

vi. Commercially important species (e.g., salmonoids, coregonids) may disappear.

2.6 Biological Interactions

Aquatic organisms influence the concentration of many substances directly by metabolic uptake, transformation, storage, and release. A number of biological processes influence the ability of aquatic ecosystems to utilize, retain, and recycle particulate and dissolved phosphorus. These interactions are summarized in Figure 2.11.

Dissolved inorganic phosphorus (DIP) is assimilated by plants and microbes into cellular constituents, thereby being transformed into particulate organic phosphorus (POP). Zooplankton, feeding on the seston, excrete soluble phosphorus and ammonia. Algae and bacteria rapidly utilize these nutrients. When phosphorus availability is low, this source of recycled phosphorus can be critical to the growth and succession of phytoplankton. Over 95% of phosphorus is bound in the particulate phase of living biota, particularly algae. Subsequently, this form of phosphorus (POP) may be excreted or released during cell lysis directly as DIP, or released as DOP, which is then broken down to DIP by bacterial activity.

Phosphorus availability is also influenced by physical and chemical transformations. Sorption of phosphate ions onto charged clays and charged organic particles occurs at relatively high DIP concentrations, while desorption is favoured by low concentrations. Such sorption-desorption reactions act as a buffer of DIP. Under aerobic conditions both DIP and DOP may complex with metal oxides and hydroxides to form insoluble precipitates. Phosphate is released under anaerobic conditions. The extent of the anaerobic zone tends to vary seasonally with organic matter loading; thus, the availability of dissolved phosphate also varies.

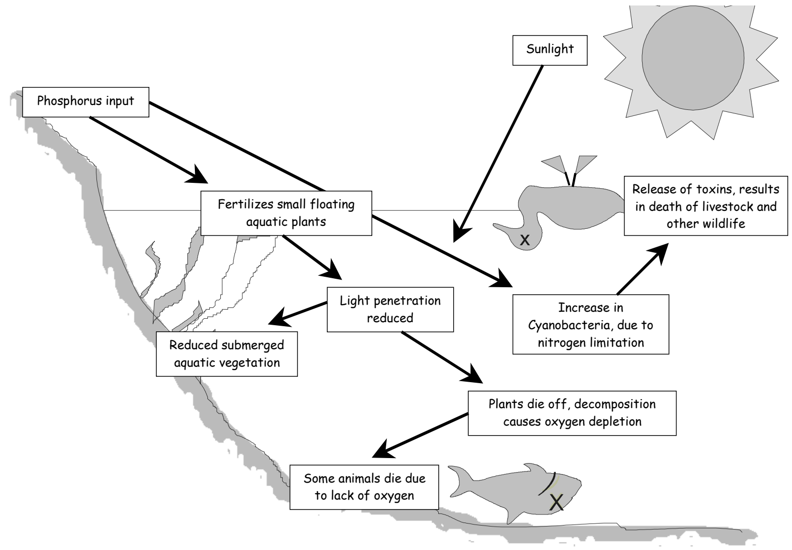

Figure 2.11 Biological effects of increased phosphorus loading to aquatic ecosystems.

2.6.1 Autotrophs

Primarily autotrophs and microbes accomplish the biological removal of nutrients from water. Wherever their biomass is large and populations are metabolically active, removal rates are highest. Aquatic macrophytes are capable of removing substantial amounts of phosphates from both flowing and standing waters.

The uptake of nutrients, and their utilization by microorganisms, depends not only on their physical and chemical properties, but also on the precise chemical speciation of each element. The rates at which elements are cycled in the aquatic environment depend upon the availability of bulk nutritional elements (carbon, nitrogen, oxygen, phosphorus, and sulphur). Solubility in water of these elements is strongly influenced by pH, temperature, and energy ($E°$), as well as by the presence of competing anions, cations, surface active polymers such as humic acids, fluvic acids, clays, etc.

Sediment microflora are important in raising the concentration of phosphorus dissolved in the interstitial water of sediments. Phosphorus mobilizing bacteria (*Pseudomonas, Bacterium, Chromobacterium*) vary in their abundance and vertical distribution with the type of sediment. Bacterial decomposition is proportional to bacterial densities at the interface and is directly related to the general productivity of the lake.

Ciliates associated with sediments are capable of hydrolyzing dissolved organic acids and releasing inorganic phosphate to the water. However, low oxygen concentrations not only produce an unfavourable environment for ciliates, but also inhibit the release of phosphate by cells.

The animal community influences phosphorus cycling in many ways. Figure 2.12 outlines the different compartments and interactions of nutrients in the sediments and the water column. Direct consumption of algal and microbial populations can either reduce or stimulate productivity and phosphorus uptake. Animals also affect phosphorus dynamics by returning phosphorus to the water by excretion, egestion, changing particle size, and resuspension through their movements. Direct consumption of periphyton and microbial populations reduces the standing stocks and reduces uptake rates (although in theory modest grazing may stimulate productivity and thus uptake rates as well). Consumers can also enhance the rate of regeneration of phosphorus. Excretion and egestion materials are most likely to contribute significantly to phosphorus dynamics in highly productive systems where phosphorus is scarce.

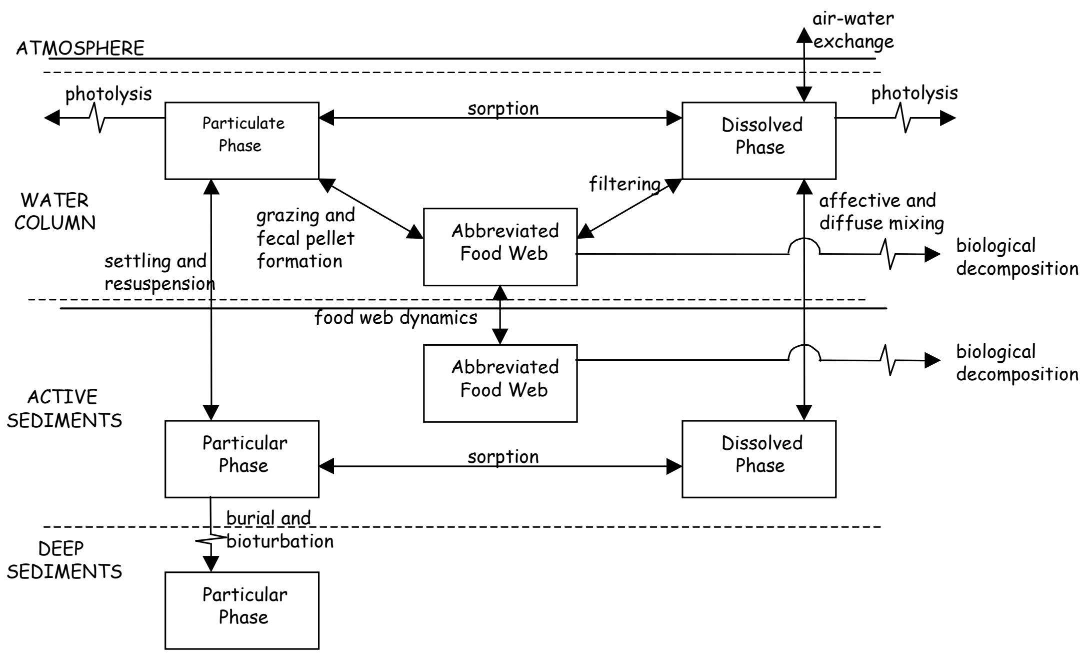

Figure 2.12 Compartments and interactions of nutrients (adapted from Damiani et al. 1987).

While bacteria are of major importance in the dynamics of phosphorus cycling in the water, their role in expediting phosphorus exchange across the sediment interface is relatively minor in comparison to chemical equilibria processes. Bacterial decomposition is proportional to bacterial densities at the interface and directly related to the general productivity of the lake. The sediment microfauna is important in increasing the concentration of phosphorus dissolved in the interstitial water of the sediments (Wetzel 2001). Algae growing on the sediments are able to effectively utilize phosphorus from the sediments and the presence or absence of bacteria has little effect on it.

Phosphorus in the photic zone is very rapidly incorporated into planktonic algae and bacteria. The recent kinetic studies (summarized in Wetzel 2001) of phosphorus uptake in natural assemblages indicate that: (i) bacteria have substantially higher phosphorus requirements than phytoplankton; (ii) bacterial consumption of phosphorus is larger than phytoplankton; (iii) bacteria contain greater amounts of phosphorus than algae; and (iv) bacteria can outcompete algae for phosphorus under a wide range of phosphorus supply rates.

2.6.2 Plankton

The concentration of chlorophyll *a* in the water is often taken as an index of the biomass of algae present. For oligotrophic lakes, the mean summer concentration of chlorophyll *a* in the epilimnion is in the range 0.3-2.5 $mg \cdot m^{-3}$, whereas for eutrophic lakes the range is 5-150 $mg \cdot m^{-3}$ (Mason 1991).

Algal requirements for phosphorus are variable among species and can lead to selective advantages of certain algae (Table 2.2). Algae must compete with bacteria for phosphorus sources; if organic substrates for bacterial growth are high, algal growth may be seriously impeded. Nutrient conditions that favour the establishment of cyanobacteria include high DIP and low dissolved inorganic nitrogen concentrations. Once established, bloom-forming cyanobacteria influence their surrounding environment physically, chemically, and biologically, in ways that favour their continued persistence (Mason 1991). These positive feedback mechanisms enable populations to grow rapidly and persist at high densities for long periods.

A spring increase in algae is followed by an increase in planktonic herbivores. The quality and quantity of food affect the ingestion rate of zooplankton. Cyanobacteria are assimilated very inefficiently by herbivorous zooplankton compared to green algae and diatoms. The bottom dwelling *Chydorus sphaericus* can eat cyanobacteria and its population often increases markedly during phytoplankton blooms. Larger cladocerans, such as *Daphnia*, filter much more efficiently than smaller plankton, such as *Bosmina*. This greater feeding efficiency may suppress smaller zooplankton when larger animals are abundant; however, predation by fish is usually on larger forms, so the smaller zooplankton may dominate.

Uptake of nutrients, especially nitrogen and phosphorus, by ingestion of food particles by herbivorous zooplankton can be significant at certain times. The nutrients are then excreted by zooplankton and rapidly re-assimilated by phytoplankton (phenomenon known as nutrient regeneration). When the concentrations of phosphorus in the trophogenic zone (the area supporting photosynthesis) are low, phosphorus and nitrogen regeneration by herbivorous zooplankton can contribute a significant portion of the nutrients required by the algae. The magnitude of nutrient uptake and recycling depends on the nutritional status of the algal cells (Lehman 1980). Removal of algae and bacteria by grazing zooplankton varies widely both seasonally and spatially, moreover, grazing of algae is often size selective. This selectivity may enhance the availability of nutrients to those algae that are not as effectively grazed. The interactions are further complicated by seasonal patterns of selective predation on larger zooplankton by planktivorous fish. As crustacean zooplankton body size has been shown to be correlated to chlorophyll concentrations (Pace 1984), a decrease in mean herbivorous zooplankton body size may decrease grazing pressure.

2.6.3 Macrophytes

Submerged, floating-leafed and emergent angiosperms are important to the dynamics of phosphorus cycling. Nutrient uptake is by the root-rhizome system in most aquatic plants. Phosphorus is released slowly from plants; release is more rapid from the epiphytic algae. The release of phosphorus from annual macrophytes at the end of the summer is an important source of phosphorus. For example, the leaching of phosphorus from dead macrophytes in sterile conditions was found to be rapid, resulting in a loss of 20 to 50% of TP in a few hours and 65 to 85% loss over longer periods (Wetzel 2001).

Table 2.3 Types of phytoplankton associated with oligotrophic and eutrophic lakes (source: Mason 1991).

	Algal Group	Examples
Oligotrophic lakes	Desmid	*Staurodesmus spp., Staurastrum spp.*
	Chrysophycean	*Dinobryon spp.*
	Diatom	*Cyclotella, Tabellaria*
	Dinoflagellate	*Peridinium, Ceratium*
	Chlorococcal	*Oocystis*
Eutrophic lakes	Diatom	*Asterionella, Fragillaria crotonesis, Stephanodiscus astraea, Melosira granulata*
	Dinoflagellate	*Peridinium bipes, Ceratium, Glenodinium*
	Chlorococcal	*Pediastrum, Scenedesmus*
	Cyanobacterial	*Anacystis, Aphanizomenon spp., Anabaena spp.*

N.B. Desmids are particularly important in lakes low in nutrients, while cyanobacteria often dominate in lakes with high nutrient concentrations. The diatom flora also changes.

Littoral flora can play a major role in the dynamics of the phosphorus cycle. Phosphorus uptake by and release from littoral macrophytes and attached algae will vary with the physical constraints of littoral development, which is determined by basin morphology. This relationship will shift in proportion to seasonal changes in the active growth of the littoral flora and subsequent decay of annual plants in late summer.

There is a direct relationship between nutrient loading and increased growth of both the macrophytes and phytoplankton and flora attached to parts of rooted aquatic plants (the periphyton). This often results in an accompanying decline in rooted, submerged macrophytes (Hough et al. 1989). In flowing water, instead of loss, excessive macrophyte growth may occur. This may include the large filamentous alga *Cladophora*, especially in waters rich in sewage effluents, and angiosperms such as species of *Ranunculus*, or in the tropics, water hyacinth (*Eichornia crassipes*). Plants may dominate a water body over a wide range of high nutrient concentrations and plants such as hornworts and water milfoils have been proposed as suitable for waste-water treatment lagoons, where nutrient levels will be extremely high.

2.6.4 Macroinvertebrates

The effect of benthic invertebrates on the dynamics of phosphorus cycling between the sediments and the water is not completely understood. The burrowing activities of benthic invertebrates can significantly alter the surficial oxidized zone by physical penetration. Studies of benthic macroinvertebrates have demonstrated the potential for these organisms to disrupt the sediment-water interface (e.g., Gellepp et al. 1978; Gellepp 1979). Most of the phosphorus released to the overlying water is readily assimilated by bacteria or algae (if light is available). Although the rate of phosphorus release from lake sediments increases markedly if the sediments are disturbed, phosphorus release caused by biotic disturbance is likely small compared to chemical-microbial regulation.

A reduction in diversity occurs in the benthic fauna with enrichment; there may also be changes in the seasonal pattern of occurrence, with consequences for organisms higher in the food chain. Under pristine conditions the main energy inputs into first- and second-order streams tend to come from terrestrial sources. As a consequence these systems have a higher proportion of macroinvertebrate shredders. Periphyton biomass and the proportion of grazers are greater in higher order streams (Vannote et al. 1980). The effects of nutrient enrichment from human activities alter energy inputs into lower order streams resulting in a shift in the benthic fauna to that resembling higher order stream communities.

Reworking of sediments by organisms such as chironomids, which burrow into the sediment and filter water through their burrows, and oligochaetes, which feed in the top few centimetres and defecate on the sediment surface, creates a smoother sediment profile than that found in sediments without these organisms. Oligochaetes are tolerant of organic pollution and tend to be more numerous than other organisms in highly polluted areas. Since the influx of chemicals is often accompanied by an increase in nutrients, worms can attain very large population densities and cause vigorous stirring of sediments. This makes phosphorus removal time hard to measure, and leads to a longer burial time for many sediment-associated pollutants (Thomas 1987).

The role of bioturbation in phosphorus exchange was described by Kamp-Nielsen et al. (1982) as the downward movement of easily degradable fresh sediment with high microbial activity from the oxic zone to a deeper more reduced zone with lower activity. If the transport is followed by a downward oxygen transport, an increase in phosphate sorption can be expected. *Chironomus anthracinus* will reduce phosphate release from the sediment. An upward transport of subsurface sediment will expose more refractory organic material, and an increased release can be expected. The upward movement will be small because the degradable part of the subsurface sediment is small and can be expected as a result of the bioturbation activities of *Potamothrix hammonensis*. Oxygen conditions determine whether the upward diffusing phosphate will be trapped in the surface sediment or not, and a simultaneous downward transport in oxygen will affect the exchange rates, which depend on the oxygen conditions at the sediment surface.

Movements and migrations by animal populations can result in either inputs or outputs of phosphorus to and from the sediments. Emergence of the adult stages of aquatic insects is one such process. In contrast spawning fish may import substantial amounts of phosphorus to streams and lakes by excretion, release of gametes, and their own mortality, especially if they die after reproducing (Allen 1995). Death due to predators and their subsequent excretion of nutrients accomplish the same end. Other influences of biota include assimilation and regeneration, and are central processes in the cycling of all nutrients.

2.6.5 Fish

Certain fish, such as coregonids (whitefish) are dominant in oligotrophic lakes and their yield increases in the early stages of enrichment and then declines. There is some increase in the abundance of percids in the intermediate stages of eutrophication but then they also decline. The yield of cyprinid fish increases sharply at intermediate stages of eutrophication and falls off sharply in heavily eutrophic waters (Figure 2.10).

Fish can impact the distribution of phosphorus in aquatic ecosystem in several ways (Wetzel 2001). Although the total amount of P in fish communities is small in comparison to other organic and inorganic pools, fish activities can promote and accelerate the transport of phosphorus from sediment to overlying water, particularly by feeding activity of benthic fishes and release of soluble phosphorus with urine and feces. When external loadings are small, as in dry summer period, these internal phosphorus inputs can be important to algae and other microbiota (Brabrand et al. 1990). Similarly in streams, fish derived phosphorus can be moved upstream for large distances by migrating fishes and subsequently released for use by algae, microbiota, and macroinvertebrates (Schuldt and Hershey 1995).

3. SUMMARY OF OTHER JURISDICTIONAL METHODS AND RATIONALE FOR THE CURRENT APPROACH

3.1 Introduction

Canadian Water Quality Guidelines are developed co-operatively by federal, provincial, and territorial governments under the auspices of the CCME. They are derived according to a nationally-endorsed, scientific protocol (CCME 1991). In deriving guidelines for aquatic life, all components of the aquatic ecosystem are considered if the data are available. The national protocol is consistent with that of the International Joint Commission Water Quality Board (1975) and the Ontario Ministry of the Environment (OMOE 1979; 1992). The protocol states that the Guidelines "are set at such values as to protect all forms of aquatic life and all aspects of the aquatic life cycles." The goal is to protect all life stages during an indefinite exposure to water. Whether this can be realized is a water management issue, and does not affect the guideline derivation procedure. Where data are available but limited, interim guidelines are deemed preferable to no guidelines.

Ambient water quality guidelines developed for the protection of aquatic life provide a science-based benchmark for a nationally consistent level of protection in Canada; these guidelines are not restricted to a particular (biotic) species. Toxicological data from fish, amphibian, invertebrate, and plant studies are used in the derivation of the guidelines. For full guidelines, at least three studies on three or more freshwater fish species resident to North America, including at least one cold water species (e.g., trout) and one warm water species (e.g., fathead minnow) need to be considered; of these, two or more should be chronic, partial or full life-cycle studies. A minimum of chronic, partial, or full life-cycle studies on at least two invertebrate species from different classes are required, one of which must be a planktonic species resident to North America (e.g., *Daphnia*). Only one North American freshwater vascular plant or algal species need be considered. The lower number of required plant studies was deemed necessary because fewer studies on plants have been conducted. For interim guidelines, a minimum of two acute and/or chronic studies on at least two fish species (including one cold water species), and at least two acute and/or chronic invertebrate studies on two or more species from different classes (including a planktonic species) need be considered.

The CWQG protocol is intended to deal specifically with toxic substances and provides numerical limits and narrative statements based on the most current, scientifically defensible toxicological data. Phosphorus does not fit this protocol as it is non-toxic to aquatic organisms at levels and forms present in the environment; however, secondary effects, such as eutrophication are a serious concern. Setting phosphorus guidelines by the CWQG protocol is difficult due to the subjective nature of what constitutes impairment. Some of the effects of phosphorus are aesthetic and, therefore, its management requires consideration of societal values.

The first response of an aquatic system to phosphorus additions is increased plant and algal productivity and biomass, which may be beneficial in some cases. However, beyond a certain point, further phosphorus additions to phosphorus limited systems cause undesirable changes in aquatic plant abundance and composition. Excessive aquatic plant growth, in turn, can cause oxygen depletion and fish kills. When the excessive plant growth includes certain species of cyanobacteria, toxins may be produced, causing increased risk to livestock and human health.

Factors such as DOC and the biotic community may modify the detrimental effects of phosphorus on the environment and because some of the effects of phosphorus are aesthetic, it is more difficult to derive ‘safe’ levels of phosphorus than it is for toxic pollutants. Approaches used by other jurisdictions for setting phosphorus guidelines, including provincial/ state, national, and international jurisdictions, were considered for possible incorporation into Canada’s phosphorus guidelines.

3.2 Canadian Provincial Water Quality Guidelines

3.2.1 Alberta

The Alberta surface water quality guidelines are numerical concentrations and narrative statements recommended to support and maintain a designated water use, and are based on a protocol document which draws on Alberta, USEPA, and CCME guidelines and criteria.

Although phosphorus is not directly toxic to freshwater aquatic life, values are included due to their broader influence on conditions that affect aquatic life. A value of 50 $\mu g \cdot L^{-1}$ TP has been derived based on the Alberta Surface Water Quality Objectives of 1993 (AEP 1993), which can be applied as chronic guidelines.

3.2.2 British Columbia

British Columbia’s environmental quality guidelines provide maximum and/or minimum values for physical, chemical, and biological characteristics of water, sediment or biota that are applicable province-wide, and which must not be exceeded to prevent specified detrimental effects from occurring to a water use, including aquatic life (Nordin 1985). In setting the BC guidelines, guidelines/criteria/objectives from the literature and other jurisdictions were evaluated, and either adopted for use in BC or developed from this evaluation. During the development process, it was recognized that guidelines from the literature do not necessarily adequately reflect the effects of contaminants in all Canadian water bodies.

A maximum biomass of 100 $mg \cdot m^{-2}$ chlorophyll *a* was used for the protection of aquatic life in streams. This criterion was derived primarily to protect fish habitat and their food source. In the case of lakes, a range of TP concentrations (5–15 $\mu g \cdot L^{-1}$) was suggested for the criterion as it was not possible to specify a single phosphorus concentration to achieve protection of aquatic life in lakes. The criterion is limited in application to include only lakes where the management of salmonid species of fish is of major importance.

For the aesthetics and recreation purposes, *a* value of 50 $mg \cdot m^{-2}$ chlorophyll *a* value is used in BC.

3.2.3 Manitoba

A variety of scientific tools and management strategies are used to protect, maintain, and rehabilitate water quality. The water quality objectives in Manitoba apply to a wide range of water-related media, including both ground and surface waters, lake and river bottom sediments, and fish tissues (Manitoba Conservation 2000). A three-tiered approach has been proposed to consolidate and harmonize Manitoba's standards, objectives and guidelines with those developed through other national efforts. Tier I provides Water Quality Standards (although not for phosphorus), and includes Canada-wide standards presently being developed by the CCME.

Water Quality Objectives are outlined in Tier II, and are limited to a short list of materials that are common pollutants in Manitoba. Most Tier II objectives are based on the principles advanced by the USEPA (EPA 1993), namely that healthy aquatic ecosystems can tolerate some stress and recover. The objectives provide protection from unacceptable impacts to all but a small percent of genera (5%). Exceptions are provided for important ecological, recreational, and commercial species, and endangered or rare species.

Tier III describes water Quality Guidelines for numerous materials developed by the CCME. Narrative guidelines are still retained for nutrients such as nitrogen and phosphorus until a more appropriate water quality objectives for nitrogen and phosphorus can be developed through the on-going Manitoba Nutrient Management Strategy. The Nutrient Management Strategy for Manitoba involves four main tasks. The first three tasks are largely technical or scientific and involve assembling and assessing the necessary information, proposing regional objectives for rivers and streams, and proposing nutrient objectives for Lake Winnipeg and other lakes where possible. The fourth task involves development of an implementation plan, particularly if nutrient reductions are required. It is expected that the first three tasks will be completed by late 2003 or early 2004. The implementation plan will be developed once the first three tasks have been completed. Two major reports have recently been released as part of this Nutrient Management Strategy. The first assessed the long-term data set in Manitoba streams for trends in N and P concentrations over the last three decades (Jones and Armstrong 2001) and the second estimated loadings of N and P to major water courses in Manitoba (Bourne et al. 2002).

It is generally recognized that the narrative guidelines for phosphorus are not likely to apply to many streams in the Canadian prairie region because other factors such as turbidity, stream velocity, nitrogen, and other conditions most often limit algal growth. Also, relatively high phosphorus levels, in excess of the narrative guidelines, may arise naturally from the rich prairie soils. Narrative biological guidelines are also proposed in Tier III to ensure the protection of ecosystem structure and function. These would augment the comprehensive chemical-specific guidelines.

In the Tier III narrative Water Quality Guidelines, “biological communities should not be altered beyond that which naturally exist.” Nitrogen, phosphorus, carbon, and contributing trace elements should be limited to the extent necessary to prevent nuisance growth and reproduction of aquatic rooted, attached and floating plants, fungi, or bacteria, or to otherwise render the water unsuitable for other beneficial uses. For general guidance, unless it can be demonstrated that TP is not a limiting factor, TP should not exceed 0.025 $mg \cdot L^{-1}$ in any reservoir, lake, or pond, or in a tributary at the point where it enters such waterbodies. In streams, TP should not exceed 0.05 $mg \cdot L^{-1}$.

3.2.4 Saskatchewan

Saskatchewan’s narrative guideline for nutrients require that nitrogen or phosphorus or other nutrient concentrations should not be altered from natural levels by discharges of effluents such that nuisance growths of algae or aquatic weeds result (SERM 1995). Also, nutrient levels should provide a minimum degree of protection of all beneficial uses. These general objectives are applicable to all portions of a receiving water body, except portions designated as an effluent mixing zone.

3.2.5 Ontario

The existing provincial water quality objectives for TP were developed in 1979 and drew on a trophic state classification scheme of Dillon and Rigler (1975) to protect against aesthetic deterioration and nuisance concentrations of algae in lakes and excessive plant growth in rivers and streams. The 1979 provincial water quality objectives (OMOE 1979) characterized lakes of low and moderate productivity by the use of two levels of water quality:

i. To avoid nuisance concentrations of algae in lakes, average TP concentrations of 20 $\mu g \cdot L^{-1}$ should not be exceeded;

ii. For naturally low phosphorus lakes, 10 $\mu g \cdot L^{-1}$ is used, which ensures a high level of protection against aesthetic deterioration.

Excessive plant growth in rivers and streams should be eliminated at a TP concentration below 30 $\mu g \cdot L^{-1}$.

3.2.6 Québec

The water quality criteria defined by the Ministry of the Environment of Québec for phosphorus is based on assessments performed by the department and on Ontario Ministry of the Environment (OMOEE 1994). A maximal value of 30 $\mu g \cdot L^{-1}$ was defined for rivers and streams to prevent excessive algal growth for the protection of aquatic life, recreational use, and aesthetic. A maximal value of 20 $\mu g \cdot L^{-1}$ was defined for rivers or streams ending in lakes with no history of environmental disturbance. A criterion for lakes is a maximum 50% increase from the reference (natural) condition, without exceeding: i) a concentration of 10 $\mu g \cdot L^{-1}$ for lakes where natural concentration is lower than 10 $\mu g \cdot L^{-1}$, and ii) a concentration of 20 $\mu g \cdot L^{-1}$ for lakes where natural concentration is lower than 20 $\mu g \cdot L^{-1}$ (MENV 2001).

3.3 Water Quality Guidelines from International Jurisdictions

A review of the methods adopted by other jurisdictions (summary given in Table 3.1) for the management of phosphorus indicates that most jurisdictions attempt to deal with regional variance in natural water quality in various ways and try to associate different criteria for different uses.

3.3.1 USA

National standards for phosphorus in surface waters in the USA are available for jurisdictions to implement (EPA 2000b; 2000c; 2001). The USEPA provides guidance to states and tribes developing their own guidelines. As a result of a National Nutrient Assessment Workshop (EPA 1995), efforts were made to address nutrient and over-enrichment in a peer-reviewed national nutrient criteria strategy. Major elements of this strategy include:

i. The use of a regional and waterbody-type approach for the development of nutrient water quality criteria;

ii. The development of waterbody-type technical guidance documents that will serve as ‘user manuals’ for assessing trophic state and developing region-specific nutrient criteria to control over enrichment;

iii. The development by the USEPA of nutrient water quality criteria guidance in the form of numerical regional target ranges, which the USEPA expects states and tribes to use in implementing State management programs; and

iv. The monitoring and evaluation of the effectiveness of nutrient management programs as they are implemented.

The state of Maine is unlike other jurisdictions in that their objectives have very specific implementation details, and like the Lakeshore Capacity Model (LCM) developed for Precambrian Shield lakes in Ontario, tries to address shoreline development as the most important stress on water quality. Their allowable increases in phosphorus are very restrictive, taking a non-degradation approach.

A summary of the different criteria set for some of the states in the USA are provided in Table 3.1.

3.3.2 European Economic Community

The European Economic Community (EEC) derives environmental quality standards at the community level for a variety of pollutants. Other European Union (EU) directives set goals for the protection of the environment, in the form of standards for specific uses. Thus, the surface waters, drinking water, bathing water, and freshwater fish directives contain standards relating to nitrogen and phosphorus for different water uses. No standards have been developed for phosphorus for the protection of bathing waters or fisheries. A directive to reduce phosphate inputs was considered but the controls exerted via the nitrate directive are currently considered adequate (Crouzet et al. 1999) to protect against aesthetic degradation, prolific algal and plant growth, and overall loss of amenities.

Several other international agreements have addressed nutrient enrichment. The UN Convention on the Protection and Use of Transboundary Watercourses and International Lakes (UN 1992) was signed by 25 countries including all EEC countries (except Iceland, Ireland, and Liechtenstein). The Convention requires signatories to prevent, control and reduce pollution of water causing, or likely to cause, transboundary impact with the aim of ecologically sound and rational water resources and environmental protection. In addition to goals set internationally, some countries, or parts of countries, have adopted their own national targets, particularly in countries were eutrophication problems are at their greatest. For example, one of the central objectives of the Dutch governmental policy on water quality is to protect and develop healthy water systems that guarantee sustainable use (Table 3.1). In a study conducted by Peeters and Gardeniers (1998), the general standard of 15 $\mu g \cdot L^{-1}$ set for TP was not considered appropriate for the protection of aquatic life.

3.3.3 Australia and New Zealand

Australia and New Zealand are presently revising their national water quality management strategy (NWQMS 1999). The new guidelines take a risk-based approach, adopt risk assessment methods, and take into account the variability and complexity of aquatic systems. The guidelines still provide default water quality guidelines, referred to as ‘trigger values’. Physical and chemical water quality and aquatic biota inherently vary within and amongst ecosystem types and so the preferred method for determining trigger values is to use reference sites with the highest available quality target levels or condition. This provides a guideline that is relevant to the ecosystem type and locality. When these trigger values are exceeded, there is a risk of an impact occurring. Trigger values can be modified into regional, local, or site-specific guidelines.

Table 3.1 Summary of phosphorus (P) criteria and values used in other jurisdictions and countries.

Jurisdiction	Comments	TP Criteria
Canada		
Alberta	• Guideline (chronic) is for the protection of freshwater aquatic life	50 $\mu g \cdot L^{-1}$
British Columbia	• Criteria vary according to type of aquatic system and intended use • Chlorophyll *a* used to monitor streams • TP range for lakes as criterion can be used as basis for site-specific WQOs	Lakes: 5-15 $\mu g \cdot L^{-1}$ Rivers: 100 $mg \cdot m^{-2}$ Chl *a* for aquatic life; 50 $mg \cdot m^{-2}$ Chl *a* for aesthetics and recreation
Manitoba	• Tiered approach: Tier I = standards, Tier II = objectives, Tier III = guidelines • Narrative guidelines retained for nutrients • Recognized that guidelines for P likely do not apply to many streams since other factors such as turbidity, stream velocity, nitrogen, and other conditions most often limit algal growth • Reservoirs, lakes, ponds, and tributaries at the point where they enter such waters are all considered together under one criteria	Lakes: 25 $\mu g \cdot L^{-1}$ Rivers: 50 $\mu g \cdot L^{-1}$
Ontario	• Aimed to protect against aesthetic deterioration and nuisance algal and plant growth • Characterizes lakes of low and moderate productivity	Lakes: 10, 20 $\mu g \cdot L^{-1}$ Rivers: 30 $\mu g \cdot L^{-1}$
Québec	• Uses Ontario's numbers • Values are for the protection of aquatic life, recreational use, and aesthetic	Lakes: 50% maximum increase from the natural concentration: i) without exceeding 10 $\mu g \cdot L^{-1}$ in lakes with natural concentration of 10 $\mu g \cdot L^{-1}$; or ii) without exceeding 20 $\mu g \cdot L^{-1}$ in lakes with natural concentration of 20 $\mu g \cdot L^{-1}$; Rivers: 30 $\mu g \cdot L^{-1}$
USA		
USEPA	• No national standard for P in fresh surface water • Provides guidance to states developing own guidelines • Presently only national water quality criteria for nitrate-N and P (1976) = *protection of marine and estuarine waters (estimate to protect against toxic effects of bioaccumulation of *elemental P*) • All states and Tribes expected to adopt and implement numerical nutrient criteria into their water quality standards by December 31, 2003.	0.1 $\mu g \cdot L^{-1}$ P*

Table 3.1 (continued)

Jurisdiction	Comments	TP Criteria
Minnesota	• Uses ecoregion approach • Eutrophication standards vary with region (i.e., the natural water quality) • Criteria developed to meet specific uses • Not legally binding - used for setting goals and priorities • P related to summer Chl *a* concentrations • Presently only lake criteria available - working on rivers and streams	Lakes: 15-90 $\mu g{\cdot}L^{-1}$ according to ecoregions and specific use
Vermont	• Site-specific management • Objectives have been developed for 13 basins • Now considering developing standards for all lakes in the state	Lakes: 10-25 $\mu g{\cdot}L^{-1}$
Wisconsin	• Ecoregional approach • 3 P-regions established, characterized by statistically different water quality • 'Best professional judgment' used to establish baseline water quality of various water bodies in each of the regions • Separate standards for impoundments and lakes	Lakes: 5-25 $\mu g{\cdot}L^{-1}$ according to region and type
Other states	• Some (e.g., Michigan and Pennsylvania) have not developed surface water criteria, but rely on effluent concentrations, discharge loadings and best management practices	
EEC	• Lays down environmental standards at a community level for certain pollutants • Other environmental standards (e.g., P) laid down by minister states • Aim to protect against prolific plant growth and loss of aesthetic value	
Netherlands	• To protect and develop healthy water systems that guarantee sustainable use	One value: 0.15 $mg{\cdot}L^{-1}$
Other EEC member states	• Directives adopted to reduce P inputs rather than the development of water quality standards for P	(*Nitrate Directive currently considered adequate*)
Australia & NZ	• Presently revising water quality standards • Risk-based approach • Provide WQG = 'trigger values' for freshwater lakes, upland rivers and lowland lakes	FW Lakes: 50 $\mu g{\cdot}L^{-1}$ Upland R: 35 $\mu g{\cdot}L^{-1}$ Lowland R: 37 $\mu g{\cdot}L^{-1}$

4. THE DEVELOPMENT OF GUIDELINES FOR PHOSPHORUS

4.1 Introduction

The protocol for developing Canadian water quality guidelines is intended to deal specifically with toxic substances and provide numerical limits or narrative statements based on the most current, scientifically defensible toxicological data available. Phosphorus does not fit this protocol, as it is essentially non-toxic to aquatic organisms. Elevated loads of phosphorus, in most freshwater systems, will increase plant growth, as well as affect plant species composition before any other part of the aquatic community. Many of the effects of phosphorus are aesthetic, and therefore setting phosphorus guidelines by the CWQG protocol is difficult due to the subjective nature of what constitutes impairment. As such, it is more difficult to derive 'safe' levels of phosphorus than it is for toxic pollutants; what is considered nuisance plant growth to some may be considered desirable to others, and its management requires consideration of societal values. Despite of this subjectivity, increase in phosphorus concentrations can have dramatic effects on aquatic food webs and these affects are much less subjective.

A variety of methods are used by different jurisdictions, both nationally and internationally, to address regional variation in natural or baseline water quality. Most of these try to associate different criteria for different uses. Jurisdictions with similar water quality to Ontario seem to have developed similar objectives, and often use a series of regional- or user-specific objectives. Other areas have adopted new approaches to those traditionally used. For example, Australia and New Zealand are presently revising their national water quality management strategy (NWQMS 1999) to include risk-based approach that considers the variability and complexity of aquatic systems. As a result, they provide default water quality guidelines, referred to as 'trigger values'.

4.2 Derivation

4.2.1 Rationale for a new approach

At international, national and provincial/state or regional levels, nutrient enrichment has been tackled through a wide range of approaches. Setting guidelines for nutrients is difficult based on the subjective nature of what constitutes impairment (i.e., filamentous green algae have a less desirable appearance than brown-coloured diatoms, even when the biomass of the two is similar). Moreover, a large amount of variation exists in benthic chlorophyll levels that are unrelated to nutrient levels. Many countries, particularly in the EEC, have adopted management practices to reduce phosphorus loading into aquatic systems and have set goals for desirable phosphorus reduction using 'Best Available Technology', rather than setting guidelines for phosphorus. Considerable success has been achieved in reducing nutrient inputs, particularly for phosphorus (e.g., Crouzet et al. 1999).

Basing criteria on the best scientific data will minimize any conflict surrounding desirable phosphorus levels and aesthetic impairment and maximize the potential benefits to controlling nutrients. Because there is a level of subjectivity concerning phosphorus criteria and desirable changes to the aquatic environment, there is a need to develop guidelines that are scientifically defensible and can be incorporated into management practices. There is also a need to define/establish baseline or reference conditions to which comparisons can be made when setting achievable criteria.

Based on a review of the approaches used by other jurisdictions for setting phosphorus guidelines, a framework-based approach is proposed. The proposed approach accommodates the non-toxic endpoints associated with phosphorus loading and can be incorporated into management strategies.

4.2.2 National guidance framework for the management of phosphorus

Because defining an adverse effect due to phosphorus can be subjective and phosphorus is not directly toxic, a framework-based approach is proposed (Figure 4.1). The framework, that includes elements of the Australian model provides a tiered approach where water bodies are marked for further assessment by comparing their trophic status to predefined ‘trigger ranges’. The trigger ranges are based on a range of phosphorus concentrations in water that define the reference trophic status for a site. Using such a scheme, sites with similar characteristics are classified together irrespective of whether they might possess these features naturally or as a result of human influence. Since the reference condition for the water body in question is defined at the onset of the framework, this problem is readily overcome; predefined states, whether determined through hindcasting or by using best available data, are always used to set the trigger range, thus the effects of human impact are negated.

The actual phosphorus concentration of the system under investigation is compared to predefined trigger range, once it has been identified. Schemes that classify by a single factor or group of auto-correlated factors have severe limitations (Moss et al. 1997). Such an approach would be compounded by the greater complexity of the systems, as they are not controlled by a single over-riding physical factor. It would therefore be desirable to develop a scheme which uses an array of variables and which is open-ended to allow the introduction of new variables, as our knowledge increases. Thus, should these trigger ranges be exceeded, or are likely to be exceeded, further assessment is required. A number of different techniques, including the development of a multivariate approach, the development of a water quality index (WQI), or the use of existing predictive models can be adopted to assess and set management goals for phosphorus or other parameters indicative of enrichment. A schematic diagram of the process is outlined in Figure 4.1. When the predictive phosphorus values or other relevant parameters at the site do not exceed 50% of the baseline or reference condition, and the phosphorus values do not exceed the trigger range, then no further action is required.

The different methods of assessment and defining phosphorus are not exclusive, and combinations of these methods can be adopted. For example, environmental variables can be compared to baseline data (hindcast or best available) and expressed as a percent change (Johnes et al. 1994), analyzed directly through multivariate analysis

(Reynoldson and Day 1998), or through a suite of indices (Kilgour et al. 1998; OMOE 1999a). However, the percent data could either be summarized as a single value as employed by Johnes et al. (1994), or if preferred, analyzed multivariately. Adaptations of existing models could also be considered. The management framework outlined above is a feedback model and incorporates continuous monitoring. When assessment suggests a likelihood that phosphorus will result in an undesirable change in the local system, a management decision must be made.

The assessment methods outlined in this document should be viewed with the caveat that many of them were developed for specific water types, or for specific areas with underlying topographic and geological assumptions (e.g., Lakeshore Capacity Model). Although many of these methods can be adapted to the specific user's situation, care must be taken in selecting a method that is both technically feasible and realistic to the users needs. The water type should also be considered when selecting an approach, and the water quality parameter to assess enrichment.

A detailed overview of each stage in the framework is provided in the following chapters. At its onset, it is crucial to define the management goals or objectives (e.g., enhance, protect, or restore) for the water system. These objectives are important because they will guide management decisions made later.

4.3 Use of the Guidance Framework

4.3.1 Set ecosystem goals and objectives

As a first step, it is crucial to set ecosystem goals and objectives (e.g., enhance, protect, or restore). The objective can be set for a healthy aquatic ecosystem, with the goals being unimpaired human uses, and a diverse and functioning aquatic ecosystem. By setting the diverse aquatic ecosystem as a goal, a desire to prevent the loss of species is established. Similarly, the desire for a functioning aquatic ecosystem recognizes that ecosystems do things that are both inherently of value and of value for human uses and desires. These objectives are important because they will guide management decisions made later.

4.3.2 Define the reference / baseline condition

Physical, chemical, and biological variables inherently vary within and among ecosystem types. To assess whether a body of water is impaired, there must be some basis for comparison. The use of regional reference sites provides an objective way to define limits of acceptable and unacceptable conditions (Hughes 1995). The preferred method for developing phosphorus criteria is to use high quality reference sites to determine target levels or trigger values. This would provide a guideline that is relevant to the ecosystem type and locality. Attempting to identify 'pristine' condition can be difficult and, particularly in heavily populated areas, impractical. In these situations other methods that are presented here can be applied.

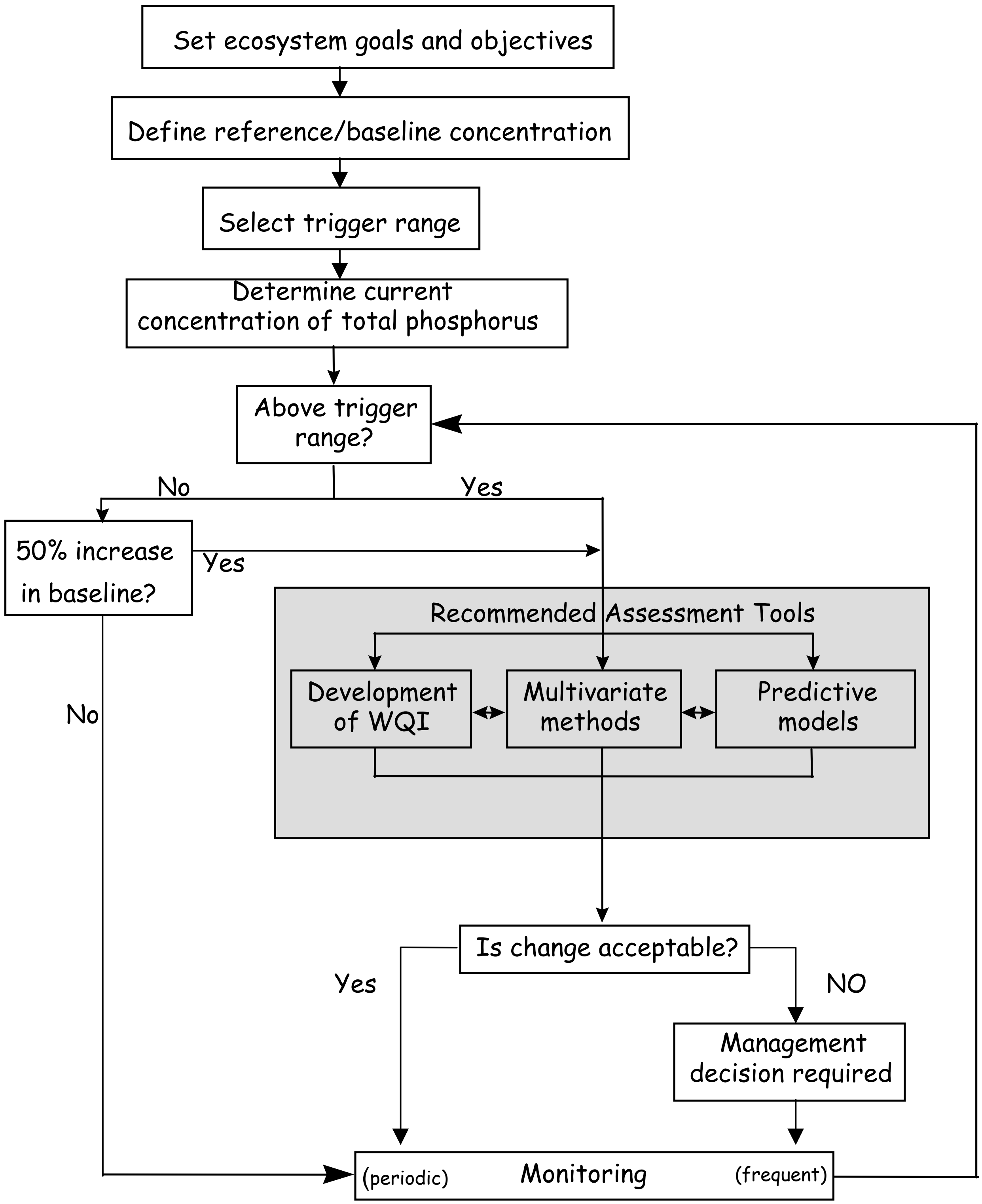

Figure 4.1 Canadian Guidance Framework for the management of phosphorus in freshwater systems.

4.3.3 Ecoregions

Ecoregions indicate how waters can be classified based on parameters other than nutrients and are critical to our understanding of regional reference conditions. It is impractical to define water quality, and develop objectives on a lake-by-lake or river-by-river basis, there is a need to subdivide water bodies into regions (or zones) based on characteristics such as topography, geology, climate, chemistry, etc. The topography of the catchment basin influences the extent of erosion and subsequent export of nutrients. Erosion is also influenced markedly by the type of vegetation and land use. Geological characteristics can be used in defining a water body as it influences physical, chemical and, in turn, biological characteristics of surface water.

Ecozones and ecoregions provide a means of classifying ecologically distinct areas. Each zone or region can be viewed as a discrete system made up of areas of similar geographical landform, soil, vegetation, climate, wildlife, water, etc. These zones are based on criteria that are considered not to change perceptually over time. The use of ecoregions can greatly improve predictability of nutrient enrichment effects. They can help differentiate between natural and anthropogenic contributions to nutrient enrichment, reduce variability in trigger ranges within a class and among classes, and contribute to improved assessment and development of trigger ranges.

In Canada, 15 ecozones have been identified largely based on generalized ecological units that are characterized by interactive and adjusting abiotic and biotic factors. Figure 4.2 illustrates these 15 ecozones.

Chambers et al. (2001) provide an overview of enrichment in the Canadian ecozones; for example:

- The five ecozones in British Columbia (Pacific Maritime, Montane Cordillera, Boreal Cordillera, Taiga Plains, and Boreal Plains) were overall deemed to have an excellent water quality with respect to nutrients for both fresh and marine waters, although the Fraser River shows signs of nutrient enrichment in its lower reaches.

- In the Prairies ecozone of southern Alberta, Saskatchewan and Manitoba, the quality of the mountain-fed rivers was generally good except downstream of cities and in areas of intensive agriculture. Rivers arising in the Prairies ecozone were usually eutrophic as a result of natural and agricultural impacts. Lakes in the Prairies ecozone tended to have nitrogen and phosphorus concentrations that were often higher than the interim provincial water quality guidelines for the protection of aquatic life in streams of the intensively agricultural cultivated areas.

- Rivers and lakes in the Boreal Shield and Boreal Plains ecozones (which account for about 29% of Canada's freshwater surface area) were overall of a very good water quality. Nutrient concentrations in the eastern Boreal Shield are naturally low, as the underlying rock throughout most of the ecozone is granite and provides little nutrients to the overlying soils and water. The Boreal Plains are naturally somewhat higher in nutrients due to thick glacial soils underlying the rivers and lakes.

- In the Great Lakes and St. Lawrence ecozones of southern Ontario and Québec, agricultural runoff, municipal sewage and industrial waste water have had a substantial impact on the loading of nitrogen and phosphorus to the rivers and lakes. Recent water quality surveys in southern Québec have shown that TP levels in most of the rivers in the lowlands were approximately 30 $\mu g \cdot L^{-1}$. In the case of the Great Lakes, 8 of the 12 areas of concern in the Canadian portion of the Great Lakes basin are impaired due to excess nutrient concentrations or undesirable algal growth.

- The Atlantic ecozones, including the Marine Atlantic and Northwest Atlantic ecozones (encompassing the provinces of Nova Scotia, New Brunswick, and Prince Edward Island), have been subject to intensive alterations for almost 400 years of settlement. Several of the large cities have discharged untreated sewage to the coastal waters for over 250 years. The inland waters throughout the region have been affected by anthropogenic inputs and show varying degrees of eutrophication. In the Atlantic region the complex nature of the coastal environments (e.g., high flushing rates and input of nitrogen rich deep oceanic waters) make it difficult to quantify the impacts of anthropogenic nutrient and develop management plans.

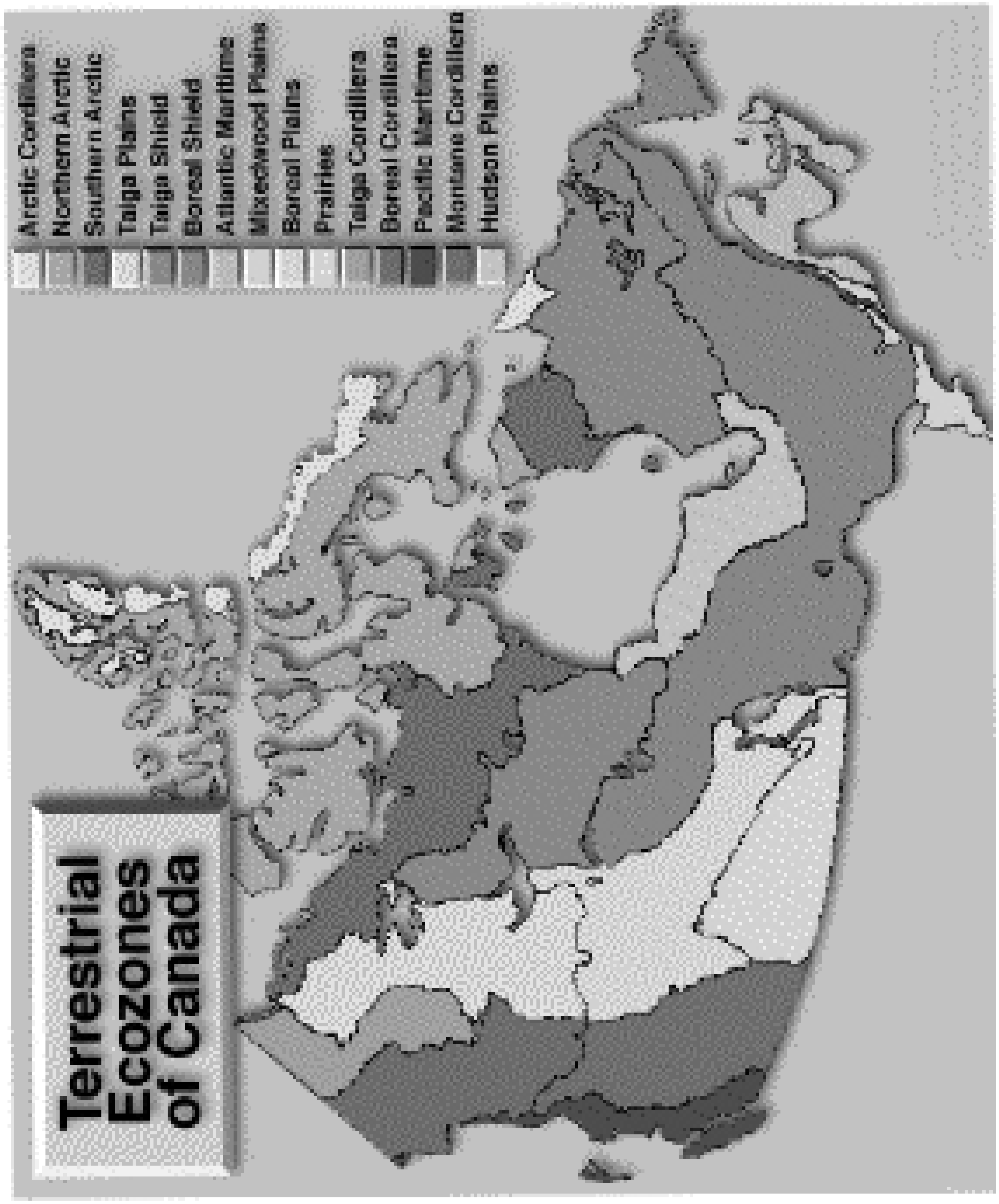

Figure 4.2 Map of the 15 ecozones of Canada (source: Government web site; www1.ec.gc.ca/~ecozones).

4.3.4 Approaches to establish reference conditions

To define a realistic reference condition or baseline state, there are a number of methods currently in use. These include the use of historical data, hindcasting, and the reference condition approach.

4.3.4.1 Historical records

Historic records exist from the earlier part of the last century for selected North American water bodies. If such records are available, these data could be adopted to define the reference or baseline conditions. Measured historical records, however, are often lacking, and the accuracy of early TP data can often be questionable, thus severely limiting the use of historical records.

4.3.4.2 Hindcasting

4.3.4.2.1 Paleolimnology

Reference/ baseline phosphorus concentrations can be defined by hindcasting phosphorus levels using paleolimnological methods (Dixit et al. 1999; Hall and Smol 1999). Over the last decade, there have been numerous applications of paleolimnological methods in addressing lake eutrophication. In brief, the method involves the investigation of physical, chemical, and biological remains in aquatic sediments. Siliceous frustules of the diatom algae (division Bacillariophyta) are widely used bioindicators in addressing nutrient related issues because they are abundant and diverse in aquatic environments (e.g., lakes, rivers, wetlands, and estuaries), they have specific environmental optima and tolerances for nutrients (P, N, and Si) and their diagnostic siliceous remains are well preserved in sedimentary records (Dixit et al. 1992).

The ability to infer eutrophication trends using sedimentary diatom assemblages has increased greatly within the last decade or so, mainly due to advances in methodology and statistical techniques (Hall et al. 1997, 1999). Using the weighted-averaging regression and calibration approach (Line et al. 1994), diatom-based inference models have been developed to infer phosphorus concentrations for many regions of North America, including western and eastern Canada (e.g., Reavie et al. 1995; Hall and Smol 1996) and northeastern and midwestern USA (Fritz et al. 1993; Dixit et al. 1999). These inference models have been successfully applied to identify reference or pre-impact conditions and the timing, rate, extent, and probable causes of eutrophication in both deep-water and near-shore habitats (e.g., Hall et al. 1997; Dixit et al. 2000; 2001). This approach has also helped to identify naturally productive lakes in British Columbia (Reavie et al. 1995) and Saskatchewan (Hall et al. 1999) as such data have important management implications, as mitigation efforts are unlikely to restore these sites to oligotrophic conditions.

The USEPA’s Environmental Monitoring and Assessment Program (EMAP) in the northeastern US provides a clear example of how this approach can be used to assess phosphorus changes on a broader, regional scale, in addition to simply provide assessment on lake by lake basis (Dixit et al. 1999). In this study, changes in phosphorus concentrations were assessed in 159 natural lakes and 79 reservoirs and because these sites were randomly selected, the data were extrapolated to a target population of 4,638 lakes and 4,906 reservoirs. The population data clearly identified that anthropogenic activities have significantly affected the water quality in the northeastern US. A small population of lakes in the region was naturally productive and phosphorus concentrations have increased in lakes and reservoirs that currently have high phosphorus concentrations. Moreover, the extent of cultural impact has been quite variable among the Adirondacks, New England Uplands, and Coastal Lowlands/Plateau ecoregions and that eutrophication has been mainly centered in the Coastal Lowlands/Plateau ecoregion.

4.3.4.2.2 Mass balance models

Empirical or mass balance nutrient models are increasingly used to quantify lake responses to changes in land use and to develop tools for shoreline development. For example, Canadian Lakeshore Capacity Model (LCM) has been in use in Ontario for over two decades (Dillon et al. 1986). The model was originally conceptualized to model past, present and future trophic conditions (e.g., phosphorus, chlorophyll *a*, Secchi depth, and oxygen) so that the improvements can be made to existing water quality objectives. It has been adopted by resource managers in the Ontario Ministry of the Environment's (OMOE) regional offices, other government agencies in Canada and the USA, and by the scientific and consulting community (OMOE 1997).

The LCM comprises a series of linked sub-models that use various catchment characteristics to estimate phosphorus loading, which is combined with lake characteristics (morphology, hydrology) in a mass balance model. The modelling tool permits the user to: (i) assess the contribution of existing shoreline development scenarios (e.g., number of cottages, amount of agricultural land in catchment) to trophic status, including determining the reference/ baseline water quality; (ii) evaluate the potential water quality implications of proposed new development, including the effects on lake trout habitat; and, (iii) establish a permissible phosphorus concentration and calculate the allowable development that will not result in adverse trophic conditions.

4.3.4.3 Use of pristine sites

Another approach in defining the reference, or baseline state is to identify a set of relatively pristine water bodies that have been unaffected by anthropogenic activities and use these to establish baseline conditions. However, few pristine sites exist, and trying to identify suitable sites for different regions of Canada would be difficult if not impossible.

Reynoldson and co-workers (Reynoldson et al. 1995; 1997; Reynoldson and Day 1998) developed a database to identify a series of reference conditions for the Great Lakes. The Reference Condition Approach (RCA) identifies the 'best available' condition by selecting relatively clean sites of similar chemical, physical, and biological parameters and grouping these water bodies into one reference condition (i.e., mean phosphorus value), to which test sites can be compared. This type of approach has been widely adopted throughout the world as it provides a more realistic baseline to which comparisons can be made.

4.3.4.4 Use of the twenty fifth percentile

The USEPA (EPA 2000a) outlines two methods to define the regional reference condition of lakes and reservoirs. Both approaches select an optimal reference condition value from the distribution of an available set of lake or river data from a given physical class of lakes or rivers. The USEPA uses at least 4 variables: TP, TN, chlorophyll *a*, and Secchi depth. Other indicators include DO and macrophyte growth or speciation; these are deemed important, but the first 4 are paramount. Because of the diverse geographic and climatic conditions, a distribution of lakes may exhibit a range of nutrient conditions. The USEPA therefore considered it inappropriate to set a single national nutrient criteria for lakes and reservoirs. Instead, it was deemed necessary to set nutrient criteria at the state, regional, or individual water body levels; this subdivision also applies to rivers and streams (EPA 2000b).

The sites can be divided into similar groups based on their physical characteristics within a proximal geographic area. Within each established group (or ecoregion), sites having the least land development or other human impact can be identified as reference sites for measuring relatively undisturbed nutrient conditions for each region. This reference condition information can then be used to define a benchmark against which other similar sites can be compared.

In establishing the reference condition, the USEPA recommends two approaches. One approach is to select a percentile from the distribution of measured variables (e.g., TP, TN, chlorophyll *a*, and Secchi depth) of known reference lakes (e.g., highest quality or least impacted lakes of that size class of all lakes in the region). Because these reference lakes are already acknowledged to be in an approximate ideal state, they recommend selecting the upper 25th percentile (i.e., a representation of the lesser nutrient quality of the distribution of the reference lakes) as the reference condition (Figure 4.3).

The second approach is to select a percentile for all lakes in the class or from a random sample distribution of all lakes in the class. In this case the lower 25th percentile can be selected (i.e., a representation of the better end of the range of all lakes) because the sample is expected to contain at least some degraded lakes if it is an entire population or a truly random selection (Figure 4.3). This option is most useful in regions where the number of true natural reference water bodies is very small such as in highly developed areas. The EPA has outlined similar approaches to define the reference condition when setting nutrient criteria for rivers and streams (EPA 2000b). It should be noted that the selection of upper or lower 25th percentile is based on the EPA's general recommendation and suggestion that the actual distribution of the observations and knowledge of the inherent regional water quality would also determine the selection of threshold point (EPA 2000a). For example (Figure 4.3), the upper 25th percentile produces a reference of 30 μg·L^{-1}and the lower 25th percentile produces a value of 35 μg·L^{-1}. Because of the small separation between these two distributions, the EPA suggested that either 30 μg·L^{-1}, 35 μg·L^{-1}, or the intermediate 33 μg·L^{-1} TP can be selected as a reference condition and recommend that the States and Tribes should apply both approaches and select the value that is most protective.

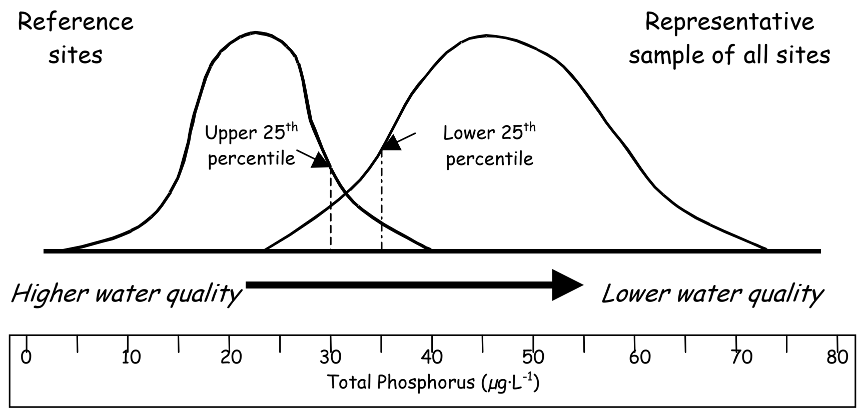

Figure 4.3 USEPA's proposed approaches for establishing a reference condition value for phosphorus (source: EPA 2000a).

4.3.4.5 Overall evaluation of reference condition methods

Among the methods outlined above, defining the best available condition is the more pertinent method to use for the derivation of guideline reference values for Canada. If data, specific to the water body in question, is unavailable it is advised that the USEPA or hindcasting approach be adopted. Data should be collected for sites throughout a specific area (or ecozone), and either the upper 25th percentile of the reference sites, or the lower 25th percentile of all of the sites be used to define the baseline condition. Before defining the baseline condition, the reference sites within the ecozone should be compared. If there are obvious differences in the reference values (identified, for example, through cluster analysis), the ecozone should be subdivided accordingly.

4.3.5 Selection of Trigger Ranges

4.3.5.1 Defining trophic status

Phosphorus is an important variable for the classification of trophic state as it is the nutrient, that is most likely to limit aquatic primary producers in fresh waters. Concentrations of total nutrients and suspended algal biomass are well correlated in lakes and reservoirs (Dillon and Rigler 1974; Jones and Bachmann, 1976; Carlson 1977). However, developing predictive relationships between nutrient and algal levels in fast-flowing streams and rivers is challenging because a major proportion of available nutrients remains in the water column and most chlorophyll *a* is in the benthos (Moss 1998), along with the confounding effects of variables such as turbidity and flow regime.

4.3.5.1.1 Defining trophic status of lakes and reservoirs

Trophic status refers to the availability of growth limiting nutrients (Smith et al. 1999). Trophic classifications for lakes have a long history and the use of trophic status categories, such as oligotrophic, mesotrophic, and eutrophic (or subdivisions thereof) (Table 4.1), have been widely adopted (Vollenweider and Kerekes 1982; Moss et al. 1997; Dodds et al. 1998; Smith et al. 1999; Wetzel 2001). Models established by Vollenweider (1968) and Dillon and Rigler (1974) provide a relationship between the phosphorus load or concentration and algal biomass. The boundaries between these trophic categories as defined by aquatic scientists are similar (Table 4.1), but not universal (e.g., Forsberg and Ryding 1980; Porcella et al. 1980; Vollenweider and Kerekes 1982).

Table 4.1 Trophic classifications of lakes, with their corresponding phosphorus and chlorophyll concentrations and transparency (Secchi depth) (sources: Wetzel 2001; Vollenweider and Kerekes 1982).

Trophic level	Total Phosphorus ($\mu g \cdot L^{-1}$)		Chlorophyll *a* ($\mu g \cdot L^{-1}$)		Secchi depth (m)	
	Wetzel (2001)	Vollenweider and Kerekes (1982)	Vollenweider and Kerekes (1982)		Vollenweider and Kerekes (1982)	
			Mean	Max	Mean	Max
Ultra-oligotrophic	< 5	< 4	< 1	< 2.5	> 12	> 6
Oligo-mesotrophic	5-10	4-10	< 2.5	< 8	> 6	> 3
Meso-eutrophic	10-30	10-35	2.5-8	8-25	6-3	3-1.5
Eutrophic	30-100	35-100	8-25	27-75	3-1.5	1.5-0.7
Hypereutrophic	> 100	> 100	> 25	> 75	< 1.5	< 0.7

4.3.5.1.2 Trophic status of rivers and streams

Stream ecosystems have traditionally been described on the basis of carbon sources (Dodds 1997) and position in the watershed (Vannote et al. 1980). The classification of ecosystems by an index of trophic state is common in aquatic sciences; the trophic classification scheme described above was developed for lakes and reservoirs, and has primarily been used to describe such water bodies. Streams are occasionally classified as eutrophic or oligotrophic, based on the above scheme (Hornberg et al. 1977; Kelly and Whitton 1995). The general trophic lake classification scheme has recently been extended to streams and rivers (Table 4.2, Dodds et al. 1998). Stream enrichment often leads to increases in algal biomass (Dodds et al. 1997), and thus, a trophic classification using both nutrients and algal biomass is a useful tool. However, it is important to point out that the correlation between TP and chlorophyll for rivers are distinctly weaker than the relationship for lakes (Wetzel 2001).

Table 4.2 Trophic classifications in streams, with their corresponding phosphorus and chlorophyll concentrations (source: Dodds et al. 1998).

Trophic status	Total phosphorus ($\mu g \cdot L^{-1}$)	Suspended Chlorophyll *a* ($mg \cdot m^{-2}$)	Benthic Chlorophyll *a* ($mg \cdot m^{-2}$)
Oligotrophic	< 25	< 10	< 20
Mesotrophic	25-75	10-30	20-70
Eutrophic	> 75	> 30	> 70

4.3.5.2 Selecting a trigger range

Australia, New Zealand (NWQMS 1999), and the USEPA (EPA 2000a) consider ecosystem classification in setting their nutrient guidelines. In the proposed Canadian framework, trigger ranges are based on the trophic classification of the baseline condition or reference sites. These are general characterisations that can also utilize ecoregional classification. At the modelling step, incorporation of local scales is further recommended.

A trigger range is a desired concentration range for phosphorus; if the upper limit of the range is exceeded, it indicates a potential environmental problem, and therefore, "triggers" further investigations. Natural physical and chemical water quality variables (e.g., salinity, pH, nutrients) inherently vary within and between ecosystem types, and so the preferred method for determining the trigger ranges is to use similar, high quality reference sites (defined in the previous chapter) to determine natural levels. These ranges are then categorized according to the trophic status (Vollenweider and Kerekes 1982; Dodds et al. 1998) of the reference lake or river (Table 4.3). This approach provides a trigger range that is relevant to the ecosystem type and locality. These phosphorus limits allow management organizations to define where their water bodies lie, and define a trigger range for that water body.

The majority of Canadian lakes fall into the category defined by the OECD (Vollenweider and Kerekes 1982) as meso-eutrophic; the classification of this range is broad (10–35 $\mu g \cdot L^{-1}$ TP) and large changes in macrophyte and periphyton community composition and biomass can be observed within this range. In setting the trigger ranges (Table 4.3), this category has been further divided into 'mesotrophic' (10–20 $\mu g \cdot L^{-1}$ TP) and 'meso-eutrophic' (20–35 $\mu g \cdot L^{-1}$ TP) to protect against such community changes.

Table 4.3 Trigger ranges based on trophic classification of lakes (Vollenweider and Kerekes 1982) and rivers (Dodds et al. 1998).

Trophic level	Trigger Ranges for Total phosphorus ($\mu g \cdot L^{-1}$)	
	Lakes	Rivers and streams
Ultra-oligotrophic	< 4	-
Oligotrophic	4-10	< 25
Mesotrophic	10-20	25-75
Meso-eutrophic	20-35	-
Eutrophic	35-100	> 75
Hypereutrophic	> 100	-

These trigger ranges are described for both lakes and rivers for each trophic status in Table 4.3. Phosphorus concentrations inherently vary spatially and temporally in all aquatic systems, and this natural variation is acknowledged in the ranges set for each of the trigger values. For example, areas where there are naturally low phosphorus levels (e.g., Boreal and Pre-Cambrian Shields) which are ultra-oligotrophic/ oligotrophic, trophic ranges are narrower (< 4 $\mu g \cdot L^{-1}$ or 4-10 $\mu g \cdot L^{-1}$) than more eutrophic systems (35-100 $\mu g \cdot L^{-1}$), which are able to sustain higher phosphorus increases before any observable effects occur. Typically, rivers can sustain higher loads of phosphorus than lakes without any observable changes in community composition and biomass (Smith et al. 1999), as phosphorus may be flushed from the system before it can be utilized. McGarrigle (1993) concluded from work carried out on Irish rivers, that if a mean annual DOP concentration of less than 47 $\mu g \cdot L^{-1}$ was maintained, nuisance growth of attached algae would be prevented and water quality suitable for salmonid fisheries would be preserved. Based on a comparative study of stream ecosystems, carried out by Dodds and co-workers (1997), it was suggested that a TP concentration of less than 30 $\mu g \cdot L^{-1}$ is necessary to keep the benthic algal biomass below nuisance levels of 100 $\mu g \cdot L^{-1}$ of chlorophyll *a*. Ultimately, however, the receiving environment is the most sensitive component to changes in phosphorus loads and as such should dictate phosphorus levels. It is recommended that the lake trigger ranges be applied to rivers as well as lakes, especially when they flow into lakes. Similarly, where rivers are entering coastal bays and estuaries, the objectives of these coastal environments should be considered as well as the river itself. Nitrogen is normally the limiting nutrient in marine environments and controlling nitrogen inputs while having increases in phosphorus loads will change the nutrient ratio and may change the biological community. In this situation, controlling both phosphorus and nitrogen will be necessary so that an acceptable N:P ratio is maintained. The river ranges outlined in Table 4.2 have been adopted by the USEPA (EPA 2000b). In defining suitable phosphorus levels, the water body of concern is compared to its respective trigger range, based on its defined reference value.

It is stressed at this point that the CCME endorses a 'no degradation' policy, and that these values therefore do not provide, and must not be used as 'pollute up to levels'.

4.3.5.3 Determination of the current phosphorus concentration

Orthophosphate (PO_4^{3-}) is the biologically active form of phosphorus, but the turnover rate of PO_4^{3-} in phosphorus limited systems is extremely rapid making its measurement difficult. Filtration of the water sample prior to analysis for PO_4^{3-} can overestimate biologically available phosphorus (Fisher and Lean 1992). Conventional methods for measuring PO_4^{3-} generally overestimate phosphate concentrations (Hudson et al. 2000). Based on these limitations, TP is generally recommended as a meaningful measurement of phosphorus in surface waters (Wetzel 2001). The appropriate trigger ranges for a given site is selected from Table 4.3, based on reference/baseline TP concentrations established from historical records or the 25th percentile approach.

Variation in discharge on both seasonal and annual time scales strongly influences whether nutrients are stored or exported. It is important that a representative value for phosphorus is considered. TP may be expressed as a single measurement taken at spring turnover or as an average of several observations made on a seasonal basis; it may be an estimate for a specific zone in a lake (e.g., the euphotic zone), or as a whole lake approximation (Clark and Hutchinson 1992).

Two of the most common reporting approaches are an ice-free mean or concentration at spring turnover. The ice-free estimate is derived as the average of several samples taken over the ice-free season, and is the sampling method favoured here. Ice-free estimates, which encompass a period of stratification in the lake, require the collection of volume-weighted samples during stratification in order to derive proper whole lake averages. Samples collected only at the surface during stratification should not be included in these estimates of TP since they would not allow for elevated hypolimnion TP concentrations. Should these data be unavailable, the value recorded immediately after ice-melt may be considered as a conservative substitute; at ice-melt the available phosphorus is at its highest. Spring turnover estimates are generally based on the results from a single spring sample. TP measurements from the ice-free period and spring turnover may differ due to the resuspension of sediment phosphorus at spring turnover in shallow lakes, the contribution of TP accumulated under ice to spring values, and the loss of phosphorus to the sediments through settling of algal cells during stratification.

Equations exist, however (Appendix A), to convert TP from one sampling regime to another. In Arctic lakes, which are often under ice for up to 8 months of the year and typically have low natural phosphorus levels, the more conservative method of sampling immediately after ice melt is recommended.

The calculation of annual loads may be inappropriate for sites receiving seasonally-variable loads. The assumption of complete and instantaneous mixing of nutrient loads may also be inappropriate for lakes that exhibit marked spatial differences in limnological characteristics because of their morphological and hydrological features (Kennedy et al. 1986). In such cases, the most sensitive areas, and areas of concern (e.g., river mouth, point sources, or littoral zones), should be identified, and these values used to measure the phosphorus concentration.

Because phosphorus concentrations in water can be highly correlated with the sediment load, which may be highly variable and episodic, in monitoring programs grab samples may not provide representative phosphorus concentrations in lake or river water. In view of the variability in TP concentration, it is important that an appropriate number of samples be collected to accurately reflect TP concentrations in a system. The optimum number of samples can be calculated using the equations outlined in Appendix B (Green 1979), and a clear guide to sampling protocol is outlined by Clark and Hutchinson (1992).

4.3.5.4 Assessment of current concentration to trigger range

The upper concentration of the trigger range represents the maximum acceptable concentration of phosphorus within each of the trophic categories. If the upper limit of the trigger range is exceeded, or is likely to be exceeded, there is a risk of an impact either occurring or having occurred. At this stage, additional information on local environmental factors needs to be considered, and thus further assessment is recommended. The assessment could potentially lead to remedial advice and restoration of degraded water body. If the trigger range is not exceeded, the risk of an impact is regarded as low.

4.3.5.5 Assessment of current concentration to baseline condition

Due to the general nature of the trigger values and the size of some of the phosphorus ranges defined, a second precaution is taken in the assessment of possible effects of phosphorus. In the event that the trigger value has not been exceeded, the question is now raised as to the degree of increase in phosphorus levels from the baseline. Up to a 50% increase in TP concentrations above the baseline level was deemed by the Ontario Ministry of Environment (OMOE 1997) as an acceptable increase, beyond which visual deterioration of water quality from excessive phosphorus levels was observed in Pre-Cambrian Shield lakes. Clark and Hutchinson (1992) found that a 50% increase in lakes with an already high phosphorus baseline (up to 12 $\mu g \cdot L^{-1}$) might not protect against decreases in DO. Decrease in hypolimnetic DO is a very important measure of environmental stress in lakes, both because of the relationship between oxygen concentration and biological stress, and because oxygen levels control many chemical oxidation/reduction reactions in aquatic habitats. However, in the absence of empirical data to recommend an alternative for these lakes, the 50% increase limit is deemed preferable to no limit. The Canadian water quality guidelines for the lowest acceptable DO concentration are 6 and 5.5 $mg \cdot L^{-1}$ for the early and other life stages, respectively, in warm-water ecosystems, and 9.5 and 6.5 $mg \cdot L^{-1}$ for the early and other life stages, respectively, in cold-water ecosystems (CCME 1999). Molot et al. (1992) observed impairment in fish community as DO concentrations declined at epilimnetic TP levels of around 15 $\mu g \cdot L^{-1}$. Therefore, if the water body supports fisheries, further assessment is advised if TP levels are greater than 12 $\mu g \cdot L^{-1}$.

In large lakes, the 50% increase should be applied to the most sensitive areas (e.g., river mouth, point sources, or the littoral zone) rather than averaged over the whole lake. To be protective of the receiving environments, the 50% increase check is also applied to river systems. As the 50% value was originally defined to protect Pre-Cambrian Shield lakes in Ontario, it is felt that it will also be sufficient in protecting other northern systems, such as Boreal Shield and Arctic lakes. If a 50% increase from the baseline is not observed, then there is considered a low risk of adverse effects, and no further action is required. Should there be an increase from the baseline of greater than 50%, the risk of observable effects is considered to be high, and further modelling is recommended

5. RECOMMENDED ASSESSMENT

5.1 Introduction

5.1.1 Lakes

Eutrophication is the most frequent cause of impaired surface water quality (Molot et al. 1992; Smith et al. 1999). Managing eutrophication, especially in lakes and reservoirs, typically involves the control of phosphorus inputs because it is often the limiting nutrient (Hession and Storm 2000). Traditional management approaches that focus on reducing phosphorus loading from a single sector or employing technology-based phosphorus limits are often inappropriate; there is a need for holistic, best-practice approaches for ensuring sustainable use of water resources (EPA 2000a). Due to the complexity of aquatic ecosystems, management activities to address water quality impairment require a watershed-level approach. In an attempt to understand watershed processes in lakes, rivers, and streams, assessment and management activities invariably include monitoring (chemical and/or biological) and mathematical modelling. Monitoring is important to assess the quality of water and to identify impacts. Hydrologic and water quality mathematical models are used to help understand and investigate complex watershed-level processes, fill gaps in monitoring data, identify sources of pollution, predict system response to change, and evaluate management alternatives.

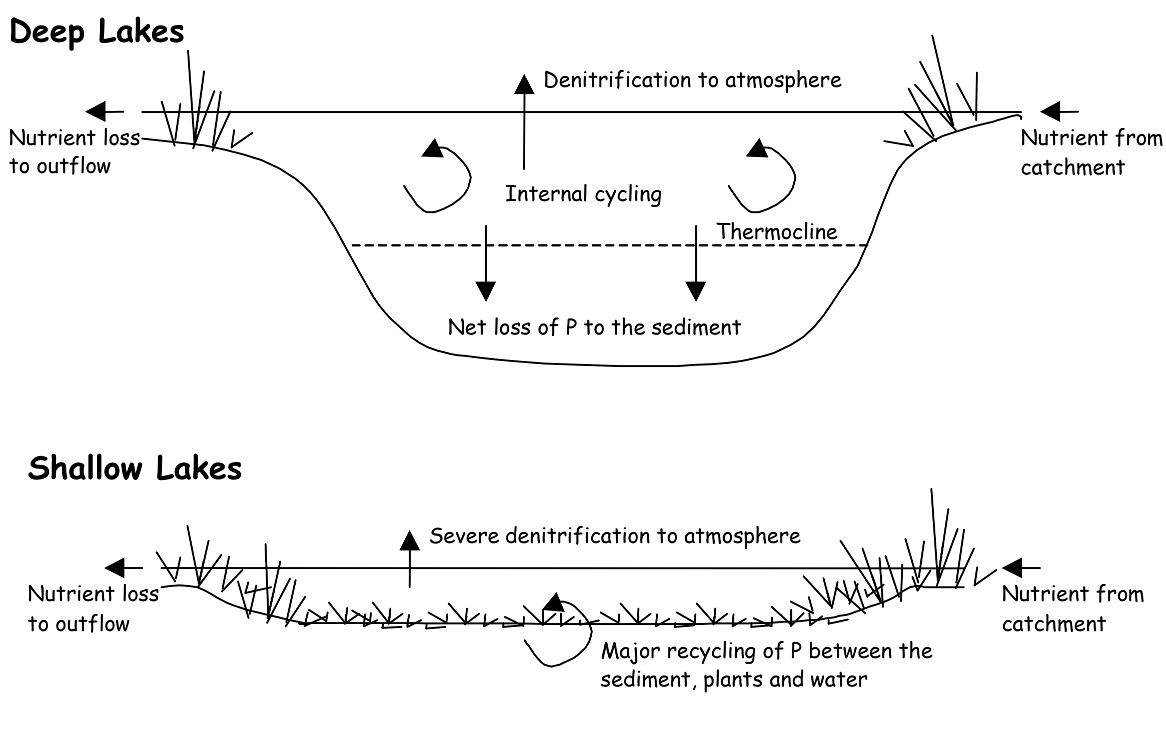

Figure 5.1 Nutrient cycling in deep and shallow lakes (source: Moss 1998).

Large lakes often attract the most attention when problems of eutrophication are being addressed. However, they occupy only a small portion of the world's freshwater area (Moss 1998). It is assumed that the mechanisms that are important in the functioning of the ecosystems of deep lakes, dominated by phytoplankton, must also apply directly to shallow lakes. The behaviour of nutrients in deep lakes can be described by simple models, as the mass of water in such a lake basin is relatively uniform (Moss 1998). In shallow lakes, however, light can often penetrate to the bottom sediments, allowing these systems to often be dominated by macrophytes (Figure 5.1). Research on shallow lake systems has shown how different the systems are to deep lakes (Moss 1998; Havens et al. 2001), and how nutrient cycling in flowing waters is governed by much different mechanisms than those of standing waters (Dodds et al. 1998).

In shallow lakes the situation is quite different to that of deep lakes because the bottom sediments are repeatedly resuspended and brought into contact with the water (Golterman 1982); this water/sediment system is often described as an equilibrium (Figure 5.2). Depending upon the hydrological conditions, the curve shown in Figure 5.2 may be more or less steep. The importance of the curve lies in the fact that it closely describes how the concentration of phosphorus slowly rises during the early stages of eutrophication. At a certain point the concentration may pass through a threshold value and it will then increase rapidly (although not exponentially).

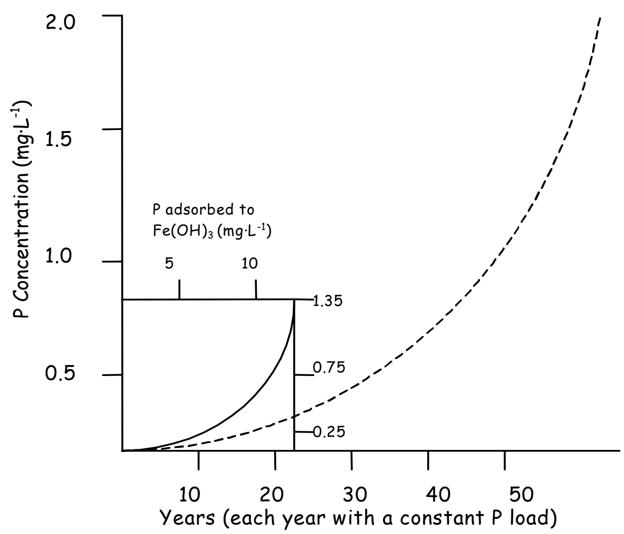

Figure 5.2 Theoretical curve (dashed line) for phosphorus concentration in a sediment/water system and phosphorus concentration adsorbed to $Fe(OH_3)$ (solid line). Phosphorus concentration is given on left and middle y-axes; lower x axis is years with each year receiving a constant phosphorus load; upper x axis is phosphorus adsorbed to $Fe(OH)_3$ (adapted from Golterman 1982).

In shallow lakes, water quality conditions (TP, turbidity, chlorophyll *a*, and algal blooms) may have complex relationships with external nutrient loads and in lake processing. Shallow eutrophic lakes can display complex nutrient dynamics, with processes such as sediment-water exchange playing a much greater role than in deep, dimictic lakes. These types of lake may display: a lack of stable long-term thermal stratification; frequent mixing of the entire water column and resuspension of unconsolidated sediments; and, substantial internal loading of nutrients from the sediments to the water column (Havens et al. 2001).

5.1.2 Rivers

The amount of phosphorus and nitrogen in river water is significantly influenced by anthropogenic inputs associated with land cover, land use, and point sources (Castillo et al. 2000). Rivers export large quantities of nutrients to lakes and coastal waters, where the adverse effect of nutrient enrichment is most clearly seen (EPA 2000b). An analysis of 175 relatively large watersheds, located throughout the USA (Omernik et al. 1981) differing in land use and without point source inputs, demonstrated a strong correlation of phosphorus and nitrogen concentrations with the fraction of land used for agriculture. Here, TP increased with increased river flow rate during storm events (Figure 5.3). It has been suggested that increased phosphorus comes from the river bottom, banks and flood plains during storm events (Verhoff et al. 1982). The water runoff from other parts of the basin can also contribute to these sources of TP.

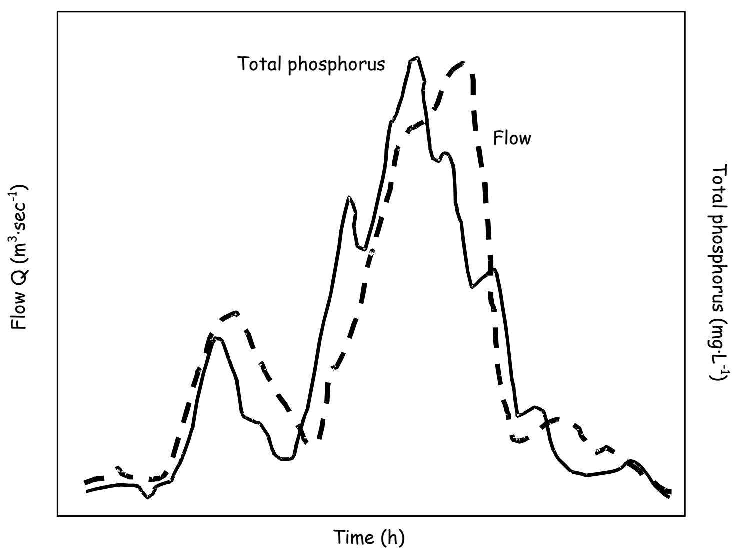

Figure 5.3 Hydrograph and chemograph for TP concentration during a storm event (source: Verhoff et al. 1982).

A number of biological processes influence the ability of stream ecosystems to utilize, retain and recycle particulate and dissolved nutrients. Primarily autotrophs and microbes accomplish the biological removal of nutrients from stream water. Wherever their biomass is large and populations are metabolically active, removal rates are highest. Aquatic macrophytes are capable of removing substantial amounts of nutrients from flowing waters. The topography of the catchment basin influences the extent of erosion and subsequent export of nutrients. Furthermore, the relative erosion is influenced markedly by the type of vegetation and use to which the land is put. Geological characteristics can, and should, be used when defining a water system as it influences physical and chemical, and thus biological characteristics of the water body. The hydrologic regime also influences nutrient uptake by its effect on the standing stock and productivity of the biological community, and thus on the amount of material stored in biological tissue. Similarly, storms can influence uptake by affecting the standing stock of leaf and organic matter (Stumm and Morgan 1996). Hydrology also interacts with instream retention devices and geomorphological features of river channels to determine the relative importance of transport and storage of nutrient dynamics. Retention and uptake is favoured by low flow, a high ratio of streambed area to channel volume, retention devices such as debris, dams, and beaver ponds, and permeable substrates that allow substantial interstitial flow. Through-flow of inputs and export of sorted materials are favoured by the opposite conditions.

Regional differences in watershed characteristics make it unlikely that a single, numerical trophic status objective could adequately protect water quality over large jurisdictions (Hutchinson et al. 1991; EPA 2000a; 2001). Different variables govern phosphorus availability in different waterbodies. The water type must be considered when modelling phosphorus, and the driving variables should be selected accordingly. Chlorophyll *a* is the parameter most often used to qualify the biomass of primary producers in the water, and as a result, chlorophyll *a* levels are often used as indicators of trophic status. Algal density has a major effect on water clarity, and there is a relationship between water clarity (Secchi depth) and algal biomass in many lakes. Consequently, Secchi depth has also been commonly used to approximate trophic status.

TP, TN, chlorophyll *a*, and Secchi depth must all be considered when modelling lakes and reservoirs. It is advised that additional variables to these also be considered. When modelling shallow lakes (i.e., lakes which are completely mixed, and in which light penetrates to the bottom sediments), sediment phosphorus plays an important role in phosphorus cycling and should also be considered. Benthic and sestonic chlorophyll *a* also needs to be considered for these systems. In modelling rivers and streams, discharge and velocity are important factors in the determination of phosphorus availability. Some of the parameters that influence, and are influenced by phosphorus uptake and availability are outlined in Chapter 2 of this report. It is strongly advised that additional variables (e.g., DO, algal community composition, pH, iron and calcium carbonate concentration) be incorporated into the models. When a lake supports communities of coldwater fish, oxygen concentrations are often critically important (Clark and Hutchinson 1992). Lake morphology has been shown to be the most influential parameter for predicting hypolimnetic oxygen concentrations in oligotrophic lakes, but the oxygen regime is also modified by changes in lake phosphorus concentrations. As a result, oxygen profiles are often considered as a consequence of trophic status. The variables selected for the models should reflect the ecosystem under investigation and the desired uses of that system.

5.2 Proposed Assessment Tools

There are a number of possible approaches that can be used to assess trophic status of aquatic environment. These are divided into three categories (Chapter 4, Figure 4.1):

i. The development of a WQI. A WQI can be used as a surrogate for TP in the framework and can also provide a single value that identify the current state of the ecosystem, including the percent change from the reference/ baseline condition and an index of biotic integrity (IBI);

ii. The use of multivariate analytical methods also assist in comparing current conditions to baseline conditions. These methods allow sites to be compared based on a suite of variables related to nutrients. The degree of impairment can be identified (both spatially and temporally). Interactions with additional environmental variables and that may impair the system may also be identified;

iii. The use of existing models. Numerous models have already been developed, and are presently being used for the management of phosphorus. For example, paleoecology based TP reconstruction and Lakeshore Capacity models have been used in the Muskoka region of Ontario; these models allow predevelopment conditions to be identified, from which the degree of allowable change in a system can be identified. Export Coefficient Modelling makes forecasting and hindcasting of changes in water quality as a result of changes in land use possible. The modelling approach provides a better understanding of fate/ effects of nutrients and thus direct management action.

5.2.1 Water quality index

A water quality index (WQI) provides a means of summarizing complex water quality data into a single value. Variables chosen in WQI provide relevant information about a particular site; however, physical, chemical, and biological parameters that affect phosphorus availability, such as depth, plant biomass, fish species, etc., do not have guidelines associated with them, and cannot be incorporated into the model. Because the use of a large number of variables may artificially deflate the WQI value, care must be taken when selecting appropriate variables for the index. Summarizing data into a single value results in a loss of information. However, it has been argued that data simplification is the goal of a multimetric index, and that this feature allows them to be used by resource managers who may not be experts in watershed management.

5.2.1.1 CCME's Water Quality Index

The CCME (2001) WQI is based on the approach developed by the BC Ministry of Environment, Lands and Parks (1995) and modified by Alberta (Wright et al. 1999). The index provides a mathematical framework for assessing ambient water quality conditions relative to water quality objectives. It is flexible with respect to the type and number of water quality variables to be tested, the period of application, and the type of water body (stream, river reach, lake, etc.) tested. The Index incorporates three elements: *scope* - the number of variables not meeting water quality objectives; *frequency* - the number of times these objectives are not met; and *amplitude* - the amount by which the objectives are not met. The index produces a number between 0 and 100, and these numbers are divided into 5 descriptive categories:

Excellent: (WQI Value 95-100) – water quality is protected with a virtual absence of threat or impairment; conditions very close to natural or pristine levels.

Good: (WQI Value 80-94) – water quality is protected with only a minor degree of threat or impairment; conditions rarely depart from natural or desirable levels.

Fair: (WQI Value 65-79) – water quality is usually protected but occasionally threatened or impaired; conditions sometimes depart from natural or desirable levels.

Marginal: (WQI Value 45-64) – water quality is frequently threatened or impaired; conditions often depart from natural or desirable levels.

Poor: (WQI Value 0-44) – water quality is almost always threatened or impaired; conditions usually depart from natural or desirable levels.

The CCME WQI requires that at least four variables, sampled a minimum of four times, be used. The selection of appropriate water quality variables for a particular region is essential for the index to yield meaningful results. Choosing a small number of variables for which the objectives are not met will provide a different picture than if a large number of variables are considered, only some of which do not meet objectives. It is up to the professional judgment of the user to determine which and how many variables should be included in index to most adequately characterize region or site-specific water quality. The CCME WQI has been applied to data sets from across Canada (e.g., Ontario, Newfoundland, Saskatchewan, Alberta) using a range of variables (e.g., nutrients, metals, pH, turbidity, and DO). These examples have provided a clear example of how a water quality index can summarize complex water quality data and facilitate communication of results among different levels of governments.

5.2.1.2 A strategy for the classification of lakes - UK

Johnes et al. (1994) developed a scheme for the assessment and monitoring of water and ecological quality in standing waters in England and Wales. Although this scheme was designed specifically for waters greater than 1 ha in area, similar methods could be adopted for other water types. Thirteen hydrological, chemical, and biological variables were used to characterize the water body: volume; maximum depth; conductivity; Secchi disc transparency; pH; total alkalinity; calcium; TN; winter oxidized organic nitrogen; TP; chlorophyll *a*; a score based on the number of submerged and emergent plant communities, which consists of a ranking scheme already in use by English Nature (Appendix C); and, the presence or absence of a fish community. The scheme developed by Johnes et al. (1994) was designed to address the state of eutrophication, acidification and sedimentation, and therefore considers key indicators of these states (Table 5.1). Oxygen profile, benthic invertebrate, phytoplankton, and fish communities were considered, but not used in the scheme. These variables were rejected because of the inadequacies in the existing data available and the great heterogeneity of the community in time or space within a water body that was expected. The scheme is not closed to the addition of more variables being incorporated, should suitable reference data become available.

Table 5.1 Variables incorporated into the WQI that are associated with particular change in lakes (source: Johnes et al. 1994).

EUTROPHICATION	ACIDIFICATION	SEDIMENTATION	SALINISATION
Inflow TP and TN	pH	Volume	Conductivity
Lake TP and TN	Conductivity	Maximum depth	
Winter nitrate	Total alkalinity	Secchi transparency	
Lake and inflow N:P ratios	Calcium	Retention time	
Secchi depth	Potential for fish community		
Maximum chlorophyll *a*	Maximum chlorophyll *a*		
Plant ranking score	Plant ranking score		

Within the scheme (Johnes et al. 1994), standing waters are classified using a very limited number of variables and essentially arbitrary categories; the variables are compared with a reference baseline state and then expressed as a percentage change from the baseline. A mean percentage change in all variables can be calculated or the values can be sub-grouped to indicate change in the directions of eutrophication, acidification, sedimentation, or potentially any other impact if the appropriate variables are incorporated and can be determined in the baseline state. The baseline state also provides an objective reference for the determination of water quality objectives for given water bodies. Changes can be classified based on their degree of change (percentage change) from the baseline. Johnes et al. (1994) suggest the following categories: Class 1, negative changes and positive changes < 25%; Class 2, 26-50% change; Class 3, 51-100% change; Class 4, 101-200% change; Class 5, > 200% change. However, the final choice of appropriate class boundaries should be determined based on the use of the water system.

5.2.1.3 Periphyton Index of Biotic Integrity (PIBI) – Canada, USA, Europe

In recent years, algal proliferations have become more of a concern in rural and urban rivers. Benthic algae (periphyton) are primary producers and are important foundation of food webs in both lakes and rivers. Among algal indicators, diatoms are in particular useful ecological indicators because they are found in abundance in most lotic and lentic ecosystems. A great number of species are sensitive indicators of environmental change and identify specific conditions of their habitat (Dixit et al. 1992). Several studies suggest that taxonomic indicators based on periphyton and diatoms can be used as a standardized protocol for monitoring ecosystem change (McCormick and Cairns 1994; Kelly et al. 1995; Hill et al. 2000; Winter and Duthie 2000). McCormick and Cairns (1994), in their overview on the role of algae as indicators of environmental change, suggest that both population- and community-level indices have inherent strengths and limitations, and information from both levels of biological organisations should ideally be used in tandem.

Hill et al. (2000) outline three approaches for using periphyton to assess stream water quality:

i. The oldest approach is based on the indicator species concept, whereby tolerances of algal species are used to determine water quality, the oldest of these being the Saprobien system of Kolkwitz and Marsson (1908);

ii. An approach based on the assumption that pristine environments support a greater diversity of organisms than degraded environments, and that measurement of community structure (diversity, evenness, richness and similarity) reveal the health of the system; and

iii. Biotic indices which use biological assemblages to assess water quality and ecosystem integrity. Biotic indices have been developed to avoid the problems associated with both indicator species and community structure approaches to water quality assessment, and are based on the correlation of assemblage data with current and past physico-chemical parameters.

The diverse nature of streams and their stressors requires an index of integrity that is broadly based, multimeric, and responsive at both the individual and assemblage levels. Hill et al. (2000) felt that such an index should be composed of individual metrics that have clear relationships between biological responses and specific stressors, and that these metrics should be integrated into a composite index. The resulting index would provide a quick estimate of the overall condition of the stream or river. The index developed by Hill et al. (2000) uses measures of species richness, trophic structure, and organism abundance. They focused their examinations of the periphyton community on 10 measures:

i. Algal genera richness;

ii. Relative abundance of diatoms, cyanobacteria, dominant diatom genus, acidophilic diatoms, eutraphentic diatoms, and motile diatoms;

iii. Chlorophyll and biomass (ash-free dry mass) standing crop;

iv. Alkaline phosphatase enzyme activity.

Table 5.2 Metrics included in the PIBI. AFDM = ash free dry mass, APA = alkaline phosphatase activity, chl = chlorophyll (source: Hill et al. 2000).

Metric	Calculation	Range	Score[a]
Relative taxa richness	No. algal genera/expected no. algal genera[b]	0-1	0-10
Diatoms	No. diatom cells/total no. algal cells	0-1	0-10
Cyanobacteria	1 - (no. cyanobacteria cells/total no. diatoms)	0-1	0-10
Dominant diatoms	1 - (no. dominant diatoms/total no. diatoms)	0-1	0-10
Acidophilic diatoms	1 - (no. acidophilic diatoms/total no. diatoms)	0-1	0-10
Eutraphentic diatoms	1 - (no. eutraphentic diatoms/total no. diatoms)	0-1	0-10
Motile diatoms	1 - (no. motile diatoms/total no. diatoms)	0-1	0-10
Chlorophyll[c]	6.67/(absolute value of 6.67 ± Chl [$mg{\cdot}m^{-2}$])	0-1	0-10
Biomass[c]	0.006/(absolute value of 0.006 ± AFDM [$g{\cdot}m^{-2}$])	0-1	0-10
Phosphatase activity	18.2/(absolute value of 18.2 ± APA [$nmol^{-}1{\cdot}h^{-1}$])	0-1	0-10
Range of PIBI scores			0-100

[a]Score range is calculated by multiplying the raw range by 10.
[b]Expected number of genera is the observed maximum genera richness of each year.
[c]Chl, AFDM and APA are 2-tailed metrics, which have low scores when both lower and higher than median standing crops. Median values for each metric are given in the numerator for each formula.

The relative richness and relative abundance (RA) of both the diatom and non-diatom genera were used to calculate the RA metrics of the PIBI. The 10 metrics (Table 5.2) and PIBI were correlated with 27 chemical, 12 physical habitat, and 3 landscape variables (Table 5.3). The summed scores for each of the 10 metrics are used to derive the PIBI score. This score ranges from 0 to 100, with a score of 100 representing a pristine condition. Hill et al. (2000) outline some of the shortfalls, as well as the advantages of the PIBI method they used and its metrics.

5.2.2 Multivariate approach

There are many variables that both directly and indirectly affect phosphorus availability that should be considered when setting criteria for phosphorus. Problems arise when trying to compare large numbers of variables, and data is inherently lost when these data are condensed into a single value such as WQI. Multivariate methods allow an array of physical, chemical and biological parameters to be considered regardless of whether guidelines are available for them or not. A visual output can be provided, and the effects of underlying data can be viewed on an ordination plot. The influence of the different parameters on the ordination can then be identified. The underlying mathematics of these models can be complex, however, this approach has become attractive in understanding impacts on environmental stressors on aquatic ecosystems.

Table 5.3 Measured chemical, physical habitat and landscape variables used by Hill et al. (2000) in the development of a PIBI.

Chemical variables	Physical habitat variables	Landscape variables
Alkalinity	% fine-grain sediments	Watershed agricultural land use
Total dissolved Al	% sand	Watershed forest land use
Acid neutralizing capacity	% sand + fine-grain sediments	Watershed urban land use
Sum of anions	% small gravel	
Sum of cations	Mean residual stream depth	
Ca	Channel sinuosity	
Cl, CO_3	Mean canopy density	
Colour	Mean channel embeddedness	
Conductivity	Mean stream reach	
Fe, HCO_3	Mean channel wetted width	
Ionic strength	Sum agricultural activity	
K, Mg, Mn, Na, NH_4, NO_3	Sum human disturbances	
Total N	Mean stream width/depth ratio	
Estimated organic anions		
pH		
Total P		
SiO_2, SO_4		
Sum of base cations		
Total suspended solids		
Turbidity		

5.2.2.1 Biological guidelines for the assessment of sediment quality – Canada

Based on a scheme developed in the UK (River Invertebrate Prediction And Classification System (RIVPACS): Wright et al. 1984; Moss et al. 1987; Armitage et al. 1987), Reynoldson and co-workers (Reynoldson and Zarull. 1993; Reynoldson et al. 1995; Reynoldson et al. 1997; Reynoldson and Day 1998) predicted benthic invertebrate community structure in clean, uncontaminated sites in the Great Lakes using simple habitat and water quality parameters. Reynoldson and Day (1998) describe the development of biological guidelines for sediments in near-shore fine-grain habitats in the Great Lakes using such a model.

There are a number of approaches to compare reference and test sites, for the purposes of making decisions on impairment. Multivariate statistical methods to examine patterns were used by Reynoldson and Day (1998), as they considered the methods to be unbiased and the methods allowed all of the information collected to be incorporated into the model. These methods are equally appropriate for determining whether a test community is equivalent to the reference condition and for setting guidelines that describe the degree of impairment. A large water quality survey on rivers conducted in the UK in 1990 provided the impetus for the development of methods to describe responses using a series of bands that represented grades of biological quality (Clarke et al. 1992). In setting guidelines for invertebrate community structure at a test site, Reynoldson and Day (1998) adopted a multivariate approach, an adaptation of Clarke's (1992) work, which would aid in defining the degree of impact (Figure 5.4). The algorithms used in this approach may be complex, but the output is relatively straightforward: the closer together two points are, the closer their community composition. The likelihood of sites being the same as the reference condition described by Reynoldson and Day (1998) was qualified by constructing three probability ellipses (90%, 99% and 99.9%) around the reference data (Figure 5.4). Data falling within the bounds of the first (90%) ellipse would be considered equivalent to the reference data, and therefore unstressed. Sites lying between this ellipse and the next largest (99% probability ellipse) would be given the designation of 'possibly different', whereas the sites between the 99% and 99.9% ellipses would be designated as 'different'. Anything lying outside of the outer (99.9%) ellipse would be considered very different from the reference condition.

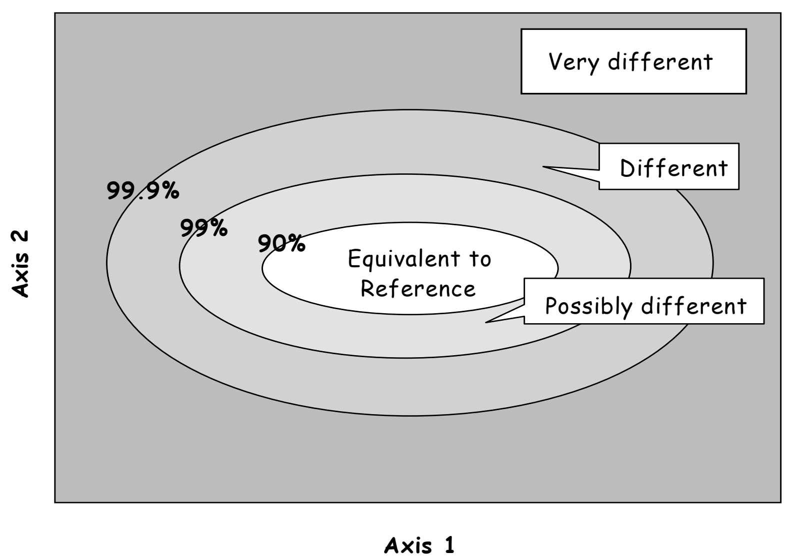

Figure 5.4 Impairment stress levels derived from reference sites in ordination space. Bands, based on 90, 99 and 99.9% probability ellipses, are identified as 'equivalent to reference', 'possibly different', 'different' and 'very different' (source: Reynoldson and Day 1998).

5.2.2.2 Using the normal range as a criterion for ecological significance – Canada

Environment Canada (1997) has defined impacts on fish population endpoints and benthic invertebrate community composition as any changes that exceed the normal range of variation observed in reference conditions. Normal ranges are defined as the range of values enclosing 95% of the population of reference values. When the number of locations used to characterize the reference condition is large, observed percentiles for the upper and lower ends of the 95% region can be considered appropriate as biocriteria. However, if there are only a limited number of reference locations, then the use of percentiles can lead to errors in assessment of test locations because there is error in the estimated upper and lower percentiles (Berthouex and Hau 1991). As a consequence, it is more appropriate to use a statistical test that determines the probability that the test location lies outside of the true normal, when there is a low number of reference locations. Kilgour et al. (1998) describe two such statistical tests which are based on non-central distributions of test statistics and applicable to both univariate and multivariate responses. Kilgour et al. (1998) compared the performance of these two tests and concluded that the 95% rule provides a generic criterion to assess the ecological relevance of environmental impacts through direct comparison to the normal range of variation at reference locations. The criterion can be used for any parameter, irrespective of the scale of measurement. A single index can be used to objectively determine if a test site is significantly different from the regional reference (Kilgour et al. 1998).

Multivariate methods are designed for analyses involving more than one variable. In particular, multivariate methods take correlations between variables into account. And as Kilgour et al. (1998) outline, we can use a multivariate *t*-test to determine whether a test site is significantly different from the reference condition using the different variables simultaneously. The statistical analysis can be illustrated with a discriminant function, which maximally separates the test site from the mean of the reference sites (Figure 5.5, OMOE 1999a). When the correlation between the two variables is taken into account, the test site is clearly outside of the range of the reference sites. Discriminant analysis provides an objective way to combine information from a series of different indices and thereby judge impairment.

Multivariate *t*-tests provide an objective way to assess a test site when reference data are available (Kilgour et al. 1998). The discriminant analysis highlights those variables that are most important for distinguishing the test site from the reference sites (OMOE 1999a).

5.2.3 Existing models

Models exist that have already been developed and implemented by other jurisdictions within Canada and the USA. These models can be incorporated into the management framework outlined in this report.

5.2.3.1 Lakeshore Capacity Model – Canada

The LCM quantifies the linkages between natural sources of phosphorus to the lake, human inputs from shoreline development, water balance of a watershed, size and shape of a lake and the resultant phosphorus concentration. The model was developed and calibrated for Precambrian Shield lakes in southern Ontario, but has been tested and used in lakes across the Precambrian Shield. It predicts several different indicators of water quality: TP concentration, algal density (chlorophyll *a* concentration), water clarity (Secchi depth), oxygen concentrations at the critical end-of-summer period, and the absolute volume of lake trout habitat.

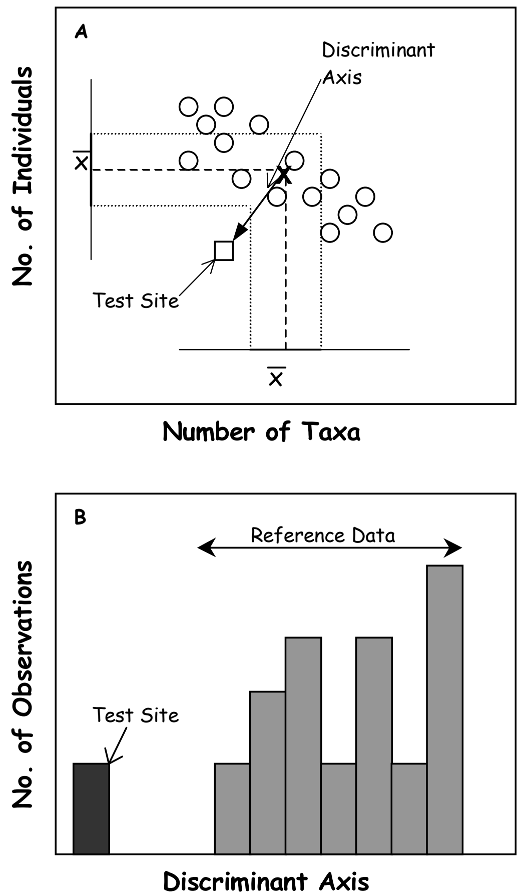

Figure 5.5 Illustration of where the discriminant axis is placed (A) and the place of the test site on this new axis (B) (source: OMOE 1999a).

The Lakeshore Capacity Strategy (Figure 5.6) provides the standards, tools, and guidance to manage nutrient enrichment from shoreline development. The strategy is based on the principles that: small increases in nutrient status are acceptable in most lakes, but all have a finite capacity for nutrient assimilation; nutrient enrichment may also reduce the amount of cold, well-oxygenated water used by fish and lakeshore capacities must therefore protect fish habitat. A consistent scientific process can then be described which will determine the maximum acceptable change in water quality for lakes. In developing this model, the strategy was applied to all inland lakes on the Precambrian Shield. Off-shield lakes were excluded, as well as lakes smaller than 10 ha (25 acres) in size, as these were considered not to constitute a popular recreational resource.

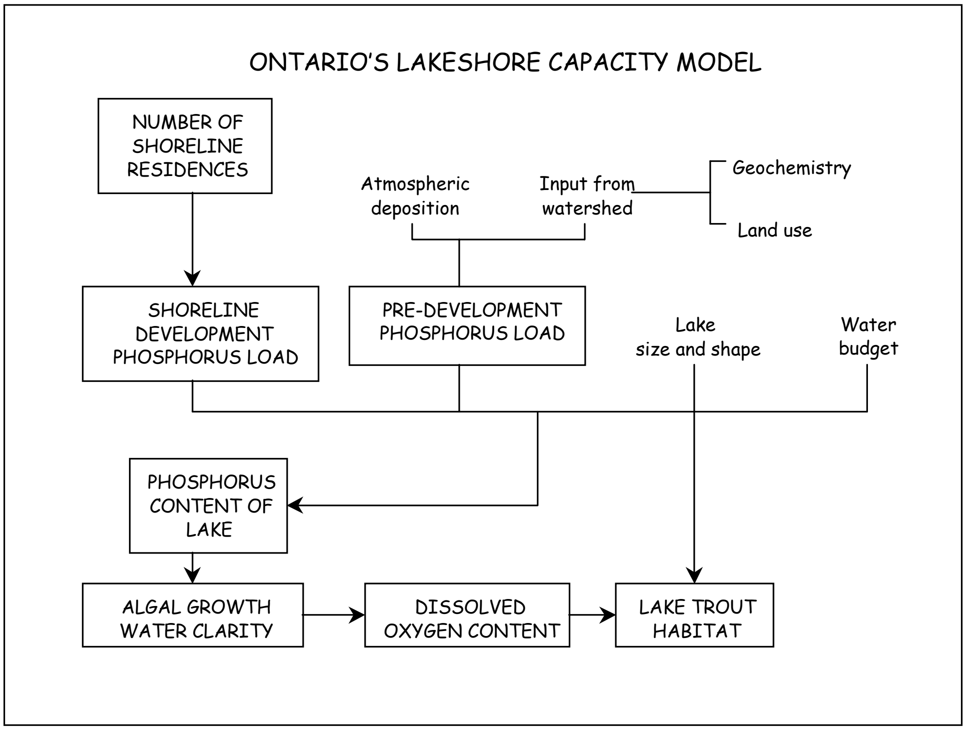

Figure 5.6 Schematic of Ontario's Lakeshore Capacity Model (source: OMOE 1997).

A critical assumption of the LCM is that all of the phosphorus added to a domestic septic system will ultimately move to the lake. This reflects the thin, organic or sandy soils and the fractured nature of the bedrock common to the Precambrian Shield which suggests that very little phosphorus can be removed by natural processes in the soil. Other assumptions of the model relate to seasonal or permanent uses of shoreline residences, and actual loading of phosphorus from the septic system or other sources.

The phosphorus budget of a lake is influenced by transport from upstream lakes and a portion of a lake's phosphorus load will be exported to the next lake downstream. These exported quantities can be predicted accurately and reliably from the information on the water budget of a lake and its oxygen content. Protection of water quality in a lake requires that these phosphorus contributions from upstream sources of shoreline development be included in its phosphorus budget and capacity limits. Development must, therefore, be limited on upstream lakes when a lake reaches its capacity. If not, phosphorus export from upstream sources will cause the lake to exceed its water quality objective.

The model considers the interactions between interconnected lakes (OMOE 1999b). Using physical and chemical characteristics, lakes and watersheds are first modelled with all existing shoreline development and lots. The pre-development phosphorus concentrations of lakes are calculated by modelling the TP budget for the lake and subtracting that portion of the budget that is attributed to shoreline development. This pre-development baseline serves as the starting point for all future assessments. Changes in the water quality, resulting from further development, are then predicted.

The model then calculates proportional increases in phosphorus concentration from the predevelopment condition. After the watershed has been modelled, by adding the required information for each lake, models can be developed to make hindcasts and/ or predictions of phosphorus and oxygen levels for the lakes in the modelled watershed. From this, the user is able to:

i. Calculate the current expected TP at spring turnover;

ii. Calculate the TP at spring turnover (with no shoreline development), or pre-impact TP at spring turnover (no shoreline development, agriculture or urbanisation);

iii. Calculate further expected TP at spring turnover on proposed development scenarios;

iv. Calculate allowable development based on defined nutrients/ oxygen targets; or

v. Calculate the end-of-summer hypolimnetic oxygen profile, percent optimal habitat volumes, and the volume-weighted hypolimnetic oxygen concentrations.

The new objective for phosphorus would be an increase of 50% above the modelled reference/ predevelopment concentration. A list of the characteristics that need to be defined in the model are provided in Appendix D. Further details of the model and its uses are outlined in two OMOE guidance documents (1999b; 1999c). The software for this model can also be obtained through the OMOE.

5.2.3.2 Export Coefficient Modelling – North America and UK

Export Coefficient Modelling (ECM) was originally developed in North America to predict nutrient inputs to lakes and streams (Dillon and Kirchner 1975; Omernik 1976; Reckhow et al. 1980; Beaulac and Reckhow 1982; Rast and Lee 1983). Unlike the traditional methods for assessing the impact of land change on water quality, which involve the development of detailed physically-based models, ECM is very simple (Johnes et al. 1996). The modelling approach aims to predict the concentrations of TN and TP at any site in a stream as a function of the export of nutrients from each of the nutrient sources in the watershed (Johnes et al. 1996). The use of nutrient export coefficients for estimating loads for nitrogen and phosphorus is based on the knowledge that, for a given climatological regime, specific land use types will yield or export characteristic quantities of these nutrients to a downstream water body over an annual cycle (Rast and Lee 1983). Knowing the area of land devoted to specific uses in a given watershed, livestock headage and human population, the quantities of nutrients exported from these areas (the nutrient coefficients), and the river discharge, these models can be used for predicting TN and phosphorus levels in receiving waters (Figure 5.7). Export coefficients are derived from the literature and results of field experiments to determine the loss of nutrients from each identifiable source to the stream. Such models provide an effective means of evaluating the impact of land use, and can be used to forecast the effects of changes in land use in the future and to hindcast past water quality to establish comparative or baseline states for monitoring change (Johnes et al. 1996).

Recently, this kind of technique has successfully been modified and used to determine nutrient loads to surface waters in the UK (Johnes et al. 1994; 1996) and has been used to estimate loading from non-point sources in Canada (Winter and Duthie 2000). Winter and Duthie (2000) used export coefficient modelling to assess the influence of land use on phosphorus loading to a southern Ontario stream.

There are a number of advantages that export coefficient modelling has over the more detailed, process-based, modelling techniques. In particular, the model has a simple format and is operated through a spreadsheet (Johnes 1996). The approach relies on data from readily available sources, has relatively few data requirements and can easily be calibrated. It provides a robust means to evaluating the impact of land use and land management on nutrient loading to surface waters. To date, this type of modelling has predominantly been carried out on river systems in Canada and on lakes of greater than 2 ha and rivers in the UK. More details to this approach, and appropriate equations are given in Appendix E, and by Johnes (1994; 1996), Chambers and Dale (1997), and Winter and Duthie (2000).

5.2.3.3 AQUATOX – USA

The USEPA has just released a freshwater toxicity simulation model AQUATOX (EPA 2000c). The model predicts the fate of various pollutants, such as nutrients and organic toxicants and their effects on the ecosystem by simulating the transfer of biomass and chemicals from one compartment of the ecosystem to another. Using the AQUATOX model, Wellman and Park (2000) were able to integrate significant ecological processes including productivity, nutrient cycling, and trophic relationships to simulate nutrient parameters. These parameters included both causal and response variables, including phosphorus, nitrate, ammonia, chlorophyll *a*, Secchi depth, and DO. Wellman and Park (2000) suggest that this type of model can be used to provide a greater understanding of the interactions between causal and response variables in order to set appropriate, defensible criterion levels.

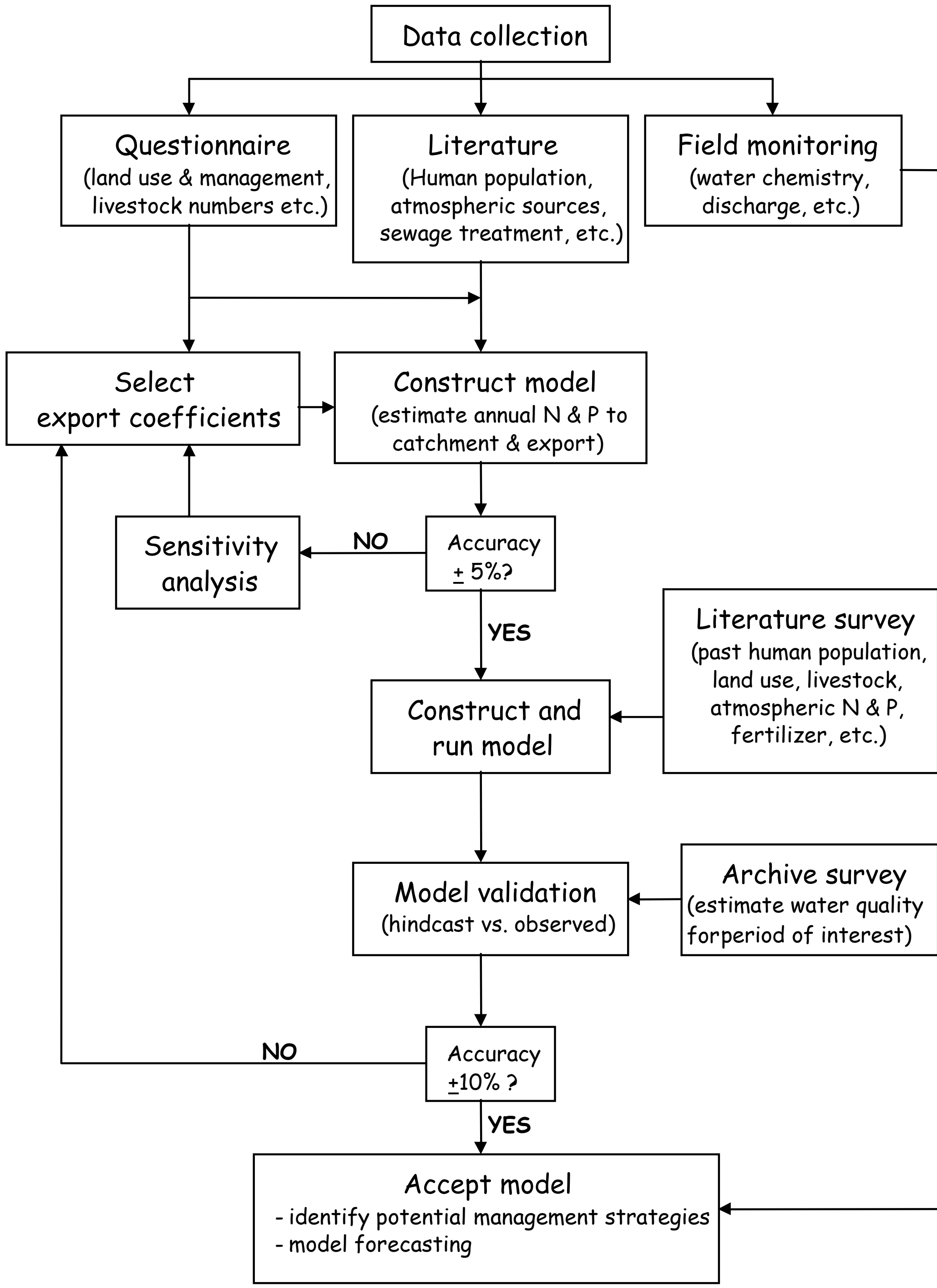

Figure 5.7 Summary of the export coefficient modelling procedure (adapted from Johnes 1996).

Export Coefficient Modelling to assess phosphorus loading in an urban watershed

Winter and Duthie (2000) used an export coefficient modelling approach to assess the influence of land use on phosphorus loading to a southern Ontario stream. The objectives of their study included: forecasting impact on the stream in terms of phosphorus loading of future urban development proposed for the watershed; and, illustrating how the export coefficient modelling approach might be used for watershed planning and decision making.

Data were collected for water chemistry, precipitation, and discharge, as well as land use. From this, appropriate export coefficients were selected for phosphorus loading from the different sources identified. Past TP data were obtained from existing OMOE databases, and precipitation data obtained from the Ontario Climate Centre, Environment Canada. Discharge data were collected by the Water Survey of Canada. Where discharge data were unavailable, the values were estimated. Land use data were compiled using maps provided by the Ontario Agricultural Resource Inventory (OMAF 1983), maps and information provided by the Grand River Conservation Authority (GRCA 1993), and the 1997 City of Waterloo zoning map. Historical land use was determined using the GRCA reports (MacMillan 1978; Veale 1981) and information from the 1976 agricultural census (Statistics Canada 1976). Export coefficients for loading of phosphorus from woodland areas in the watershed were selected to reflect forest type and geology. The input of phosphorus to the watershed via atmospheric deposition was taken from Kuntz (1980), and Reckhow et al. (1980) provided a list of urban export coefficients.

Sensitivity analysis of the model gave an indication of the relative importance of each nutrient source to the watershed, and reflected both the area of the nutrient source and the size of the export coefficient used. Phosphorus concentrations in the stream were largely determined by loading from the urban areas (Winter and Duthie 2000). A predominant increase in TP load was predicted when the model was used to forecast TP loads from future urban development planned for the stream's watershed. Thus, reductions in the export from the urban areas in the future were deemed to be the most efficient in lowering the phosphorus concentrations in the stream.

5.2.3.4 Diatom-based Phosphorus Inference Model

Diatom-inferred phosphorus reconstructions offer a powerful tool for eutrophication assessment and management (Hall and Smol 1999). TP inference models that can be applied to an individual site are available for many geographical regions of Canada. Using this approach, baseline or pre-disturbance P targets can be established and long-time series data can be developed for addressing natural variability and responses to eutrophication and recovery. The approach is applicable to addressing both site and region-specific issues (Dixit et al. 1999).

Diatom inferred phosphorus data have been used alongside with LCM data in southern Ontario. Both these models identified major trend in increase in TP in the southeastern Ontario lakes (Reavie et al. 2002). Nutrient increases were higher in lakes that have high present-day TP, although the trend was more obvious in LCM data. Diatom data identified phosphorus increase in 78% of lakes, with significant increase (>7.3 $\mu g{\cdot}L^{-1}$) in

29% of the studied lakes. It is important to mention that an increase of >7.3 $\mu g \cdot L^{-1}$ in some lakes may represent 100% or greater increase in P concentration. These results were somewhat different from a diatom-inference model study in south-central Ontario Precambrian Shield lakes (Hall and Smol 1996), where 78% of lakes showed no significant TP change, 4% increased in TP and 19% decreased. These comparative regional data identify that lakes in southeastern Ontario have apparently been more susceptible to cultural impacts than south-central Ontario lakes.

A detail discussion on advantages and limitations of diatom-inferred phosphorus model and LCM is provided in Reavie et al. (2002). In summary, the LCM is less labour-intensive than a diatom approach, especially for lakes with simple watersheds (i.e., few upstream lakes). However, for lakes with complex watersheds, LCM calculations become more difficult. The LCM has the advantage of being able to reconstruct the effects of continued human development in a lake's watershed because theoretical development scenarios can be incorporated. A major limitation of the LCM may be its inability to reconstruct a decline in nutrient concentrations. Because anthropogenic development in almost all catchment tends to be static or increasing with time, the model inherently infers no change or an increase in nutrients, respectively. This is a potential problem when using the model to infer TP changes in southern Ontario since declines in epilimnetic TP concentrations have been measured, while development has continued to increase. It would be necessary to refine LCM coefficients for a particular region or ecozone, especially in terms of recent conditions (e.g., better sewage treatment methods and more reliable engineering of septic systems). The use of an individual or a combination of modelling approaches would largely depend on the site or region specific requirements.

5.3 A Cautionary Note

The different methods discussed above are not exclusive, and often a combination of these methods may need to be adopted. For example, environmental variables have been compared to the baseline data (hindcast or best available) and expressed as a percent change (Johnes et al. 1994; Dixit et al. 1999), analyzed directly through multivariate analysis (Reynoldson and Day 1998), or through a suite of indices (Kilgour et al. 1998; OMOE 1999a; Reavie et al. 2002). However, the magnitude of change from the baseline can either be summarized as a single value (e.g., Johnes et al., 1994; CCME 2001), or if preferred, analyzed using multivariate techniques. Adaptations of existing models could also be considered. The management framework outlined above is a feedback model and incorporate continuous monitoring.

The methods outlined in this document should be viewed with the caveat that many of the methods described were developed for certain water types, and to answer specific questions. They may have underlying assumptions regarding the topography, geology and climate of those areas, as in the case of the LCM, and may require further development and validation for individual regions. Although many of these methods are applicable to different areas, care must be taken in selecting a method that is both applicable to specific water type and realistic to the users needs.

6. MANAGEMENT DECISIONS

6.1 Introduction

Once the potential increases in phosphorus concentrations have been assessed, the results are compared to the original goals (reference or baseline conditions) set at the beginning of the framework. The degree of change is then assessed on a management level, and the question raised, “are these changes acceptable?”. This question is largely based on the managing body’s interpretation of what constitutes “acceptable”. The protection of water resources is often linked to management (Porter 1978), and many of the outputs of management may be complementary but others may conflict. The most efficient way to manage aquatic resources is at the catchment or river basin level. In this way resources are better used, as they are considered together in relation to regional needs. Thus, defining whether an impact is or is not acceptable should be done at a catchment level. However this may not always be possible when trans-boundary catchments are involved.

There are numerous management solutions that may be utilized to control or remediate the effects of nutrient enrichment. Because the effects of nutrient inputs on aquatic ecosystems are often difficult to predict, appropriate management actions need to be undertaken to have the desired outcome. According, the environmental managers should approach phosphorus management within an adaptive management framework. The concept of 'adaptive management' was originally developed in the 1970's; it is a formal process for continually improving management policies and practices by learning from their outcomes. Management decisions are a critical step in the framework that links back to the objectives and outcomes of the program via monitoring. Management of eutrophication should include:

i. Short-term management strategies, which primarily focus on operational activities;

ii. Long-term management strategies, which focus on nutrient reduction, flow management, education, monitoring, and research (Banens and Davis 1998).

In practice it is often necessary to apply both strategies simultaneously to ensure some measure of success.

Rast and Holland (1988) provide a basic framework for the development of a eutrophication management strategy. It consists of the following steps:

i. Identify the problem and establish management goals;

ii. Assess the extent of available information about the water body;

iii. Identify available, feasible eutrophication control methods;

iv. Analyze the costs and expected benefits of alternative management strategies;

v. Analyze the adequacy of existing institutional and regulatory frameworks for implementing alternative management strategies;

vi. Select desired control strategies and disseminate plan;

vii. Use institutional mechanisms to minimize future eutrophication problems.

The first two steps of this management strategy are considered in the guidance framework described in this document. Management goals are identified in the first step of the framework, i.e., set goals and objectives, defining the reference or baseline condition. The problem, or potential for problems, are identified when existing conditions are compared to the predefined trigger range. The extent of the problem, sources and available information are all identified within the framework as part of the recommended assessment option. Examples of feasible eutrophication control and management options are provided in this chapter. Cost benefit analysis of these options, analysis for implementation of these management strategies, and selection of appropriate strategies are all managerial decisions that need to be made on a site-by-site basis, and lie outside the scope of this document.

Selected management options that are currently being used, both nationally and internationally, are outlined in this chapter. The options are not limited to those outlined below.

6.2 No Action

Should management decide that the degree of impairment does not warrant the implementation of further management strategies to reduce or manage phosphorus inputs into the system, they can chose to take no further action. At this stage it may be necessary to re-evaluate the established guidelines and trigger ranges for phosphorus. In any event, it is imperative that the system be monitored regularly to ensure no unacceptable degradation occurs to the system.

6.3 Reduction in Phosphorus Inputs/Loading

An important element of any reduction strategy is setting goals, particularly where these are numerical targets, since these allow progress to be measured and the success of the policies adopted to be assessed (Crouzet et al. 1999). Both point and non-point sources of pollution need to be considered when attempting to achieve the set goals.

6.3.1 Point source pollution

The term ‘point source’ refers to any discernible, confined, and discrete conveyance, including but not limited to any pipe, ditch, tunnel conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel, from which pollutants are or may be discharged (EPA 2000a; 2000b). In Europe, methods to reduce point source discharge have primarily focused on urban waste water treatment plants and some key industries (Crouzet et al. 1999). The Urban Waste Water Treatment Directive of the EU has been the key directive for water quality management. The application of best available technology to reduce nutrient inputs from industrial and municipal sources have been implemented, and best environmental practices for reduction of nutrients from diffuse sources are encouraged (especially agriculture).

6.3.2 Non-point sources of pollution

Non-point source pollution generally results from land runoff, precipitation, atmospheric deposition, drainage, seepage, or hydrologic modification. The ultimate causes of non-point source pollution from agricultural lands are excessive fertilizer application, and the use and development of high-density livestock operations. Fertiliser and manure applications can be reduced to match crop needs. Wastes from high-density livestock operations can be managed as a point source of pollution (Carpenter et al. 1998a).

Nutrient management for Lake Simcoe

TP estimates in the Lake Simcoe watershed emphasize the need to keep urban and agricultural sources at the front of remediation efforts. The control of agricultural sources has been addressed in special programs (e.g., the Lake Simcoe Region Conservation Authority's (LSRCA) Landowner Environmental Assistance Program targeting manure management, cattle access to streams and soil erosion from cultivated streams). Urban runoff is still a major component of the total contribution from urban areas. Efforts to contain urban runoff in stormwater ponds have met with limited success.

Forecasts for the area predict continued population growth with concomitant urban development and sewage treatment requirements (Winter et al. 2001). Further efforts to reduce nutrient loads thus focus on these as well as agricultural sources, particularly as these sources are high in nitrogen and dissolved forms of phosphorus that are readily available for plant growth. Stormwater control facilities are common throughout the catchment, particularly for new urban development. These facilities currently focus on P removal, and are usually wet detention ponds estimated to reduce P loading by 50% or constructed wetlands that reduce P loads by up to 87% (LSRCA 1998). Efforts are being made in the catchment to retrofit existing urban areas for stormwater facilities as well as encourage their construction in new developments. Another concern for the future is that cumulative fertilisation in agricultural areas may lead to accelerated TP losses from the aquatic environment after a certain period of time due to a build-up of high soil P contents (Haygarth et al. 1998). No new point-source discharges of phosphorus within the Lake Simcoe basin have been permitted by the OMOE since 1985, and under LSEMS, the limits on phosphorus export from sewage treatment plants have been set by the hydraulic design capacity of each plant. Because the design capacity is now being approached for several sewage treatment facilities in the Lake Simcoe basin, there is no scope for further TP-dependent human population growth (Nicholls 2001).

Nitrogen and phosphorus runoff can be greatly reduced if fertilisers and manure are applied at rates that match the N and P uptake by crops, and when crops are growing rapidly. Source management can significantly reduce concentrations of P runoff entering streams and lakes. The transport of P and N from croplands to surface waters by erosion and runoff may be reduced by maintaining vegetated riparian zones or buffer strips, creating retention ponds, or adopting farming practices such as conservation tillage, terracing, contour tillage, and cover crops.

Approaches to controlling urban non-point source pollution include the creation of retention ponds, wetlands, and greenways as integrated components of storm water management systems; the reduction of impervious areas such as concrete and asphalt paving that enhances runoff; and reduction in erosion, especially from construction sites.

Management measures for various types of non-point source pollution are summarized below (EPA 2000a). These measures should be considered when implementing programs targeting nutrient release into water (EPA 1993):

1. Agricultural runoff

    - Erosion and sediment control
    - Control of waste water and runoff from confined animal facilities
    - Nutrient management planning for cropland
    - Grazing management systems
    - Irrigation water management

2. Urban runoff

    - Control of runoff and erosion from existing and developing areas
    - Construction site runoff and erosion control
    - Construction site chemical control (including fertilizers)
    - Proper design, location, installation, operation, and maintenance of onsite disposal systems
    - Pollution prevention education (e.g., household chemicals, lawn and garden activities, golf courses, pet waste, onsite disposal systems)
    - Planning, siting, and developing roads, highways, and bridges (including runoff management)

3. Silvicultural runoff

    - Streamside management
    - Road construction and management
    - Forest chemical management (including fertilizers)
    - Revegetation
    - Preharvest planning, harvesting management

4. Marinas and recreational boating

    - Design
    - Operation and management
    - Storm water runoff management

- Fish waste management
- Pollution prevention education (e.g., proper boat cleaning, fishwaste disposal, and sewage pump out procedures)

5. Hydromodification (channelisation, dam construction, etc.)

- Minimize changes in sediment supply and pollution delivery rates through careful planning and design
- Erosion and sediment control
- Chemical and pollutant control
- Stabilization and protection of eroding stream banks or shorelines

6. Wetlands and riparian areas

- Protect functions of wetlands and riparian areas through vegetative composition and cover, hydrology of surface and ground water, geochemistry of the substrate, and species composition
- Promote restoration of existing, damaged and destroyed wetland and riparian systems
- Promote the use of engineered vegetated treatment systems

6.4 Remediation and Conservation

6.4.1 Chemical remediation

A number of chemicals can be added to lakes (either lake surface or via a sprinkler system, or dosed to the influent water stream) to flocculate phosphorus. Artificial flocculation has also been used to remove phosphorus from the inputs to lakes.

6.4.2 Reservoirs

The construction of pre-reservoirs without artificial phosphorus stripping may also be used as a method of nutrient removal, of particulate phosphorus by sedimentation and nitrogen by denitrification.

6.4.3 Artificial mixing

In lakes where the surface waters rapidly become nutrient depleted and stratification is maintained during the growing season, abstraction of nutrient-rich hypolimnetic water (released from the outlet) has been used with some success. In contrast, lakes and reservoirs may be artificially mixed as a method of eutrophication control. Artificial mixing serves two purposes: lower waters remain oxygen rich, thereby inhibiting the release of phosphorus from sediments, and phytoplankton are circulated out of the surface waters into deeper waters where light is too low for algae to photosynthesize effectively. In many cases, however, especially where phosphorus near the bottom is high, artificial mixing can actually increase algal standing crop, rather than decrease it.

6.4.4 Biomanipulation

Biomanipulation (the artificial management of fish communities to control algal standing crops) has received an enormous amount of attention in Europe, where numbers of fish feeding on zooplankton are decreased. In Eastern Europe, experimental work has been carried out using fish to directly control algal levels. The introduction of grass carp (*Ctenopharyngodon idella)* has successfully been used to control macrophytes in several countries, for example, in large canals in the USA (Crouzet et al. 1999).

6.4.5 Wetland restoration

Forests and vegetation along riverbanks and shorelines can significantly reduce the flow of non-point nutrients into surface waters. Wetland restoration is a cost-effective method of decreasing non-point N and P pollution.

6.5 Alternative Methods of Phosphorus Control

6.5.1 Phosphorus trading

Under the TP Management program, phosphorus loading quotas would be allocated among those individuals or jurisdictions responsible for TP sources for the lake. The system further allows these TP allocations to be bought and sold among quota holders ("phosphorus trading"). Such approaches can effectively reduce loads to the lake if the net result of the phosphorus "trade" is a lower TP export.

7. IMPLEMENTATION OF THE FRAMEWORK – CASE STUDY 1: LAKE SIMCOE ENVIRONMENTAL MANAGEMENT STRATEGY

7.1 Introduction

Lake Simcoe is the largest lake in southern Ontario, excluding the Laurentian Great Lakes. A three-fold increase from pre-settlement rates in TP loading to the lake since the 1970s have resulted in recruitment failure of native cold-water fishes, as well as excessive growth of aquatic macrophytes and algae (Johnson and Nicholls 1989; Evans et al. 1996). As water-related activities are of major economic importance, there is concern over TP loading to the lake. The Lake Simcoe Environmental Management Strategy (LSEMS) was developed based on recommended remedial measures to reduce phosphorus loading, which were the result of studies that were carried out to identify and measure the sources of phosphorus (LSEMS 1985). Although LSEMS was developed and implemented outside of the guidance framework for phosphorus (outlined in this document), this case study illustrates how existing methods can easily fit into the framework.

7.2 Defining the Baseline/ Reference Condition

In recent decades, natural recruitment of the native cold-water fish lake trout (*Salvelinus namaycush*), lake whitefish (*Coregonus clupeaformis*), and lake herring (*Coregonus artedii*) has declined (Evans et al. 1996). The major concern identified for fish recruitment failure is eutrophication. Excessive growth of aquatic macrophytes and algae have resulted from a three-fold increase in phosphorus load over the past few decades, affecting the hypolimnetic summer habitats of the native cold-water fish. The goal of LSEMS was to “restore a self-sustaining cold-water fishery in Lake Simcoe”. Conditions suitable for this kind of fishery were set as the baseline condition in Lake Simcoe of 9-10 $\mu g{\cdot}L^{-1}$ TP (LSEMS 1985).

7.3 Defining the Trigger Range

According to the modified OECD trophic classifications (Vollenweider and Kerekes 1982) used to set the trigger ranges (Chapter 4, Table 4.3). This classification falls within the ‘oligotrophic’ range of 4-10 $\mu g{\cdot}L^{-1}$ TP.

7.4 Exceedance of the Trigger Range?

The concentration of TP in Lake Simcoe ranged from 20 $\mu g{\cdot}L^{-1}$ TP in the late 1970’s, when the problem was first addressed, to 15 $\mu g{\cdot}L^{-1}$ TP in the mid 1980s; the concentration around the time of release of the initial LSEMS report (Nicholls 1995). These values exceeded the baseline value of 9-10 $\mu g{\cdot}L^{-1}$ TP, which sits on the periphery of the oligotrophic trigger range. The trigger range is thus exceeded, and further assessment is advised. It can be argued that TP may increase by natural processes, and that no allowance is made for this because the baseline is at the upper limit of the trigger range. If the trigger range were set at the next level (mesotrophic, 10-20 $\mu g{\cdot}L^{-1}$ TP), the observed concentration would fall within the set trigger range. However, a second check is incorporated into assessment, which asks the question ‘is there a 50% increase from the baseline?’ (which allows for an increase in TP with minimum impact to the system). In the case of Lake Simcoe, there is. Thus, further assessment is recommended, regardless of which trigger value was initially set.

7.5 Further Modelling

The first evidence of impairment of Lake Simcoe was observed in the early 1970s (Ralston et al. 1975) and, as a result, an initial 5 year investigation of Lake Simcoe was authorized. LSEMS (1985) set out to identify phosphorus loading into the system (Figure 6.1). Pre-1800 loading values were estimated and recent sedimentation and phosphorus loads were obtained by scaling up the pre-1800 rates using ratios of recent to pre-1800 mass sedimentation rates and phosphorus loads (Johnson and Nicholls 1989). The loads presented for 1990-1998 were calculated using extensive monitoring data.

7.6 Management Decision

Based on the outcome of the LSEMS (1985) report, twenty-one recommendations were made, including the need to update and refine phosphorus loading estimates, and develop phosphorus loading strategies (Nicholls 2001). From the initial study (LSEMS 1985), a proposed target of 75 tonnes per year (a 25% reduction from the 100 tonnes/year estimated in the 1980’s) was predicted to generate a lake volume-weighted springtime (mixed water column) TP concentration of 9 to 10 $\mu g \cdot L^{-1}$ (Nicholls 1997).

7.7 Monitoring

Since the start of the Lake Simcoe study there has been ongoing monitoring of the system, and many of the twenty-one recommendations made in the LSEMS report (1985) were addressed during subsequent phases of the LSEMS program. The goal of LSEMS still remains the restoration of a self sustaining cold-water fishery (9-10 $\mu g \cdot L^{-1}$ TP) in Lake Simcoe (Scott et al. 2001).

7.8 Realization of Objectives

It has been estimated that the program reduced the TP loading into the lake by 14 tonnes during the first five years of implementation (1990-1995) (LSEMS1995). Since then, the program has continued to work towards the long-term loading target of 75 tonnes per year.

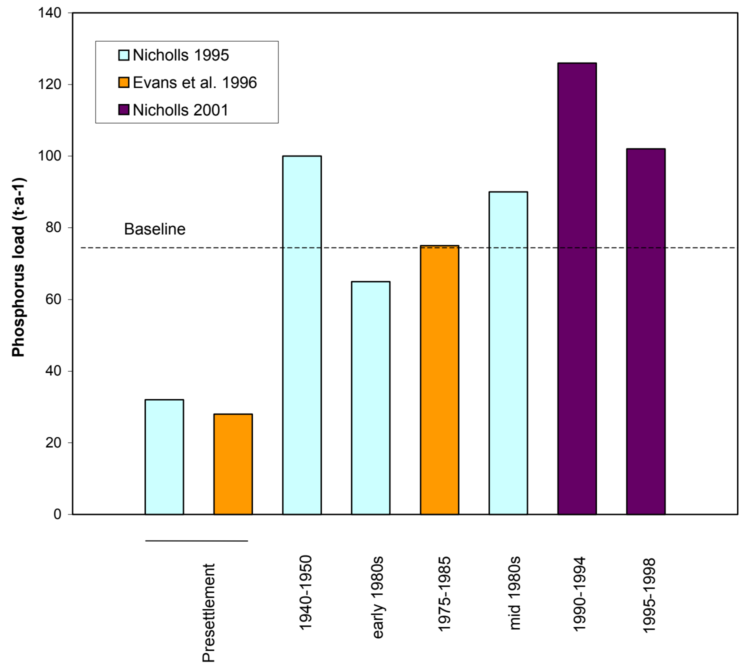

Figure 7.1 Historical phosphorus loading (metric tonnes per year) into Lake Simcoe. Baseline represents a loading of 75 tonnes per year set to achieve a TP concentration of 9 to 10 $\mu g \cdot L^{-1}$ (sources: Nicholls 1995; Evans et al. 1996; Nicholls 2001).

Objectives of the subsequent phases of LSEMS include reducing phosphorus input associated with urban development, and inputs originating from rural agricultural activities (LSEMS 1995). As a result of these programs, a need for improved estimates of the TP loading have been identified. Quantitative hydrological information is required to construct catchment and lake water balances, and there is now an intensive chemical and hydrological monitoring program of the subcatchments of the watershed, as well as measurements of the inputs via deposition. Estimates of TP loads are now considered more accurate as they are based on quantified measurements. A major component of recent studies has been the development of mass balance models to predict the impact of phosphorus input on lake water quality. TP input into the lake remains well above the 75 tonnes per year target set for the lake (Scott et al. 2001; Winter et al. 2001), and excessive growth of inshore filamentous algae continues to be a problem despite improvements in water quality, which have been attributed to zebra mussel filtration.

8. IMPLEMENTATION OF THE FRAMEWORK – CASE STUDY 2: KODIAK LAKE, EKATI DIAMOND MINE, NWT

In 1994, BHP Diamonds Inc., proposed to develop Canada's first diamond mine. The mine would be located in the Lac de Gras area of the NWT, about 300 km north-east of Yellowknife). An Environmental Impact Assessment (EIA) was carried out to identify and assess possible sources of environmental effects of the project. At this time, phosphorus was not considered to be a source of impairment. At the regulatory stage, an effects monitoring program was made a requirement of the Water License. This monitoring identified signs of enrichment (increased nitrates, phosphorus and primary producers). Increases in nitrogen were largely attributed to be a result of blasting activities in the area using NH_4NO_3, and the discharge of treated sewage to Kodiak Lake. These effects were expected to remain as long as the operation exists (Rescan 1999) or as long as treated sewage was added to the lake. In early 1999, all treated sewage was diverted to the Long Lake Containment Facility where the phosphorus in the treated sewage seems to bind to the waste kimberlite tailings. No phosphorus is being detected in the effluent discharged from this facility.

The arctic environment around Ekati is pristine and fragile, and lakes in the study area are characterized by clear, soft, low-nutrient waters, typical of pristine northern aquatic environments. In this chapter, we apply the framework *a priori* to predict whether phosphorus would have been identified as a potential problem.

8.1 Nutrient Loading to Kodiak Lake

Historically, Kodiak Lake has had three primary external point sources of nutrient loading; Koala Lake dewatering, treated sewage effluent (TSE) and the Panda Diversion Channel (PDC). Dewatering of Koala Lake occurred in 1997 and TSE was pumped into Kodiak Lake from 1997 until January, 1999 after which it was transferred into the Long Lake Containment Facility. Treated sewage effluent was reintroduced into Kodiak Lake for 5 days in September 1999, and for one day on September 26, 2000 due to routine maintenance of the processed kimberlite pumping system. The primary external point source of nutrient loading into Kodiak Lake in 2000 was the PDC. A second potential source of nutrients to Kodiak Lake is internal loading. Internal loading can occur through two main mechanisms: physical suspension of lake sediments into the water column, and chemical changes to lakebed sediments that result from low dissolved oxygen concentrations. Physical suspension of sediments is most likely to occur in the open-water season from high energy flow in the PDC and wave action. Low dissolved oxygen concentrations are most likely to occur when the lake is ice covered. Low dissolved oxygen concentrations at the sediment/water interface can cause the efflux of nutrients (ammonium, nitrate, phosphorus) and metals (iron, manganese) from the sediment into the water column. When the sediment/water interface becomes reoxygenated, iron will oxidize, causing the precipitation of iron-phosphate complexes, resulting in most phosphorus being returned to the sediments.

Over 55% of the total water volume that flowed through the PDC in 2000 was discharged in June. As expected, this corresponded to the period when the greatest external loading took place. During June, 44% of the nitrogen loading and 61% of the phosphorus loading occurred. Nitrogen loading was mostly in the form of organic nitrogen (total Kjeldahl nitrogen minus ammonium), followed by nitrate. The source of organic nitrogen is plant and animal matter (alive and dead). Nitrate is possibly originating from ammonium nitrate/fuel oil (ANFO) residues on waste rock used for historical stabilization in the channel, and/or from fugitive dust from blasting activities in nearby Panda Pit. Phosphorus is most likely coming from sediments/soils and decomposing organic matter (terrestrial and aquatic) being transported from upstream waterbodies and the PDC itself. The N to P ratio ranged from 24 to 105, with an annual average of 33. These ratios are significantly higher than the ratio typically required by phytoplankton for optimum growth (N:P should be approximately 7:1 by weight). These values indicate that PDC water had low amounts of phosphorus relative to nitrogen in 2000 and therefore phosphorus would be the limiting nutrient for phytoplankton in the PDC water.

There was no measurable internal loading of phosphorous from sediments during the winter of 1999-2000, with the exception of the final sampling day in May. Bottom waters remained fairly well oxygenated throughout the ice-covered season of 1999-2000. Nitrogen concentrations were elevated during the winter of 1999-2000 compared to open-water season values. However, soon after the ice melted, mid-depth nitrogen concentrations dropped significantly. The winter internal loading of nutrients to Kodiak Lake occurs at a time when biological organisms cannot take full advantage of the elevated concentrations due to cold temperatures and low light. Winter nutrient concentrations return to open-water season levels once the lake becomes ice-free and oxygenated. As such, winter loading in 2000 likely did not contribute significantly to increased summer algal biomass.

8.2 Defining the Baseline/Reference Condition

The baseline studies used to prepare the 1994 EIS (Environmental Impact Statement) (Rescan 1995) and the monitoring conducted from construction and start of operation (1997) to present in both reference lakes (inside and outside of the study area) and Kodiak Lake undertaken by the mine as part of their monitoring programs. Thus, physical, chemical and biological data on this lake exists from the years 1994 onward. Since pre-operation baseline data are available for Kodiak Lake, these values will be used to define the baseline levels in the phosphorus framework. Construction of the mine was initiated in 1997, therefore data from 1994 to 1996 were used to define the baseline. Annual TP in Kodiak Lake ranged from 9.6 to 13.5 $\mu g \cdot L^{-1}$ for the years 1994 to 1996 and the average ice-free mean of 11.1 $\mu g \cdot L^{-1}$ was used as the baseline concentration.

8.3 Defining the Trigger Range

The baseline concentration of 11.1 $\mu g \cdot L^{-1}$ TP defined from the available data puts the lake into the mesotrophic category (10-20 $\mu g \cdot L^{-1}$) of the framework.

Table 8.1 Historical total nitrogen and phosphorus loadings into Kodiak Lake, 1997–2000.

Year	Source[1]	Total Nitrogen Loading		Total Phosphorus Loading	
		($kg \cdot a^{-1}$)	%	($kg \cdot a^{-1}$)	%
1997	KLD	181	4	77	15
	TSE	1306	26	286	56
	PDC	3600	70	148	29
	Total	**5087**	**100**	**511**	**100**
1998	KLD	0	0	0	0
	TSE	2704	74	477	91
	PDC	938	26	48	9
	Total	**3642**	**100**	**525**	**100**
1999	KLD	0	0	0	0
	TSE	124	4	15	10
	PDC	3086	96	134	90
	Total	**3210**	**100**	**149**	**100**
2000	KLD	0	0	0	0
	TSE	0	0	0	0
	PDC	1037	100	31	100
	Total	**1037**	**100**	**31**	**100**

1: KLD = Koala Lake Dewatering; TSE = Treated Sewage Effluent; PDC = Panda Diversion Channel

8.4 Exceedance of the Trigger Range?

There will be two main sources of phosphorus to Kodiak Lake once the mine becomes operational: treated sewage effluent and a diversion channel from a nearby lake. These sources will result in a total loading of 288 $kg \cdot a^{-1}$ to the lake (Rescan 1995). In this case study, the LCM was used to determine the input of phosphorus from natural sources (OMOE 1999A). The sources modelled were atmospheric deposition and runoff from the watershed, which was primarily tundra. The data required to run the model included precipitation, watershed area, runoff volume and land use. A phosphorus coefficient of 5 $kg \cdot km^{-2} \cdot a^{-1}$ was used for tundra on granitic bedrock (P. Dillon 2002, pers. com.). The load from the proposed mine was added to the bseline, and was converted to an overall concentration based on the area of the lake, mean depth and residence time. An increase in the overall ice-free mean to 25.24 $\mu g \cdot L^{-1}$ TP in Kodiak Lake was predicted once the mine becomes operational.

Although the baseline value is at the lower end of the predefined trigger range, the predicted increase in phosphorus concentration in the lake exceeds the mesotrophic range, thus further modelling is recommended.

8.5 Further Modelling

Models, such as the LCM package or Export Coefficient Modelling can be used to estimate the maximum allowable load of phosphorus that would prevent an increase in TP concentration to a level above the trigger range or greater than 50% from the baseline. At this stage, a management decision must be made as to the feasibility of reducing phosphorus loads to the required level.

8.6 Management Decision

The management options available for Kodiak Lake include:

i. Reduction in the overall external loading of phosphorus;

ii. Assess the internal phosphorus loading and draw a plan to reduce it, if found to be a potential source.

8.7 Monitoring

Whichever management option is selected, ongoing monitoring of the lake is imperative to assess the accuracy of predictions made, and the effectiveness of the remediation action.

8.8 Realization of Objectives

The framework was able to successfully identify the problems of TP input to Kodiak Lake not identified in the initial impact assessment. It also provides a means of predicting the effects of different management scenarios and assists in making management decisions.

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APPENDIX A
Conversion equations to estimate total phosphorus from different sampling regimes (source: Clark and Hutchinson 1992)

Ice-free vs. spring turnover	$[TP]_{if} = 0.80\ [TP]_{so} + 2.04$
Ice-free vs. fall turnover	$[TP]_{if} = 1.04\ [TP]_{fo} - 0.15$
Ice-free vs. summer stratified	$[TP]_{if} = 1.04\ [TP]_{ss} - 0.33$
Spring turnover vs. ice-free	$[TP]_{so} = 1.18\ [TP]_{if} - 1.19$
Fall turnover vs. ice-free	$[TP]_{fo} = 0.88\ [TP]_{if} + 0.91$
Summer stratified vs. vs. ice-free	$[TP]_{ss} = 0.96\ [TP]_{if} + 0.33$
Epilimnetic vs. ice-free	$[TP]_{epi} = 0.95\ [TP]_{if} - 1.23$
Epilimnetic vs. summer stratified	$[TP]_{epi} = 0.98\ [TP]_{ss} - 1.44$

if = ice free
so = spring turnover
fo = fall turnover
ss = summer stratified
epi = epilimnetic

APPENDIX B

Equations to calculate optimum sample numbers to use (source: Green 1979)

The equations outlined below can be used to determine the number of samples required to be within 10% or 20% of the long term mean (with 95% confidence).

$$\overline{X} * \left(\frac{\%}{100}\right) = 2\frac{SD}{\sqrt{n}}$$

$\overline{X}$ = long term mean
% = percent difference from the mean required
SD = standard deviation
n = number of samples

For estimating to within 10% of the mean and where '2' represents *t* at 95%:

$$\overline{X} * 0.10 = 2\frac{SD}{\sqrt{n}} \quad \text{or} \quad \frac{\overline{X}}{10} = 2\frac{SD}{\sqrt{n}}$$

Solving for n yields:

$$\sqrt{n} = \left(20\frac{SD}{\overline{X}}\right) \quad \text{or} \quad n = \left(20\frac{SD}{\overline{X}}\right)^2$$

Substituting coefficient of variance for SD/X:

$$n=(20*cv)^2$$

The '20' in this equation can be replaced by '10' where numbers within 20% of the mean are required with a confidence of 95%.

APPENDIX C

Water Quality Index

C-1 Driving variables (Johnes at al. 1994)

A set of 4 driving variables are used to derive the baseline state, these are extrapolated from easily exploited databases:

1. Retention time:

$$\text{Retention time (a)} = \frac{\text{lake volume (m}^3\text{)}}{\text{Catchment area (m}^2\text{) x Net effective precipitation (m}\cdot\text{a}^{-1}\text{)}}$$

(Net effective precipitation = mean annual actual precipitation – mean annual actual evaporation)

2. Conductivity:

Conductivity estimates can be obtained from geological surveys relating run off water and ground water chemistry to local rock type. Composition of standard bicarbonate waters according to conductivity (concentrations given in mg·L^{-1} at 20°C) (Source: Johnes et al., 1994; Rodhe, 1949)

Specific conductance (ìmhos·cm^{-1}) (x 10^6)	Salinity	Ca^{2+}	Mg^{2+}	Na^+	K^+	HCO_3^-	Cl^-	SO_4^{2-}
20	10.5	2.5	0.4	0.7	0.3	4.4	0.7	1.5
40	21.5	5.1	0.9	1.5	0.5	9.0	1.5	3.0
60	32.9	7.9	1.3	2.2	0.8	13.8	2.2	4.7
80	44.7	10.8	1.8	3.0	1.1	18.7	3.0	6.3
100	56.5	13.5	2.3	3.8	1.4	23.7	3.8	8.0
120	68.3	16.3	2.7	4.6	1.7	28.7	4.6	9.7
140	80.3	19.2	3.2	5.4	2.0	33.7	5.4	11.4
160	92.1	22.0	3.7	6.2	2.3	38.6	6.2	13.1
180	104.7	25.0	4.2	7.1	2.6	43.8	7.1	14.9
200	117.3	28.1	4.7	7.9	2.9	49.2	7.9	16.6
220	129.7	31.1	5.2	8.7	3.2	54.4	8.7	18.4
240	142.4	34.1	5.7	9.6	3.5	59.7	9.6	20.2
260	155.1	37.1	6.2	10.4	3.8	65.2	10.4	22.0
280	167.5	40.1	6.7	11.3	4.1	70.2	11.3	23.8
300	180.8	13.3	7.3	12.2	4.4	75.8	12.2	25.6
320	193.9	46.4	7.8	13.1	4.7	81.3	13.1	27.5
340	206.4	49.4	8.3	13.9	5.0	86.6	13.9	29.3
360	219.4	43.8	8.8	14.8	5.4	92.1	14.8	31.2
380	233.1	55.8	9.4	15.7	5.7	97.7	15.7	33.1
400	246.1	59.0	9.9	16.6	6.0	103.2	16.6	34.9

APPENDIX C (continued)
Water Quality Index

3. Inflow of Total Nitrogen

4. Inflow of Total Phosphorus:

Databases for determining inflow TN and TP concentrations include Annual Agricultural Census databases. The area of land use for each of the different categories (temporary and permanent grass, cereal crops, root crops, field crops, orchards, woodland, rough grazing, etc.), numbers of poultry, sheep and goats, cattle, pigs and horses also need to be considered, as well as actual fertiliser use. From the accumulated literature it is possible to estimate rates of input of nitrogen and phosphorus to each of the land use types and thus the proportions of these nutrients leached or excreted to establish the amounts exported to streams.

APPENDIX C (continued)
Water Quality Index

C-2 Derived variables (Johnes at al. 1994)

Regression equations for derived variables

Variable	Equation
Mean annual inflow TP ($\mu g \cdot L^{-1}$)	$[P]_i = [EP]/Q$
Mean annual inflow TN ($\mu g \cdot L^{-1}$)	$[N]_i = [EN]/Q$
Mean annual lake TP ($\mu g \cdot L^{-1}$)	$[P]_ë = 1.55 * ([P]_i/(1+vt(w)))^{0.85}$
Mean annual lake TN ($\mu g \cdot L^{-1}$)	$[N]_ë = 5.34 * ([N]_i/(1+vt(w)))^{0.78}$
Mean winter inflow total oxidized N ($\mu g \cdot L^{-1}$)	$_{MAX}[N]_{wi} = 0.264 + (0.322*[N]_i)$
Max. Chl *a* in P limited lakes ($\mu g \cdot L^{-1}$)	$_{MAX}[chl]P = 0.74 * [P]_ë^{0.89}$
Max. Chl *a* in N limited lakes ($\mu g \cdot L^{-1}$)	$[chl]N = 0.073 * [N]_ë^{0.87}$
Mean annual Secchi disk transparency (m)	$_{MIN}[SEC] = 14.7 * [P]_ë^{-0.39}$
Min. Secchi disk transparency (m)	$[SEC] = 9.33 * [chl]^{-0.51}$
Plant ranking score	$[PRS] = 303 + (1.96*\log[cond]) +(0.117*\log[N]_j)-(0.094*\log[P]_j)$
Mean annual lake calcium ($mg \cdot L^{-1}$)	$[Ca^{2+}] = (47.6 * (\log[cond])) -85.2$
Mean annual pH	$[pH] = 3.99 + (1.47*\log[cond])$
Mean annual lake total alkalinity ($meq \cdot L^{-1}$)	$[Alk] = (0.33*[cond]-16.5)/61$

where:
- Q = mean annual discharge from the catchment to the lake ($m^3 \times 10^6$)
- [EP] = total annual export of phosphorus from catchment area to lake (kg)
- [EN] = total annual export of nitrogen from catchment area to lake (kg)
- t(w) = retention time (days)
- [P]j = mean annual inflow TP concentration ($\mu g \cdot L^{-1}$)
- [N]j = mean annual inflow TN concentration ($\mu g \cdot L^{-1}$)
- [P]ë = mean annual lake TP concentration ($\mu g \cdot L^{-1}$)
- [N]ë = mean annual lake TN concentration ($\mu g \cdot L^{-1}$)
- $_{MAX}$[N]wi = mean winter total oxidized N concentration ($\mu g \cdot L^{-1}$)
- [chl] = maximum annual lake chlorophyll *a* concentration ($\mu g \cdot L^{-1}$)
- $_{MIN}$[SEC] = mean annual secchi disk transparency (m)
- [SEC] = minimum annual secchi disk transparency (m)
- [cond] = mean annual lake conductivity ($\mu S \cdot cm^{-1}$)
- [PRS] = Plant Ranking Score
- [Ca^{2+}] = mean annual lake calcium concentration ($mg \cdot L^{-1}$)
- [pH] = mean annual pH
- [Alk] = mean annual lake total alkalinity ($meq \cdot L^{-1}$)

APPENDIX D

Lakeshore Capacity Model

D-1 Range of lake attributes used to calibrate equations for the Lakeshore Capacity Model (Hutchinson et al. 1991)

List of the attributes used in Lakeshore Capacity Modelling:

Attribute	Units
Total Phosphorus	$\mu g \cdot L^{-1}$
Chlorophyll *a*	$\mu g \cdot L^{-1}$
Total Nitrogen/Total Phosphorus	$\mu g \cdot L^{-1}$
Colour	Hazen Units
Lake area	ha
Emergent Macrophytes	$\mu g \cdot L^{-1}$
NO_3^-	$\mu g \cdot L^{-1}$
NH_4^+	$\mu g \cdot L^{-1}$
Total Kjeldahl Nitrogen	$\mu g \cdot L^{-1}$
Total Organic Nitrogen = TKN – NH_4^+	$\mu g \cdot L^{-1}$
Total Inorganic Nitrogen = TKN - NO_3^-	$\mu g \cdot L^{-1}$
pH	
Dissolved Organic Carbon	$mg \cdot L^{-1}$
Lake Volume	$x10^6 m^3$
Mean Depth	m
Maximum Depth	m
Phytoplankton Biomass	$\mu g \cdot L^{-1}$
Diatom Corrected Phytoplankton Biomass	$\mu g \cdot L^{-1}$
Secchi Depth	m
Area of Hypolimnetic Oxygen Deficit	$mg\ O_2 \cdot m^{-2} \cdot d^{-1}$

APPENDIX D (continued)
Lakeshore Capacity Model

D-2 Lakeshore study trophic status model for TP (Source: Clark and Hutchinson. 1992)

The Ontario Status Model for TP is explained in detail in the final report of the Trophic Status component of the Lakeshore Capacity Study (Dillon et al. 1986):

$$[TP]_{if} = L(1-R_p)/0.956q_s$$

where:

$[TP]_{if}$ = the total phosphorus expressed as an ice-free mean
L = the load of total phosphorus to the lake

$$L = \frac{J_A + J_{PR} + J_N}{A_O}$$

J_A = anthropogenic total phosphorus input
J_{PR} = total phosphorus input from precipitation
J_N = natural total phosphorus input from watershed
A_O = the area of the lake
R_p = the retention of total phosphorus by the lake

$$R_p = 7.2/(7.2+ q_s) \quad \text{(anoxic hypolimnion)}$$
$$R_p = 12.4/(12.4+ q_s) \quad \text{(oxic hypolimnion)}$$

where:

7.2 and 12.4 are the setting velocities and,

q_s = the areal water loading rate ($m \cdot a^{-1}$)

$$q_s = Q/A_O$$

where:

Q = annual lake outflow volume
A_O = lake area

APPENDIX D (continued)
Lakeshore Capacity Model

D–3 The Molot Dillon chlorophyll model (Molot and Dillon 1991; Dillon et al. 1991)

$$Chl_{if} = 0.332\ TP + 0.571$$

Where TP is measured as a long term epilpmnetic average (TP_{epi})

D–4 The Molot et al. oxygen model (Molot et al. 1992)

The end-of-summer oxygen concentrations at each stratum (z) are determined by:

$$\log_{10}O_2(f)_z = 1.83 - 1.91/VSA_z - 7.06/\ O_2(i)_z - 0.0013\ TP^2_{so}$$

where: $O_2(f)$ = the end summer oxygen concentration at depth 'z'
VSA = the volume – sediment surface area ratio at each 2m stratum
$O_2(i)$ = the spring turnover oxygen concentration at depth 'z'
TP_{so} = the spring turnover total phosphorus concentration

Spring turnover oxygen concentrations are measured directly or determined for each stratum (z) by:

$$\log_{10}O_2(i) = 0.99 - 5.74/A_o + 0.64/z$$

(where the maximum distance from shore to shore across the deepest portion of the lake is less than 1.4km)

or:

$$\log_{10}O_2(i) = 1.07 - 6.95/A_o - 0.0043z/MD$$

(where the maximum distance (MD) from shore to shore across the deepest portion of the lake is greater than 1.4km)

A_o = the area of the lake in km.

APPENDIX D (continued)
Lakeshore Capacity Model

D–5 The Lakeshore Capacity Study water clarity (Secchi depth) model (source: Clark and Hutchinson 1992)

Average ice-free Secchi depth is determined by:

$$SD_{if} = -1.26DOC_{AN} – 0.065TP_{SO} – 0.039(\text{chl } a)_{AN} + 10.27$$

or

$$SD_{if} = -1.03DOC_{AN} – 0.061TP_{SO} + 10.14$$

Where: DOC_{AN} = measured average annual DOC
$(\text{chl } a)_{AN}$= average annual chlorophyll *a*
TP_{SO} = spring turnover total phosphorus

APPENDIX E

Export Coefficient Modelling

The modelling procedure is based on that outlined by Johnes (1996) and Johnes et al. (1996), and is outlined in Chapter 7, Figure 7.7. Information is obtained from existing databases on discharge, landuse and catchment characteristics. The rates of nutrient export from each type of land use are selected from the literature. Export coefficients for loading phosphorus from woodland are selected to reflect forest type (Dillon and Kirchner 1975). Chambers and Dale (1997) summarise export coefficients for watersheds throughout North America, primarily draining crops. Soil erosion, and consequently total phosphorus loss in runoff, are greater in row crops than non-row crops (Winter and Duthie 2000). Chambers and Dale (1997) also provide coefficients for pastureland. Reckhow et al. (1980) list urban export coefficients, The input of phosphorus to the watershed via atmospheric deposition is outlined by Kuntz (1980). The percent of total annual rainfall lost to runoff is estimated from field studies. Low export coefficients should be selected to reflect large proportions of low-density and mid-density residential development (Winter and Duthie 2000).

Simple empirical models are constructed in a spreadsheet to predict the total phosphorus (and total nitrogen) load transported to streams as the sum of the export from the watershed:

$$L = \sum_{i=1}^{n} E_i A_i + P$$

Where: L = nutrient load delivered to the stream (kg)
E = export coefficient selected for the nutrient source i ($kg \cdot ha^{-1}$)
A = area of the watershed occupied by landuse type i (kg)
P = input of nutrients from precipitation

P is calculated as: $P=daQ$

where: d = the annual areal deposition rate of P ($kg \cdot ha^{-1}$)
a = area of the watershed (ha)
Q = percentage total annual rainfall lost to runoff

Q is calculated as: $Q = (R/p)*100$

where: R = mean annual runoff (mm)
p = total annual precipitation (mm)

R is calculated as: $R = (F/a)*100$

where: F = mean annual flow volume ($10^6 m^3$)
a = catchment area (km^2)