Literature Review Related to Setting Nutrient Objectives for Lake Winnipeg

May 2006
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A Report Prepared for Manitoba Water Stewardship by

North/South Consultants Inc.

May 2006
Acknowledgements

Manitoba Water Stewardship, in particular Sharon Gurney, Technical Secretariat of the Lake Winnipeg Stewardship Board, is thanked for guidance and support during the preparation of this report.

Many researchers provided references that were important for this the review. In particular, Hedy Kling, Tanya Mayer, Greg McCullough, Alex Salki, and Mike Stainton are gratefully acknowledged for provision of unpublished data, information on current research, and/or manuscripts in press.

Nicole Armstrong, Water Quality Management Section, Manitoba Water Stewardship, is thanked for providing unpublished water quality data.

Members of the Science Committee of the Lake Winnipeg Stewardship Board are gratefully acknowledged for their valuable review of the report.
Report Preparation

The text of this report was prepared by Friederike Schneider-Vieira (sections 1.0, 2.0, 3.0, 6.0 and 7.0) and Megan Cooley (sections 1.0, 2.0, 4.0, 5.0 and 7.0) of North/South Consultants Inc.

Lyle Lockhart, North/South Consultants Inc., provided advice and literature to include in the review.

Garry Scrimgeour, Limnos Aquatic Ecosystems Consulting, provided an extensive listing of literature for inclusion in the review and a detailed review of the draft synthesis report.

Kathleen Martin, North/South Consultants Inc., provided technical support, retrieving and compiling literature and creating the reference database.
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1.0 INTRODUCTION

Nutrient enrichment is one of the most significant and widespread water quality issues in the global environment. In Canada, Chambers et al. (2001) concluded that nutrients released to the environment from human activity are impairing the health of certain ecosystems, contributing to quality of life concerns for Canadians, and, on occasion, endangering human health. In the United States, nutrients have been identified as the leading cause of impairment in lakes and coastal waters and the second leading cause of impairment to rivers and streams (the leading cause is siltation, United States Environmental Protection Agency (USEPA) 1998). In a recent state of the European environment report (European Environmental Agency (EEA) 2005), it was reported that, although phosphorus (P) and nitrate concentrations have declined on the whole in inland freshwater systems, eutrophication was not 'halted'.

Manitoba Water Stewardship (MWS) is currently working towards developing and establishing long-term water quality objectives for nutrients in provincial fresh waters, including Lake Winnipeg. The Lake Winnipeg Stewardship Board (LWSB) in its first interim report identified strategies to reduce nutrient loading to Lake Winnipeg (LWSB 2005). The Board examined a large number of issues and developed recommendations around 32 separate issues, aimed at protecting the health of Lake Winnipeg.

In summer 2005, Manitoba Water Stewardship issued a Request for Proposals that related to addressing the LWSB Recommendation 29.3:

“Manitoba Water Stewardship must continue its work towards completing the Nutrient Management Strategy announced in April 2000. In particular, draft water quality objectives for nutrients in Lake Winnipeg that are based upon ecologically sensitive end-points must be developed as quickly as possible. Following the development of these objectives, broad consultations will need to be undertaken involving the local Lake Winnipeg communities, scientists, contributing sectors within Manitoba, upstream jurisdictions, and others. These long-term water quality objectives will then replace the interim targets identified in the Lake Winnipeg Action Plan”.

A number of draft principles to consider in the development of water quality objectives for nutrients in Lake Winnipeg have been developed by the Lake Winnipeg Stewardship Board’s Science Committee for consideration by Manitoba Water Stewardship. These draft principles are:
1. That water quality objectives for nutrients in Lake Winnipeg need to preserve or restore the important ratio between nitrogen (N) and phosphorus, and need to be reflective of both in-lake concentrations and watershed loadings.

2. That water quality objectives for nutrients need to be developed to consider historical concentrations of nitrogen and phosphorus in Lake Winnipeg.

3. That water quality objectives for nutrients in Lake Winnipeg be developed to ensure the healthy functioning of the Lake Winnipeg ecosystem.

4. That water quality objectives for nutrients in Lake Winnipeg need to minimize the duration, frequency, and intensity of blue-green cyanobacteria blooms including the need to minimize the production of cyanobacterial toxins harmful to aquatic life, recreation, and drinking water.

5. That water quality objectives for nutrients in Lake Winnipeg need to minimize the duration, frequency, and intensity of blooms of other forms of algae including those leading to fouling of commercial and subsistence fishing nets or that otherwise interfere with the successful harvest of fish.

6. That water quality objectives for nutrients in Lake Winnipeg need to be developed to ensure that an optimum balance is achieved between nutrient enrichment, productivity of the commercial and subsistence fishery, and subsequent economic return to communities, while protecting the lake’s ecosystem health and recreational uses.

7. That water quality objectives for nutrients in Lake Winnipeg need to be protective of the downstream environment in the Nelson River and Hudson Bay.

8. That water quality objectives for nutrients in Lake Winnipeg need to recognize water quality objectives established for the contributing watersheds, and that water quality objectives for nutrients established in the contributing watersheds need to recognize Lake Winnipeg.

9. That water quality objectives for nutrients in Lake Winnipeg need to be considerate of the social and economic implications of implementation and compliance.

Towards developing these nutrient objectives, Manitoba Water Stewardship contracted North/South Consultants Inc. to conduct the following tasks:

1. Identify and compile relevant literature to provide the necessary background information to assist Manitoba Water Stewardship in setting long-term ecologically sensitive nutrient objectives for Lake Winnipeg.
2. Prepare an annotated bibliography of selected relevant nutrient-related research conducted on Lake Winnipeg, prairie ecosystems, marine waters, and other systems.

3. Prepare an overview or interpretation of the above mentioned research.

The final product, as contained in this submission, consists of the following:

- a listing of approximately 1000 key references, including citation and abstract (where available) in electronic format (separate file). This listing includes all references suggested by researchers contacted as part of the literature gathering process (see Section 2.0);
- an annotated bibliography of approximately 100 reports, including citation and abstract (where available) (Appendix I); and
- an overview of the collective research (this report) which synthesizes relevant information in approximately 300 references.

As the review of literature available with respect to Lake Winnipeg progressed, it became apparent that analysis of data gathered with respect to the issue of nutrients and eutrophication in the lake had not progressed sufficiently to allow identification of key issues on the lake. Likewise, an analysis of data gathered in the upstream and downstream watersheds with the intent to identify key issues with respect to management has not been produced in a published report. Therefore, the review of the global literature could not be focussed on issues in Manitoba waters to the extent anticipated at the initiation of the project. Rather, as discussed in Section 2.0, information on Lake Winnipeg was compiled to provide an overview of what is known regarding physical, chemical and biological conditions in the lake that are relevant to the issue of nutrient management, and the global literature was synthesized to provide a cross section of issues that should be considered in the analysis of Lake Winnipeg and other Manitoba waters (the review emphasized, but was not limited to temperate, prairie ecosystems).

Given the volume of literature pertaining to eutrophication, the review was not meant to, nor could within the practical constraints of the current contract, be comprehensive; rather, it was selective to address the major issues, particularly those highlighted in recent research. The annotated bibliography consists of complete citations and abstracts of approximately 100 publications selected for their relevance. In most instances the abstract summarized the key information as referenced in this report; where it did not, or no abstract was available, further annotations are provided.
2.0 DESCRIPTION OF INFORMATION SOURCES

The body of literature related to nutrients and eutrophication in the aquatic environment is extremely large (for example, the Aquatic Science and Fisheries Abstracts list 19,035 references for the term eutrophication). Therefore, the literature review was scoped to focus on reports directly related to Lake Winnipeg as well as the most recent literature related to eutrophication that would provide background information relevant to setting nutrient criteria for Lake Winnipeg and its watershed within Manitoba.

As much of the recent research related to Lake Winnipeg is not yet available in published reports, researchers currently involved in studies on the lake were contacted to obtain an overview of work being conducted.

The literature search was conducted as follows:

1. Literature search of electronic databases (Aquatic Fisheries and Science Abstracts and Environmental Sciences and Pollution Management). The search was structured to include terms related to nutrients (i.e., eutrophication, nitrogen, phosphorus, and nutrients) and geographic area (e.g., Lake Winnipeg). The initial selection was comprised of publications within the last two years and then supplemented by references within these papers. The majority of literature available for Lake Winnipeg was included where it consisted of original research.

2. Search of in-house library. North/South Consultants has a large in-house library related to the effects of nutrients in the aquatic environment and the development of nutrient criteria.

3. Search of relevant government internet sites including Environment Canada, Manitoba Water Stewardship, and Alberta Environment. Other sites were also searched, in particular where agencies have been active in developing nutrient criteria (i.e., United States Environmental Protection Agency and the European Environment Agency).

4. Key person contacts. The Technical Secretariat provided a list of researchers to contact. The primary purpose was to obtain a description of current studies from researchers conducting work on Lake Winnipeg where such studies have not yet been published. Researchers also provided lists of literature recommended for inclusion in the annotated bibliography.
Citations for all the reports identified above were examined and copies of relevant reports were obtained; of these approximately 100 were identified for inclusion in the annotated bibliography. Selection of these reports was based on:

- inclusion of scientific data on the current and historic conditions describing the physical, chemical, and biological conditions in Lake Winnipeg;

- cross-section of major issues, particularly those highlighted in recent research, that could be relevant to Lake Winnipeg and its upstream and downstream watershed, including the marine ecosystem. Issues included topics such as nitrogen and phosphorus limitation, the role of nitrogen: phosphorus ratios, internal loading, alternate stable states, and trophic cascades/top down effects. Syntheses and reviews as well as recent reports were emphasized as these tended to encompass key findings of earlier work; and

- reports describing the development and application of nutrient criteria in relevant systems.

A draft table of contents of the report was provided to the Technical Secretariat in December 2005 and a draft list of reports to be included in the annotated bibliography was provided in January 2006.
3.0 LAKE WINNIPEG

Lake Winnipeg, with a surface area of about 23,750 km², covers about 3.7% of the surface area of the Province of Manitoba, Figure 1. It supports a large commercial fishery, subsistence fishery, recreation, lakeshore communities (e.g. Gimli, First Nation communities), and is an important component of Manitoba's hydroelectric system.

The lake's drainage basin, at nearly 1,000,000 km² in size, is the second largest watershed in Canada, including parts of four provinces and four American states. The basin is home to six million people and 17 million livestock, and includes 55 million ha of agricultural land (LWIC 2005). The contamination of Lake Winnipeg has been the subject of intermittent study over the past four decades: though there have been issues related to other contaminants (e.g., the commercial fishery was closed one year in the early 1970s due to mercury levels) the primary focus has always been related to eutrophication, as a result of organic loading in the 1960s and more recently, enrichment by nitrogen and phosphorus.

The Province of Manitoba, recognizing the impact of nutrients, not only on Lake Winnipeg, but on all provincial waters, is developing and implementing a Nutrient Management Strategy, a draft of which was released in 2000. In the late 1990s, satellite images showing large areas of the north basin of the lake covered by surface algal blooms focussed public concern more specifically on Lake Winnipeg. The Province of Manitoba implemented the Lake Winnipeg Action Plan, which included establishment of the Lake Winnipeg Stewardship Board in 2003, to assist in addressing the issue of nutrient loading to Lake Winnipeg. The Stewardship Board issued its first interim report in early 2005, followed by a public discussion document released in fall 2005 (LWSB 2005). The Lake Winnipeg Implementation Committee (LWIC), which had the mandate to report on the environmental and socio-economic aspects of Lake Winnipeg, as well as provide advice on steps to be taken to improve the health of Lake Winnipeg, issued a report in late 2005, supported in early 2006, by a technical annex (LWIC 2006).

The intent of the following section is not to repeat information in the aforementioned documents, but to summarize technical information that is available for Lake Winnipeg relevant to the past and current status of the lake's ecosystem, trends in conditions in the lake, and information pertaining to nutrients in aquatic ecosystems to provide necessary background information required to set objectives for the lake and other provincial waters. As discussed below, analysis of the most recent data collected on the lake is on-going and results to the extent that they are publicly available have been included in this report; however, in recognition of the evolving nature of the understanding derived from these analyses, preliminary interpretations have generally not been included in this document. This report is not meant to be an exhaustive
Figure 1. Lake Winnipeg and watershed.
account of all recent and on-going research on the lake, as the primary focus is on published reports; rather a summary of work related to the issue of nutrients in Lake Winnipeg is provided based on material that has been provided to the authors by researchers contacted during the course of this review.

3.1 SYNOPSIS OF STUDIES

This section provides a brief overview of the major scientific studies that have occurred on Lake Winnipeg. Findings of these studies relevant to the subject of nutrients are provided in subsequent sections.

3.1.1 Pre 1990

The Geological Survey of Canada conducted surveys of the shores of Lake Winnipeg in the 1890s (Dowling 1900, and Tyrell and Dowling 1900 In Thorleifson et al. 2000). Surveys of the waters and biota were conducted in the 1920s and 1930s by Bajkov (1930, 1934) and Neave (1932, 1933, 1934). The work by Bajkov and Neave has served as the baseline for comparisons by later investigators.

Much of the scientific study in the 1950s and 1960s focused on issues pertaining to the commercial fishery (e.g., Hewson 1959, Rybicki and Doan 1966, Howard 1974). Investigations related to the fishery have been on-going (e.g., Davidoff 1978, Schlick 1978, Kristofferson 1985, Lysack 1995).

The Province of Manitoba initiated a pollution investigation of the south basin in 1962 and this continued to 1970. Work consisted of surveys in September and March of 16-70 stations set in a grid across the south basin and included limnology, water quality, and benthic invertebrate sampling; additional work was conducted during the summer months in some years (Stone and Cober 1965, Cober 1966, Crowe 1969, 1973a, 1973b, 1975). Rybicki (1966) reported on limnological, water quality, and benthic invertebrate surveys of the north basin in 1963-1964. Rybicki's stations were revisited in summer 1968 (Pollard 1973) and 1974 (Kristofferson et al. 1975, Derksen and Hangasjarvi 1979).

A comprehensive lake-wide survey of water quality, phytoplankton, zooplankton and benthic invertebrates using the coast guard vessel the Canadian Government Ship Bradbury was conducted during the open water season of 1969 as well as under ice cover by scientists from the Freshwater Institute (FWI). Published results from the 1969 survey included: an analysis of nutrient supply to the lake (Brunskill 1973); a record of temperature, oxygen, conductance and dissolved major elements (Brunskill et al. 1979a); light attenuation in the lake waters (Brunskill et al. 1979b); patterns of sedimentation (Brunskill and Graham 1979); lake morphology,
hydrology and watershed data (Brunskill et al. 1980); and the crustacean planktonic community (Patalas 1981; Patalas and Salki 1992). Later reports provided information on the biota collected during the 1969 survey, including comparisons to studies conducted by Bajkov and Neave in the late 1920s and early 1930s including analyses of changes in Trichoptera (Flannagan and Cobb 1981) and *Hexagenia* (Flannagan and Cobb 1984). Data reports were also published on: aquatic insects (Chang et al. 1993, 1994); oligochaetes (Chang et al. 1992); benthic crustaceans (Flannagan and Cobb 1994); molluscs (Flannagan and Cobb 1991); and crustacean plankton (Salki and Patalas 1992). Unpublished data on nutrients and phytoplankton from the 1969 survey are available (Brunskill 1969, unpublished data in a draft manuscript).

There was little work not related to the fishery conducted on the lake from the mid 1970s to the mid 1990s. Limited water quality work in the south basin was conducted in 1973 (Manitoba Department of Mines, Resources and Environmental Management 1973) and 1980-82 (Hughes 1983). Healey and Hendzel (1980) reported on a comparative analysis of nutrient samples involving samples from several lakes, including the south basin of Lake Winnipeg, based on sampling conducted in 1974.

### 3.1.2 Post 1990

The Lake Winnipeg Project was launched by the Geological Survey of Canada and the Manitoba Geological Survey in 1994, with the primary objective to gather geophysical information required in support management of issues for the lake such as shoreline erosion. The study was expanded to include limnological and biological sampling by scientists from the Freshwater Institute and several university programs. A four week cruise of the Canadian Coast Guard Ship *Namao* in August 1994 was followed by a similar effort in 1996, shoreline investigations in 1997 and a one day cruise in 1999. Reports on the surveys are provided in a series of open files of the Geological Survey of Canada (Todd et al. 1996, 2000 and Simpson et al. 2003); some results have also been published in peer reviewed journals (e.g., Kling 1998, Mayer et al. 2006). The 1994 cruise included sediment, biological and water sampling, geophysical surveys, and coring (Todd et al. 1996). The 1996 cruise focused on filling information gaps.

In response to the 1997 Red River flood, the International Joint Commission formed the International Red River Basin Task Force to study aspects of the flood and make recommendations regarding future floods. A cruise was conducted in the south basin of Lake Winnipeg in 1998 to assess effects of the flood to contaminant and nutrient levels as well as gather information that could assist in determining the frequency of large floods in the basin (Stewart et al. 2000, 2003).
In 1998, the Province of Manitoba established the Lake Winnipeg Shoreline Erosion Advisory Group to review specific issues related to the erosion of Lake Winnipeg shorelines; the work of this group focused on the south basin. A final report was prepared (Lake Winnipeg Shoreline Erosion Advisory Group 2000) and one of the technical reports provided analyses of lake inflows, outflow, and levels, and shoreline erosion (Baird and Stantec 2000).

The Lake Winnipeg Research Consortium Inc. (LWRC) was formed in 1998 to facilitate research on Lake Winnipeg with membership from universities, public corporations, First Nations, commercial fishers, other industry members, citizens, and municipal, provincial and federal governments. The LWRC's activities include: (i) holding an annual forum to discuss and coordinate research; (ii) providing the retired Canadian Coast Guard Ship Namao as a research platform; and (iii) providing educational opportunities for students.

A pilot study in August 1999 found evidence of eutrophication in the north basin (Salki and Stainton 2004). In 2002, agencies participating in the LWRC initiated a 3 year program designed to provide the information on the physical, chemical and biological aspects of this ecosystem (Salki and Stainton 2004). Studies were designed to address potential effects of eutrophication and climate change, including: a study to examine the impact of climate warming on the Lake Winnipeg fishery; initiatives to use satellite imagery to estimate surface chlorophyll a (chl a) and turbidity in water; development of a carbon (C) budget; evaluation of the carbon cycle; and measurements of nitrogen fixation rates and algal physiology.

Open-water surveys were conducted in 2002, 2003, and 2004 in early summer, mid to late summer, and fall (Salki and Stainton 2004), and in fall 2005 (A. Salki, pers. com.). Sixty five stations located throughout the north and south basins were sampled (with the exception of areas too shallow to reach with the vessel). The program varied between years, in part as a result of the previous years' findings. Physical (including temperature, colour, light extinction), chemical (including ions, total suspended solids (TSS), particulate organic carbon, chlorophyll a, turbidity, conductivity, nutrients, oxygen) and biological (phytoplankton, zooplankton, benthos) samples were collected. Between stations, continuous measurements of specific water quality parameters (pCO2, conductivity, turbidity) were made. Small fish and samples of net plankton were also collected. Samples for the analysis of contaminants were collected in 2002. In 2003, vertical profiles of temperature, oxygen, CO2, phosphorus, nitrogen, chlorophyll a, and suspended sediments (turbidity) were measured at eight stations. Environment Canada meteorological buoys were deployed in the central areas of the north and south basins and in the narrows; additional instrumentation included equipment to log pCO2, temperature data, and oxygen.
The data collected by agencies participating in the LWRC are in various stages of analysis; results of initial assessments have been provided in various forums including at scientific and other public presentations and in non-technical reports. However, analysis is on-going and results have not yet been published in either technical reports or peer-reviewed journals; several researchers provided unpublished data that is referenced in the following sections.

3.2 THE PHYSICAL ENVIRONMENT

3.2.1 Drainage Basin

Lake Winnipeg is a remnant of glacial Lake Agassiz and is situated on the boundary between the Precambrian Shield to the east and sedimentary strata to the west and south (Brunskill 1973).

The watershed extends westward through Saskatchewan and Alberta to the Rocky mountains, east into Ontario and south into North Dakota and Minnesota and small portions of South Dakota and Montana. Lake Winnipeg is unusual among the world's largest lakes, in that the ratio of the drainage basin to lake area is approximately 40:1, which is considerably larger than the world's other great lakes (LWIC 2005).

Four major river systems discharge to the lake: the Winnipeg River (mean monthly flow of 771 m$^3$/s); the Saskatchewan River (mean monthly flow of 667 m$^3$/s); Red-Assiniboine rivers (mean monthly flow of 159 m$^3$/s); and the Dauphin River (mean monthly flow of 57 m$^3$/s (Lewis and Todd 1996). Numerous smaller rivers enter the lake, in particular along the eastern shore. The eastern watershed, including the Winnipeg River, drains the soils, muskegs and boreal forests overlying the igneous bedrock of the Precambrian Shield (Brunskill et al. 1980). The southern, western and north-western portions of the drainage basin are within the Paleozoic and Mesozoic sedimentary terrains of the prairie provinces and north central United States of America (Burbidge et al. 2000). These areas were originally prairies to the south and mixed forests to the west and northwest; the southern areas now support extensive agriculture and several large cities (Brunskill et al. 1980).

Terrestrial drainage areas and runoff of the tributaries as calculated by Brunskill et al. (1980) are provided in Table 1.
Table 1. Terrestrial drainage areas of rivers tributary to Lake Winnipeg, and annual runoff for the watersheds (Brunskill et al. 1980).

<table>
<thead>
<tr>
<th>Basin</th>
<th>Tributary</th>
<th>Geology of drainage area</th>
<th>Drainage area (km²)</th>
<th>Runoff (cm)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>South</td>
<td>Winnipeg River</td>
<td>Precambrian Shield</td>
<td>126,400</td>
<td>31</td>
</tr>
<tr>
<td></td>
<td>Red River</td>
<td>85% Sedimentary</td>
<td>287,500</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Other</td>
<td>-</td>
<td>16,700</td>
<td>-</td>
</tr>
<tr>
<td>North / Narrows</td>
<td>East side rivers</td>
<td>Precambrian Shield</td>
<td>41,050</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Saskatchewan River</td>
<td>Sedimentary</td>
<td>340,000</td>
<td>6.4</td>
</tr>
<tr>
<td></td>
<td>Dauphin River</td>
<td>Sedimentary</td>
<td>80,000</td>
<td>3.7</td>
</tr>
<tr>
<td></td>
<td>Other</td>
<td>Sedimentary</td>
<td>61,200</td>
<td>-</td>
</tr>
</tbody>
</table>

* Runoff = annual river discharge (1969-1974)/drainage area

As noted by Brunskill et al. (1980) and discussed further in Section 3.3.1.5, the characteristics of these watersheds result in very different water quality and nutrient loads in tributaries entering the lake.

Both the Saskatchewan and Dauphin rivers have large lakes a short distance upstream of their entry into Lake Winnipeg; in contrast, the other systems do not. The Saskatchewan River has formed a large delta in Cedar Lake, upstream of Lake Winnipeg. The delta of the Red River is Netley Marsh, formed along the southernmost shore of Lake Winnipeg. No delta has formed at the mouth of the Winnipeg River or the smaller rivers along the eastern shore, presumably because of the smaller sediment load in these rivers, though further upstream there are extensive bogs and areas of rooted aquatic vegetation within the watersheds.

Human alteration of portions of the watershed began in the late 1800s. Starting around 1892, water control projects in Ontario, Manitoba, Saskatchewan, Alberta, Minnesota and North Dakota affected natural inflows to the lake (Baird and Stantec 2000). Hydroelectric generating stations were first constructed along the Winnipeg River in Manitoba over 80 years ago. Drainage from Lake Manitoba into the Dauphin River has been regulated by the Fairford dam since the early 1960s. Construction of the Grand Rapids Generating Station, which impounded a short section of the Saskatchewan River below Cedar Lake and flooded Cedar Lake was completed in 1968. In 1976, the outflow of Lake Winnipeg was regulated by the construction of the Jenpeg Generating Station just upstream of Cross Lake on the west channel of the Nelson River (flows along the much smaller eastern channel are unregulated). Construction of Jenpeg, in conjunction with works to increase the outflow capacity of the lake, allowed water to be retained through the open-water season to increase the supply down the river during the winter,
resulting in a relative decrease in summer flows and increase in winter flows out of the lake (LWIC 2005).

Brunskill (1973) presented an analysis of changes in land use for the sedimentary portions of the watershed: changes in human population, area under cultivation, and number of livestock were gradual for the period 1945-1970 (data was not available for all parameters for the full period), but the use of fertilizers increased exponentially.

3.2.2 Hydrology

3.2.2.1 Water Levels and Flows

The Lake Winnipeg Shoreline Erosion Advisory Group (2000) and one of its associated technical reports (Baird and Stantec 2000) represents the most complete published report, describing mean monthly lake levels, inflows and outflows for Lake Winnipeg prior to and after regulation, located during this review. Lake Winnipeg water levels for the period 1913-2005 are provided in Figure 2.

Baird and Stantec (2000) reported that the long-term average lake level for the period 1914-1976 and 1977-1998 were very similar, both being about 217.6 m (713.5 ft). Lake levels vary seasonally, with highest average levels occurring in July and August and lowest levels in January and February. Regulation had little effect on average monthly levels, which generally differ by less than 0.03 m (0.1 ft) pre and post-regulation. Baird and Stantec (2000) noted that the average annual fluctuation in lake level is only 0.30 m (1 ft); however in any given year the difference between the highest monthly mean and the lowest monthly mean typically ranges between 0.3-0.6 m (1-2 ft) and had been as great as 1.2 m (4 ft) (prior to regulation) and as little as 0.15 m (0.5 ft). Manitoba Hydro's licence for regulation requires that the "wind-eliminated" lake level be maintained between elevations 216.9 m (711 ft) and 218.1 m (715 ft) in so far as possible; Figure 2 clearly illustrates the greatest effect of regulation on lake levels, which has been to eliminate both extreme high and low levels (prior to regulation, lake levels ranged from less than 216.5 m (710 ft) to about 219.0 m (718 ft)).

Lake Winnipeg also undergoes long-term (i.e., months to seasons to years) changes in levels that result from variations in the amount of precipitation, evaporation, inflow to and outflow from the lake, including the effects of regulation; over the scale of centuries, isostatic rebound is raising the outlet such that levels in the south basin are gradually increasing at an estimated rate of 0.2 m/century (Baird and Stantec 2000).
Figure 2. Lake Winnipeg mean monthly levels (Manitoba Hydro 2006).
Baird and Stantec (2000) noted that comparison of the inflow and outflow data indicates that there is also a decadal pattern of increased vs. decreased flow. Based on a 5 year moving average, inflows were above average during the 50s, the mid 60s to mid 70s, and the mid 90s. In particular, water levels on Lake Winnipeg were exceptionally high in the 12 years prior to regulation in 1976. Summer outflows follow the same pattern as inflows, although following regulation they were below the long term average until the mid 1990s. Since regulation, winter outflows have not followed the pattern of inflow, but have generally been above the long term average. This change in seasonal distribution flows, with larger flows now occurring during the winter, rather than summer, would also have altered seasonal water residence times in the lake.

No published report providing a complete hydrologic model of Lake Winnipeg, including estimation of all inputs from rivers, direct runoff, groundwater\(^1\) and precipitation, and losses through outflow and evaporation, was located during the course of this review.

### 3.2.2.2 Ice

Freeze-up typically occurs in mid November with lake ice at Gimli reaching a maximum thickness of about 1 m before breakup in mid April (Forbes 2000). Pressure ridges develop at consistent locations within the lake and pressure ridge keels interact with the lake bottom to form extensive scours. Ice push at the shores has been observed to cause minor scour.

### 3.2.2.3 Wind and Wave Generated Currents

Water levels on Lake Winnipeg may be affected by the influence of strong winds and, to a lesser extent, by changes in atmospheric pressure. The increases above normal water level are referred to as "wind-setup" or "storm surges". This topic was extensively investigated in the south basin as part of the Lake Winnipeg Shoreline Advisory Board (2000). Wind-setup can increase lake levels at a given location by 0.6-1.2 m (2-4 ft). Storm surges are generally greater from September to November compared to May to August (Baird and Stantec 2000). The frequency of wind setup has varied over the past decades: during the 18 year period from 1974 to 1991 there were 8 wind setup events greater than or equal to 0.6 m (2 ft); in the 8 year period from 1992-1999 there were also 8 events greater than or equal to 0.6 m (2 ft). From 1974 to 1991, no wind setups greater than 0.8 m (2.5 ft) occurred when wind-eliminated lake levels were above 217.6 m (713.5 ft); during the period from 1993 to 1999, 2 wind set up events greater than 0.8 m (2.5 ft) occurred when lake levels were above 217.8 m (714 ft).

\(^{1}\) It is not known whether groundwater provides a significant input to this lake; groundwater inputs upstream in the Dauphin drainage are sufficient to cause measurable changes in water quality.
Wind set up creates seiches, which result in continuing oscillations in lake levels after the wind event has ended, and strong reversing flows between the north and south basins (McCullough 2001).

Wind and wave generated currents are important for the resuspension and transfer of sediments in the lake (Forbes 2000, McCullough et al. 2001).

### 3.2.3 Morphology

Basic morphological information for the lake was calculated by Brunskill et al. (1980). Data were generated from Canadian Hydrographic Service Charts and the authors' 1969 bathymetric data; as these charts have several different datum levels, they were all adjusted to a common datum. Results are for an elevation of 218.1 m ASL (approximately 0.5 m above the mean lake elevation); the authors noted that over the period 1913 to 1975 the minimum lake level was 216.5 m and the maximum was 218.4 m, and that this range would result in a variation of about 17% in volume estimates. Results are provided in Table 2.

Table 2. Morphometric data for Lake Winnipeg (Brunskill et al. 1980).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>North Basin</th>
<th>Narrows</th>
<th>South Basin</th>
<th>Whole Lake</th>
</tr>
</thead>
<tbody>
<tr>
<td>Area (km²)</td>
<td>17,520</td>
<td>3,450</td>
<td>2,780</td>
<td>23,750</td>
</tr>
<tr>
<td>Volume (km³)</td>
<td>232.4</td>
<td>24.6</td>
<td>27.0</td>
<td>284</td>
</tr>
<tr>
<td>Mean depth (m)</td>
<td>13.3</td>
<td>7.2</td>
<td>9.7</td>
<td>12</td>
</tr>
<tr>
<td>Maximum depth (m)</td>
<td>19</td>
<td>36*</td>
<td>14</td>
<td>36</td>
</tr>
<tr>
<td>Maximum length (km)</td>
<td>232</td>
<td>143</td>
<td>93</td>
<td>436</td>
</tr>
<tr>
<td>Maximum breadth (km)</td>
<td>111</td>
<td>30</td>
<td>46</td>
<td>111</td>
</tr>
<tr>
<td>Shoreline (km)</td>
<td>761</td>
<td>640</td>
<td>349</td>
<td>1,750</td>
</tr>
</tbody>
</table>

* This deep hole is a v-shaped channel < 1 km in length and is an insignificant volume in relation to the rest of the lake.

Based on hydrological and morphological conditions in 1969-1974 (a period of exceptionally high inflows and lake levels), Brunskill et al. (1980) estimated a theoretical water renewal time of 0.4 – 0.8 years for the south basin and 2.9 – 4.3 years for the whole lake.

### 3.2.4 Water Temperature and Thermal Stratification

The north and south basins differ in their annual temperature cycle. Measurements in 1969 indicated that the north basin warmed more slowly in spring and cooled more slowly in fall, while the south basin was frequently near maximum daily air temperature in spring and summer and cooled abruptly in fall (Brunskill et al., 1979a). This difference was due to both the greater
volume and depth of the north basin, as well as the influence of the tributaries, which warm and cool more rapidly than the lake. In spring and sometimes in summer, inflow from the tributaries form a plume of warmer water over the underlying lake water. In 1969, temperatures measured at 1 m below the surface in June in the south basin ranged from 10-14 °C while, in the north basin the water was approximately 4 °C cooler (Brunskill et al. 1979a). By mid summer, the south basin was 20-22 °C, while the north basin ranged from 18-20 °C.

Preliminary results of a 2002 study including compilation of mean monthly water temperatures for the south basin shows that mean water temperature did not show significant changes over the period 1909-2000 except for August, where water appears to have warmed by 1.4 °C (McCullough 2004).

In general, Lake Winnipeg is reported to not thermally stratify, or that any stratification is weak and transient, as the lake is relatively shallow for its depth and calm periods seldom last long (e.g., Brunskill et al. 1979a). Review of published reports regarding thermal stratification in Lake Winnipeg indicates that, although the lake does not become strongly stratified for prolonged periods, the lake can be either unstratified or weakly stratified, as noted below:

- two of five profiles collected by Bajkov in mid summer in 1927 and 1928 (Bajkov 1930) exhibited some stratification (temperature change of 4 °C over 2 – 5 m), while at other locations there was little or no temperature difference from surface to bottom;
- Rybicki (1966) found no evidence of thermal stratification along transects in the north basin during summer 1963 and 1964 (< 1 °C difference top to bottom);
- Pollard (1973) reported that maximum surface to bottom temperature differences were less than 2 °C at stations in the north basin in June and July 1968;
- Brunskill et al. (1979a) measured temperature at 1 m below the surface and then at 5 m intervals at many of the sites surveyed in 1969. In mid summer, a 4-5 °C difference was observed between the surface and bottom at many of the sites and in one deeper area the difference was 8 °C (insufficient measurements along the profiles were obtained to determine whether there were sharp gradients in temperature);
- Kristofferson et al. (1975) reported a difference of up to 10 °C (average 6 °C) between surface and bottom temperatures at sites in the north basin in July 1974. After a large storm, the water column appeared to be mixed such that the temperature difference was less than 4 °C (average 2 °C) at the remainder of the sites surveyed; and
• Stainton (2005) reported thermal stratification in 2003 in deeper areas of the north basin (16 m isopleth) with the bottom 3 m approximately 7 °C cooler than the overlying water mass, which was of uniform temperature.

3.2.5 Shorelines, Erosion and Sedimentation

3.2.5.1 Lake Shorelines

Forbes (2000) described the shorelines of Lake Winnipeg and existing depositional and erosional features. Parts of the eastern shore are characterized by Precambrian shield rocks with little surface sediment while other areas have developed extensive sandy beaches with coastal dunes. Along the eastern and northern shores of the north basin, extensive areas of fine grained sediment are present and form prominent bluffs. The Long Point peninsula and George Island are both associated with a glacial moraine deposits. Along the south eastern shore, glacial and other deposits mask the underlying bedrock. Silty clays of Lake Agassiz form much of the western shoreline of the south basin. Lowland areas of the lake are often backed by peat bogs, and at the shore there may be a low peat scarp or a thin barrier. In many cases Lake Winnipeg beaches are confined to narrow wedges at the base of cliffs. At some locations, barriers or coastal dunes have developed.

Netley Marsh, the delta of the Red River, is at the southern end of Lake Winnipeg and separated from the lake by a thin sandy barrier which is slowly migrating southward into the marsh as water levels in the south basin rise. The marsh consists of a 26,000 ha complex of shallow lakes, lagoons and channels of the Red River (Grosshans et al. 2004).

3.2.5.2 Erosion and Resuspension

Shoreline recession is widespread in the south basin of Lake Winnipeg. Studies of shoreline erosion have found that it is a natural phenomenon, driven in part by the long-term differential isostatic uplift of the northern lake outlet and the southward tilting of the basin (Forbes et al. 2002). Shore profiles along the south western shore are characterized by narrow beaches and wide erosional nearshore surfaces cut into silty clays with too little sand and gravel to support the development of beaches to protect the shoreline. Forbes (2000) estimated that 96% of the eroded material from the nearshore was fine silts and clays that were ultimately deposited in deep water.

Thorleifson et al. (2000) noted that shoreline erosion is causing Lake Winnipeg to not only expand southwards but northwards as well. Surveys in the mid 1990s reported shoreline recession at a number of north basin sites, including the cliffs from Warren Landing to Limestone Bay, as well as the south shore at Netley Marsh and the cliffs comprised of glaciolacustrine sediments along the south western shore (Forbes 2000). Nearshore bathymetric
and sub bottom acoustic profiles at numerous sites around the lake in 1996 revealed widespread
evidence of nearshore erosion in relict Lake Agassiz clays (Forbes 2000). Penner and Swedlo
(1974, In Forbes 2000) estimated that nearshore profile erosion extended out to about 2.4 to 3.7
m below the long-term mean water level. Baird and Stantec (2000) described the nearshore
erosion and resulting expansion of the lake, with permanent deposition of fine materials in deep
waters, as the natural evolution of a glacial lake such as Lake Winnipeg.

3.2.5.3 Sedimentation

Forbes (2000) described the sources of suspended sediment generated within the lake as a result
of the erosion of muddy cliffs in the south basin and at the north end of the lake, erosion of
glacial lake clay exposed in the nearshore throughout the lake, and wave driven re-suspension of
bottom mud. Wind generated waves were considered a major source of shore erosion and
sediment transport in Lake Winnipeg.

Burbidge et al. (2000) examined stable lead isotopic composition to determine the source of
sediments of the north and south basins of Lake Winnipeg. The authors found that the
Saskatchewan and Dauphin rivers and the eastern rivers did not supply much sediment to the
basin. They attributed the almost complete retention of lead in the western rivers to the
extensive lake systems upstream of Lake Winnipeg (e.g., Cedar and Moose lakes on the
Saskatchewan River). Much of the north basin sediment appeared to be derived from shoreline
erosion. In contrast, the south basin appeared to receive much of its sediment from the Red-
Assiniboine River system, which carries material derived from glaciolacustrine deposits rich in
clay. Brunskill and Graham (1979) reached a similar conclusion based on the chemical
composition and particle size of sediments in the depositional areas of the north and south basins.

No estimates of the total volume of suspended material introduced into the water column as a
result of shoreline and nearshore erosion were located during the course of this review. Based on
data for 1969-1974, Brunskill and Graham (1979) estimated that during high discharge years the
Red River transports nearly an order of magnitude more sediment (~3X10^6 tonnes yr^-1) than the
Winnipeg River, but in low discharge years the two rivers transport nearly an equal amount
(0.2 X10^6 tonnes yr^-1).

A number of short sediment cores collected in August 1994 were subjected to lead-210 dating to
estimate the period of deposition for each slice (Lockhart et al. 2000). Cores typically extended
to the late 19th century and therefore captured most of the period of European impact.
Sedimentation rates from approximately 1923-1994 were 645 gm^-2 yr^-1 in the north basin, and
1000 gm^-2 yr^-1 and 1420 gm^-2 yr^-1 at two locations in the south basin (Lockhart et al. 2000).
Additional cores were collected in the south basin in 1999. Average sedimentation rates for
these cores, which were collected slightly further south towards the Red River, were 861 $\text{gm}^2 \text{yr}^{-1}$ and 635 $\text{gm}^2 \text{yr}^{-1}$ for the period from the late 1800s to present; sedimentation rates shifted from 1.3 – 1.5 mm $\text{yr}^{-1}$ in the pre 1900 period to an average of 2.6 mm/yr post 1900 (Wilkinson and Simpson 2003).

Brunskill and Graham (1979) sampled surficial lake sediments in the deeper waters of the north and south basins. Over 70% of the samples were silty clays and clayey silts (the authors noted that coarser materials are often observed in nearshore shallow areas that were not included in their survey). Major areas of sediment accumulation were described in the deepest portions of the north and south basins.

### 3.3 THE CHEMICAL ENVIRONMENT

This review will focus on nitrogen and phosphorus as these are the elements that may ultimately limit phytoplankton growth, under conditions where nutrient limitation occurs.

#### 3.3.1 Nitrogen and Phosphorus

##### 3.3.1.1 Recent Study Results

Nitrogen and phosphorus concentrations during the open-water season in the north and south basins of Lake Winnipeg have been measured in numerous surveys conducted since 1992; however, during the course of this review, no published report synthesizing these findings was located. Such a synthesis is particularly important given the seasonal and spatial differences in nitrogen and phosphorus concentrations; any analysis to determine whether there are long term trends in nutrient concentrations must consider differences in sampling programs as well as specific conditions during a given survey.

The most recent measurements of nitrogen and phosphorus on the lake were collected by agencies participating in the LWRC during surveys conducted in early summer, mid to late summer and fall during the period 2002-2004 and fall 2005. Data from these surveys are still being analyzed, but M. Stainton provided preliminary maps of nutrient concentrations in the top 0.5 m (integrated sample) which he created in ARCGIS 9.1 and Spatial Analyst using the 2002 cruise data. Based on interpretation of these maps, the range of total phosphorus (TP) in the lake during the three cruises in 2002 was 0.02-0.22 mg/L. In spring, concentrations in the south basin were greatest at the mouth of the Red River (0.13 mg/L) and declined to 0.03 mg/L at the northern end of the basin. TP in the north basin were generally lower, and exhibited a gradient from peak values of 0.06 mg/L in the south east to minimum values of 0.02 mg/L in the north west (near Grand Rapids). In summer the same spatial pattern was evident in the south basin, though concentrations of TP were greater, with peak values of 0.22 mg/L at the Red River...
declining to 0.08 mg/L in the north end. In the north basin, the same southeast to northwest gradient was observed, ranging from a high of 0.08 mg/L to a low of 0.02 mg/L. In fall, the south basin graded from 0.16 mg/L at the Red River to 0.10 mg/L in the northern portion, while the north basin ranged from 0.08 mg/L in the southeast to 0.04 mg/L in the northwest.

Based on preliminary mapping, total nitrogen (TN) concentrations during the 2002 surveys ranged from 0.3 to 1.4 mg/L. As with TP, there was a gradient in the south basin with peak values occurring at the Red River (0.9 mg/L) and low values of approximately 0.5 mg/L occurring at the north end in spring. As with TP, concentrations of TN were lower in the north basin and ranged from 0.5 mg/L in the southeast to 0.4 mg/L in the northwest. Summer TN levels were higher than those recorded in spring, ranging from 1.3 mg/L at the Red River to 0.6 mg/L at the north end of the south basin (though other areas of high concentration were present), and from 1.0 mg/L to 0.4 mg/L in a southeast to northwest gradient in the north basin. In fall, concentrations in the south basin graded from approximately 1.0 mg/L at the Red River to approximately 0.7 mg/L at the north end; in the north basin peak levels occurred along the western shore near Poplar River (0.8 mg/L) but lowest levels still occurred in the northwest near Grand Rapids (0.48 mg/L).

A summary of mean TN and TP for Lake Winnipeg for the period 1992-2005 (Water Quality Management Section, unpublished data from Manitoba Water Stewardship, and Fisheries and Oceans Canada 2006) is provided in Table 3.
Table 3. Mean total nitrogen and total phosphorus for the south basin, narrows, and north basin of Lake Winnipeg for the period 1992 – 2005.

<table>
<thead>
<tr>
<th>Year</th>
<th>South Basin</th>
<th>Narrows</th>
<th>North Basin</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total Nitrogen mean (mg/L)</td>
<td>Total Phosphorus mean (mg/L)</td>
<td>Total Nitrogen mean (mg/L)</td>
</tr>
<tr>
<td>1992</td>
<td>26 0.475</td>
<td>26 0.069</td>
<td>18 0.458</td>
</tr>
<tr>
<td>1993</td>
<td>11 0.556</td>
<td>11 0.066</td>
<td>6 0.278</td>
</tr>
<tr>
<td>1994</td>
<td>28 0.844</td>
<td>28 0.098</td>
<td>10 0.654</td>
</tr>
<tr>
<td>1995</td>
<td>2 0.808</td>
<td>2 0.094</td>
<td>-</td>
</tr>
<tr>
<td>1996</td>
<td>14 0.751</td>
<td>14 0.076</td>
<td>3 0.615</td>
</tr>
<tr>
<td>1998</td>
<td>28 0.829</td>
<td>28 0.109</td>
<td>-</td>
</tr>
<tr>
<td>1999</td>
<td>34 0.661</td>
<td>34 0.093</td>
<td>23 0.623</td>
</tr>
<tr>
<td>2000</td>
<td>25 0.690</td>
<td>25 0.092</td>
<td>14 0.670</td>
</tr>
<tr>
<td>2001</td>
<td>20 0.725</td>
<td>20 0.119</td>
<td>8 1.130</td>
</tr>
<tr>
<td>2002</td>
<td>61 0.573</td>
<td>61 0.121</td>
<td>35 0.412</td>
</tr>
<tr>
<td>2003</td>
<td>38 0.880</td>
<td>38 0.131</td>
<td>14 0.799</td>
</tr>
<tr>
<td>2004</td>
<td>20 0.942</td>
<td>20 0.103</td>
<td>9 0.621</td>
</tr>
<tr>
<td>2005</td>
<td>27 0.792</td>
<td>27 0.167</td>
<td>37 0.654</td>
</tr>
</tbody>
</table>
Sampling in these years spanned the open water season and a range of sampling depths from surface samples to those collected at the bottom of the lake; however, given the differences in sampling effort and season, differences between years need to be interpreted with caution. Average values of TP in the south basin ranged from approximately 0.065 mg/L to 0.167 mg/L, while in the north basin values generally ranged from 0.020 mg/L to 0.080 mg/L (the average in one year was 0.12); in both basins there was a tendency for concentrations to be higher in more recent years. Average values of TN ranged from 0.475 mg/L to 0.942 mg/L in the south basin and 0.342 mg/L to 0.999 mg/L in the north basin; as with TP, there was a general trend to higher values in more recent years though the tendency was less marked than for TP.

### 3.3.1.2 Trends in Nutrient Concentrations

The first complete survey providing nutrients at stations located throughout Lake Winnipeg was completed during the limnological survey of 1969; these data were never published though some data are available in a partial draft manuscript (Brunskill 1969, unpublished data). Table 4 provides the range of nitrogen and phosphorus values obtained from maps within this draft manuscript for the open water season (June values represent a single survey, while two surveys were conducted in July and three in September/October). As with the 2002 samples, in general highest levels in the south basin were recorded near the Red River and decreased northwards, while in the north basin there was also a south to north gradient of decreasing concentration. The spatial and seasonal patterns and concentrations seen on the mapped data of 1969 and 2002 for TN and TP were remarkably similar, though peak values near the Red River tended to be higher in 1969.

An unpublished summary of the 1969 data provided by the Water Quality Management Section, Manitoba Water Stewardship (data from Brunskill 1969 survey) is also provided in Table 4 (note that the ranges for TN for the north basin derived from the mapped data seem high in comparison to the mean value; this apparent difference may have arisen due to the manner in which the maps were created or reflect an error in the draft manuscript). Comparison to unpublished data for the period 1992 – 2005 indicates that the 1969 average values fall within the range of the latter time period, though generally at the lower end of the range of concentrations, with the exception of TP in the south basin in 1969, which was among the higher values recorded for 1992-2005.
Table 4. Range of TN and TP obtained from data maps and means of TN and TP (provided by MWS) for a 1969 survey of Lake Winnipeg. Data sources are provided in the text.

<table>
<thead>
<tr>
<th></th>
<th>June</th>
<th>July</th>
<th>September - October</th>
<th>Mean Total (n)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TN (mg/L)</td>
<td>TP (mg/L)</td>
<td>TN (mg/L)</td>
<td>TP (mg/L)</td>
</tr>
<tr>
<td>South basin</td>
<td>0.420 - 1.400</td>
<td>0.062 - 0.185</td>
<td>0.560 - 1.820</td>
<td>0.031 - 0.618</td>
</tr>
<tr>
<td>North basin</td>
<td>0.420 - 0.560</td>
<td>0.031 - 0.093</td>
<td>0.420 - 1.050</td>
<td>0.015 - 0.062</td>
</tr>
</tbody>
</table>
In a study conducted to examine the effects of the 1997 Red River flood on Lake Winnipeg, Stewart et al. (2000) examined the historic record of TN and TP in the south basin using samples from surveys conducted in 1969 (Freshwater Institute [FWI]), 1992 (Province of Manitoba), 1994 (FWI), 1996 (FWI) and 1998 (FWI). Summer mean concentrations for the south basin provided in the report are provided in Table 5.

Table 5. Summer mean concentrations of total nitrogen and total phosphorus from the south basin of Lake Winnipeg.

<table>
<thead>
<tr>
<th>Year</th>
<th>TN (mg/L)</th>
<th>TP (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1969</td>
<td>0.573</td>
<td>0.083</td>
</tr>
<tr>
<td>1992</td>
<td>0.446</td>
<td>0.074</td>
</tr>
<tr>
<td>1994</td>
<td>0.880</td>
<td>0.095</td>
</tr>
<tr>
<td>1996</td>
<td>0.751</td>
<td>0.060</td>
</tr>
<tr>
<td>1998</td>
<td>0.902</td>
<td>0.117</td>
</tr>
</tbody>
</table>

The authors noted that TN appeared to have increased in the south basin since 1969, but not TP.

It is worth noting that, although the data presented in Stewart et al. (2000) and the unpublished values provided by Manitoba Water Stewardship follow the same general patterns, average values are not identical, even though the analyses were based on the same data sets. This indicates that the authors used different criteria to determine which data to include and highlights the need for a comprehensive synthesis of nutrient data to assess potential long term trends in nutrient concentrations in Lake Winnipeg.

### 3.3.1.3 Sediment Core Analysis

Mayer et al. (2006) analyzed phosphorus levels in dated sediment cores collected in the south basin of Lake Winnipeg in 1994 and 1999. Total phosphorus and three operationally defined forms of P (comprising the TP) were analyzed: non-apatite inorganic P (generally considered as the maximum potential particulate P that can be rendered soluble by diagenesis); apatite P (a form thought to be biologically inert); and organic P (P associated with carbon atoms in C-O-P or C-P bonds). Total phosphorus concentrations gradually increased from about 20 cm (corresponding to the 1930s) to the sediment water interface; more noticeable increases at 12 cm correspond to the 1960s. The authors suggested that deviations from a smooth trend could be related to variations in loading, such as that caused by changes in the inflow of the Red River. Phosphorus enrichment of the sediments was determined through comparison with aluminium, a conservative element. The doubling of TP relative to aluminum over the last 50 years was largely due to increases in the non-apatite P fraction, suggesting that much of the sedimentary
increase in P was due to anthropogenic inputs. Small increases in organic P in the upper 20 cm were attributed to increases in the primary productivity of the lake.

The authors noted the presence of several anomalous TP spikes at depth, which could have occurred during periods of greater mineral precipitation or as a result of the diffusion of P within the sediments. Mixing of the sediments in the surficial layers, which was observed to extend to 6 cm at one site and 12 to 13 cm at another, would effectively smooth out short term variations. The difference in the TP content and relative fractions of the P forms among the cores collected in the south basin was attributed to differences in the relative contribution of incoming sediments from the Red River among the sites.

Buhay and Simpson (2003) conducted carbon and nitrogen elemental and isotopic analysis of a core collected in the south basin of Lake Winnipeg in 1999. This core represented approximately 1000 years of lake history; the percentage of carbon and nitrogen increased in the top portion of the core, which corresponds to the 20th century. The C:N$_{\text{org}}$ and isotopic analysis of C suggested that the organic material was derived predominately from lacustrine algae, which increased in the latter part of the 20th century. Increases in N and changes in the isotopic N composition may be related to a variety of factors, including increased N from the deposition of plankton, the influx of N from fertilizer used in the watershed, or contributions due to sewage. The authors felt that a combination of the first two scenarios was the most likely cause.

Snowdon and Simpson (2003) examined the composition of organic matter in cores collected in the south basin of Lake Winnipeg in 1999. The authors concluded that organic matter undergoes extensive alteration in the water column prior to deposition, but that diagenetic processes continue after deposition, although at a much slower rate. Observed co-variance between total organic carbon and several elements (mercury, lead, phosphorus, uranium and zinc) could be coincidental or the result of continual migration and biological recycling of metals in the upper 20 cm of the lake-bottom sediments\textsuperscript{2}. The authors postulated that as organic matter buried in the sediments degrades, metals are released and migrate upwards to the organic-rich sediment interface where they are again fixed through the formation of organo–metallic complexes.

Nitrogen and phosphorus concentrations have also been measured in a sediment core collected in the north basin of Lake Winnipeg in 1994 (Salki 2005). The rate of phosphorus and nitrogen deposition has increased since approximately the mid 1960s with very high levels occurring since the 1990s. This change has been linked to increased nutrient levels in, and deposition from, the water column. However, analysis of the core in a manner similar to that conducted for the south basin by Mayer et al. (2006) has not been completed. Most importantly, levels of

\textsuperscript{2} It is not clear in the text whether the authors feel that P is migrating in the same manner as the metals.
nutrients in terms of their concentration in the sediments have not been determined, enrichment of P relative to a conservative element such as aluminum has not been calculated, and the potential effect of diagenesis leading to the upward migration of P has not been evaluated.

### 3.3.1.4 Nutrient Ratios

The ratio of N:P is often used to assess whether one or the other nutrient is potentially limiting algal growth. Although there is widespread agreement that high ratios are indicative of P limitation and low ratios indicative of N limitation, there is no widespread agreement on the levels at which changes occur (see Section 4.3.2 for a discussion of this topic). However, a ratio of less than 10:1 (by mass) is generally considered to indicate N limitation. Whether either nutrient is limiting growth also depends on absolute concentrations and other factors affecting growth (see Section 4.0).

Although the ratio of N:P changes during the course of the growing season, as a general approximation, the yearly average TN:TP mass ratios were examined to describe overall conditions. Based on the unpublished data provided by Manitoba Water Stewardship, the TN:TP ratio in 1969 was 6 and 11 for the south and north basins, respectively, indicating the potential for N limitation in the south basin. Values for the period 1992-2005 in the south basin were always less than or equal to 10 (indicating potential N deficiency). Ratios in the north basin were more variable: generally values were in the 10 to 20 range, though in three years the average ratio was below 10.

Stainton et al. (2004) reported on changes in the N:P ratio in the north and south basins in 2002. During the open-water season, concentrations of N generally declined relative to P and by August N:P ratios were centered near 10, indicative of excess P.

The topic of N and P limitation has been the subject of several investigations in Lake Winnipeg and results are discussed in relation to phytoplankton studies in Section 3.4.1.

### 3.3.1.5 Sources and Sinks of Nitrogen and Phosphorus

The cycling of nitrogen and phosphorus in aquatic systems is extremely complex, involving both biotic and abiotic pathways in the water column, sediments, and, in the case of nitrogen, atmosphere (see Section 4.0). No published report attempting to develop either an N or a P budget for Lake Winnipeg was identified during the course of this review, though several researchers described net changes in terms of changes in either the incoming loads (e.g., Brunskill et al. 1980) or the loads in the inflows and outflows (McCullough 2001).
Internal loading

Nitrogen-fixing cyanobacteria may result in a large influx of nitrogen to the lake ecosystem. Hendzell (2006, LWRC meeting) estimated, based on measurements of nitrogen fixation during a bloom of nitrogen-fixing cyanobacteria in the northern basin of Lake Winnipeg, that annual loading of nitrogen by fixation could be substantial (9300 tonnes/yr). It was noted that the activity of denitrifying bacteria could reduce the net gain of nitrogen to the lake ecosystem, however, denitrification has not been measured in studies conducted in the north basin to date. An on-going study is measuring the production of greenhouse gases, including nitrous oxide; levels of nitrous oxide measured indicate the de-nitrification is occurring (Stainton and McCullough no date).

Internal loading (regeneration) of phosphorus from the sediments is often an important part of the phosphorus cycle in lakes (see Section 4.5) but P release from the sediments has not yet been studied in Lake Winnipeg. Brunskill and Graham (1979) in an analysis of sediments from the deeper parts of Lake Winnipeg in 1969, reported that concentrations of P ranged from 13-27 umol P/g. Approximately 50% appeared to be labile. Based on C:N:P ratios, the authors concluded that most of the P had an inorganic rather than organic origin. Hecky et al. (1986) examining seasonal patterns of nutrient flux, noted an increase in P in the water column in fall, which was potentially attributed to a flux from the sediments. Adkins and Leduc (n.d.) obtained 18 sediment samples in fall 2004 and measured acid and alkaline phosphatase activity. Phosphatase activity was found in the sediments of both basins, with greater amounts in the north basin; the authors noted that this indicates that P is being released from the sediments. The lower activity in the south basin was potentially attributed to higher P concentrations in the water column, as microbial phosphatases are not released in the presence of excess dissolved P.

Transport in the Watershed

Estimation of the load of N and P entering and leaving Lake Winnipeg via tributary inflow and outflow is hindered by the availability of tributary water quality data. Much of the load may be transported during the spring freshet, a time of year with extremely variable water quality and discharge rates that can not be adequately represented by monthly or even biweekly sampling. In addition, the routine water quality sampling locations are not located at the mouths of rivers. Water samples on the Red River are generally collected at Selkirk, which is upstream of Netley Marsh, a potential sink of nutrients. For example, in summer 1973, water samples were collected during the same period at the mouth of the Red River and at Breezy Point, upstream of Netley Marsh (Manitoba Department of Mines, Resources, and Environmental Management 1973). Concentrations of nitrate and phosphate in water samples collected at the mouth were
35% lower than those at Breezy Point. Similarly, the average concentrations of TP and TN were 60% and 30% lower on the Saskatchewan River at Grand Rapids than upstream of the Carrot River (Water Quality Management Section, Manitoba Water Stewardship, unpublished data), indicating that significant reductions in nutrient concentrations had occurred as the river passed through its delta at Cedar Lake.

Similar issues may occur at the outlet of Lake Winnipeg, the Nelson River. Obtaining a representative sample of water leaving the lake is challenging as there are two outflow channels immediately entering Playgreen Lake, a large shallow lake which is the source of the east and west channel of the Nelson River. The routine water quality station at Norway House, downstream of Playgreen and Little Playgreen Lakes on the east channel, carries a small fraction of the total flow of the Nelson River. As noted by Jones and Armstrong (2001), this location may not be representative of water quality in the Nelson River as a whole. In addition, water quality may change at the outlet and as it passes through Playgreen Lake. Derksen and Hangasjarvi (1979) noted that water at Warren Landing (the sole outlet prior to Lake Winnipeg Regulation) was markedly more turbid than in the north basin proper. This observation is supported by the TSS levels shown in satellite photography, in which TSS levels at Warren Landing and the adjacent shoreline are higher than in the north basin as a whole in several images (McCullough et al. 2001).

Stainton et al. (2003) examined yields of N and P from 3 pristine watersheds in the Winnipeg River basin and 3 agricultural watershed in the Red River basin and found that 80% of the N and P loading is predicted from watershed yield. Stainton (2006, LWRC meeting) reported that the increased P loading from the Red River over the past decades was directly correlated to increased flow in the river.

Bourne et al. (2002, updated 2005) estimated that for the period 1994-2001, Lake Winnipeg received 86,701 tonnes/yr of TN and 7,845 tonnes/yr of TP, of which the Red River supplied 46% and 73%, respectively (as estimated by concentrations measured at Selkirk). The much larger flow of the Winnipeg River provided 27% of the TN and 13% of the TP, while the TN contribution of the Saskatchewan River was comparable to atmospheric contributions (15% of total) and TP accounted for 12% of the total load. Based on flow adjusted estimated changes in river concentrations produced by Jones and Armstrong (2001), the TN and TP loads to Lake Winnipeg have increased 13% and 10%, respectively, over the last 30 years. This increased loading was attributed to the Red River at Selkirk, which has experienced a 58% and 29% increase in TN and TP concentrations, respectively, and the Winnipeg River, where TP has increased 29% (Jones and Armstrong 2001). The authors noted that the concentration of TP at the outlet of Lake Winnipeg, the Nelson River, has declined by 21% over the same period, but
noted that the station on the east channel may not be representative of the river as a whole. The concentration of TN at this station did not change during the same period.

McCullough (2001) calculated monthly C, N and P fluxes into and out of Lake Winnipeg for major rivers for period for which sufficient water chemistry data were available (usually the late 1960s to 2000). In agreement with the analysis by Bourne et al. (2002), McCullough reported that most of the nutrients flowing into Lake Winnipeg are delivered to the south basin (70% of the TN and 84% of the TP). Annual watershed yields for all nutrients were correlated with annual water discharge; therefore, annual fluxes showed no simple linear trend but followed the pattern of discharge, which was generally low from the late 1970s to the early 1990s and then higher in the 1990s.

Researchers at the Freshwater Institute are currently developing models of N and P concentrations in Lake Winnipeg based on estimated loads entering and leaving the lake (M. Stainton presentation of work conducted jointly with R. Hesslein, LWRC meeting February 2006).

3.3.2 Other Parameters

3.3.2.1 Total Suspended Solids, Turbidity, and Light Penetration

Bajkov (1930) provided the earliest scientific account of suspended solids and light in Lake Winnipeg. Bajkov reported that water in the southern portion of Lake Winnipeg is discoloured from the mud brought in by the Red River. Farther northward, as this mud is precipitated, the water becomes greenish-brown due to the presence of diatoms, and north of George Island, the water is so clear that it appears green. Water was clearest in the deepest portions of the lake and in weedy, protected bays (e.g., Limestone Bay). The mouths of many of the eastern rivers were reddish brown due to the presence of peaty bogs. Secchi disk measurements reported by Bajkov (1930) are provided in Table 6 (the dates of the measurements were not provided).
Table 6. Light penetration results from various sampling locations in Lake Winnipeg 1930, using a 20 cm Secchi disk (Bajkov 1930).

<table>
<thead>
<tr>
<th>Sampling Location</th>
<th>Secchi depth (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Traverse Bay</td>
<td>1.00</td>
</tr>
<tr>
<td>Middle of southern portion</td>
<td>1.50</td>
</tr>
<tr>
<td>Narrow region near Gull Harbour</td>
<td>0.75</td>
</tr>
<tr>
<td>Northern portion</td>
<td>1.75</td>
</tr>
<tr>
<td>Northern portion in Asterionella area</td>
<td>1.50</td>
</tr>
<tr>
<td>Pigeon Bay</td>
<td>1.15</td>
</tr>
<tr>
<td>Saskatchewan River</td>
<td>0.10</td>
</tr>
</tbody>
</table>

Brunskill et al. (1979b) reported on Secchi depths measured during the 1969 limnological survey. Secchi depths in the south basin were generally lowest at the mouth of the Red River (0.25 m) and increased to 0.75 to 1.0 for much of the basin. The north basin was markedly clearer, with the deepest waters of the central northern basin generally ranging from 2.0 to 2.5 m, though all Secchi values were less than 2 m during the fall cruises. Secchi depths were generally lower along the northern shore and at the southern part of the entrance from the narrows. Secchi depths in the narrows were intermediate between the south and north basins. Brunskill et al. (1979b) measured light transmission and found that light transmission was closely correlated to suspended sediment concentrations. Suspended sediment was highest (5-80 g m$^{-3}$) near the mouth of the Red River and near the west and southwest shores of the south basin, and lower values (1-20 g m$^{-3}$) were found through most of the narrows and the north basin. The authors noted that the Red River is a major source of sediment for the south basin, while the Saskatchewan and Winnipeg rivers contribute much less, despite their large annual discharges. This difference was attributed to the sedimentary watershed and population of the Red River, in contrast to the Precambrian Shield watershed of the Winnipeg River. The Saskatchewan River drains a sedimentary watershed, but most of its sediment load is deposited in its delta (Cedar and Moose lakes) and more recently in reservoirs along its course.$^3$

McCullough et al. (2001) used remote satellite sensing to conduct an analysis of the spatial and temporal patterns of TSS distribution in mid summer in Lake Winnipeg for the period 1983-2000. As discussed by the authors, satellite sensing is an indirect measure of TSS as it is dependent on calibration between TSS measurements and turbidity meter results and the relationship between turbidity and spectral radiance measured by the satellite. In addition, satellite imagery is only available for cloud free days and periods of low wind; therefore it is

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$^3$ No analysis was found of the relative amount of sediment trapped by Cedar and Moose lakes prior to and after impoundment by the Grand Rapids G.S. However, Brunskill and Graham (1979) noted that many early explorers commented on the trapping of Saskatchewan River sediment in Cedar and Moose lakes.
biased for calm, cloud free conditions. The pelagic zone in the north basin was consistently the least turbid region and within it the area north of Long Point was usually the clearest. The maximum TSS in the central north basin was <10 mg/L: and the average for a large part of the region was less than 5 mg/L. The eastern shore, out to 10-20 km from shore was often more turbid in response to episodic wind-driven shore erosion and bottom resuspension; the highest TSS (in excess of 200 mg/L) was measured along this shore. The south basin was more turbid, averaging 10-20 mg/L over most of the basin. Traverse Bay at the mouth of the Winnipeg River is more clear. High TSS was frequently observed along the bays of the western shore of the narrows, and in the broad shallow mixing area between the narrows and the north basin.

McCullough et al. (2001) reported that correlation analyses of TSS as estimated by satellite imagery and various environmental variables suggested that TSS in Lake Winnipeg is more dependent on wind than any other factor. Correlation with the flow of influent rivers was weak, except for the north basin where there was a widespread inverse correlation with Saskatchewan River mean spring and summer discharge. The authors found it noteworthy that there was no correlation between discharge from the Red River and TSS in the south basin.

### 3.3.2.2 Oxygen Conditions

Oxygen levels are one of the critical indicators of eutrophication, and depressed oxygen levels are one of the primary pathways by which the excessive growth of algae and plants affects other organisms in the aquatic ecosystem. In addition, anoxic or hypoxic conditions at the sediment / water interface may result in a flux of nutrients from the sediments to the water column (see Section 5.4).

The earliest records of oxygen were collected by Bajkov (1930). Oxygen ranged from 9.5-11 ppm at 5 locations in both the north and south basin in mid summer (depth range 13-17 m); oxygen depletion at the bottom was not evident, even at sites that were weakly thermally stratified. During winter, a vertical gradient in oxygen levels was evident at the single station sampled, a 15 m deep site within the narrows where oxygen ranged from 25.6 ppm\(^4\) at the surface to 14 ppm at depth.

Synoptic records of oxygen levels during fisheries studies (e.g., Hewson 1953a, b, 1959) during the winter months did not indicate oxygen depletion to critical levels. Rybicki (1966) reported on surface and bottom oxygen concentrations at 30 stations in the north basin during sampling periods in early to mid June; late June to early July; and mid to late July in 1963 and 1964. Temperature did not differ by more than 1 °C. No evidence of oxygen depletion at depth was

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\(^4\) 25.6 ppm appeared in the text with no explanation, but is believed to be an error.
reported. Pollard (1973) revisited these sites in mid summer 1968 and reported that maximum surface to bottom temperature differences were less than 2 °C and that all oxygen values (both surface and bottom) were near or above saturation. In contrast, Kristofferson et al. (1975) reported thermal stratification in mid summer 1974, with depression of oxygen concentrations at depth (saturation ranged from 57 to 80%); after a storm eliminated the thermal stratification, oxygen saturation at depth was generally greater than 90%.

Surveys of the south basin in September and March through the 1960s seldom recorded oxygen levels below 5 ppm in the open water season, with few sites showing vertical differences (e.g., maximum observed difference occurred at a 7.9 m (26 ft) deep station with a surface and bottom concentration of 7.0 and 3.6 ppm, respectively) (Crowe 1975). Only a few instances of strongly depressed oxygen levels were recorded under ice (in March), and these were limited to bottom waters. During studies in the 1960s, elevated BOD was reported at locations close to the mouth of the Red River, but the effect of these inputs to the remainder of the south basin were limited in extent and degree (Crowe 1975).

Brunskill et al. (1979a) reported on results of bottom oxygen measurements at only a handful of stations, noting that Rybicki (1966) had completed a thorough survey only a few years earlier.

In 2003, agencies participating in the LWRC survey recorded thermal stratification and oxygen depletion in deep waters of the north basin in mid August (Stainton 2005). Vertical temperature profiles showed relatively uniform temperatures until the bottom 1-2 m of the water column, which were 6 to 7 °C cooler than the overlying water (14-15 °C vs. 21 °C) (Stainton 2005). The temperature differential suggested that the bottom waters had become isolated in June, 6-8 weeks before the sampling. Bottom samples were hypoxic (oxygen of 2-3 ppm vs. surface 8-9 ppm) with CO₂ and NH₃ enrichment. Most bottom samples had significant accumulations of algal remains. During fall sampling, no stratification or oxygen depletion of bottom waters was evident. This was the first occasion during which hypoxia was observed under open water conditions during the LWRC cruises; oxygen depletion in the lower 1 to 2 m of the water column had been observed previously during winter sampling (M. Stainton, pers. com.).

### 3.4 THE BIOLOGICAL ENVIRONMENT

The following section provides a summary of the current understanding of phytoplankton in Lake Winnipeg, based on current and historic studies, as well as long-term records on a geological scale derived from sediment cores. A brief overview of the current status and information regarding trends in the zooplankton, zoobenthos and fish community is also provided, as these groups may be impacted by changes associated with eutrophication and, in turn, may affect the eutrophication process.
3.4.1 Phytoplankton

The phytoplankton community in Lake Winnipeg has been measured using a variety of methods. The most direct method is through microscopic enumeration of cells in a sample of surface water, and measurement of cell dimensions to calculate cell volume and thus an estimate of biomass (Kling 2006). This method provides information, not only on total biomass, but on the relative abundance of various algal groups, which differ in their optimal growing conditions, as well as individual species, which can be used as indicators of environmental conditions. The major drawback of this method is that enumeration is time-consuming and expensive, and regimes specialized taxonomic expertise.

Chlorophyll \( a \), the major photosynthetic pigment, is the most widely used indicator of algal abundance, and is a relatively inexpensive means by which to obtain a large amount of data. The major disadvantage is that cellular chlorophyll \( a \) content varies with growing conditions and among groups of algae, so that the biomass of some groups (e.g., diatoms) is consistently underrepresented. In addition, differences in analytical methodology can lead to large differences in the amount of chlorophyll \( a \) reported (this is especially critical when comparing chlorophyll \( a \) levels over time), and chlorophyll \( a \) measurements may or may not be corrected for the presence of phaeophytin, a degradation product of chlorophyll \( a \) that interferes with its measurement.

Recently, satellite imagery has been used as a means of estimating the spatial extent and frequency of surface algal blooms. The advantage of this method is that it can provide information on a large spatial area, and during periods when no field samples are being collected. The disadvantages are: it measures only a specific component of the algal community, it is an indirect measure dependent on the calibration between measured chlorophyll \( a \) and surface reflectance, it is subject to interference from other water quality parameters (e.g., suspended sediments, dissolved organic matters), and it can only be used under specific conditions (calm, clear, sunny days).

Finally, several cores have been collected in Lake Winnipeg and these have to some extent been analyzed for the presence of fossil algal remains. This method provides a longer record of conditions in the lake than any other method, but, as with other data obtained from cores, is subject to uncertainties related to the preservation of material within the core.

Obtaining comparable and representative data over space and time for the phytoplankton community is further complicated by the extremely heterogeneous distribution of algae. Blooms can be very short lived and develop and disappear rapidly in response to shifts in growing conditions. Cells are also not uniformly distributed horizontally in the lake, nor within the water column or even the photic zone.
During the course of this literature review, no published reports were located that provide an analysis of changes in the phytoplankton community based on comparable measures (e.g., chlorophyll $a$, biomass) from the surveys of 1969, 1990s, and the early 2000s. Information in the following sections was obtained from unpublished manuscripts, various presentations, and other information/raw data provided directly by researchers involved in the studies. A brief summary of this information is provided below, but an analysis of the data is beyond the scope of this review.

### 3.4.1.1 Recent Study Results

The most recent measurements of chlorophyll $a$ on the lake were collected by agencies participating in the LWRC during surveys conducted in early summer, mid to late summer, and fall from 2002-2004. Although data from these surveys are still being analyzed, M. Stainton provided preliminary maps created in ARCGIS 9.1 and Spatial Analyst using the 2002 cruise data. Based on these maps, the total range of chlorophyll $a$ measured was 2-36 µg/L and the distribution was usually quite heterogeneous, with a gradient in the south basin from maximum levels at the southern end by the Red River to lowest values near the narrows. The north basin tended to also display a gradient with maximum values in the south east grading to minimum values in the north west (generally near Grand Rapids). However, chlorophyll $a$ values in both basins were quite heterogeneous. Specifically, in the first 2 weeks of June, chlorophyll $a$ on the lake ranged from 2-20 µg/L. The highest concentrations were in the south basin at the mouth of the Red River and around Elk Island, and concentrations generally decreased northwards to 4 µg/L.

Chlorophyll $a$ in the north basin ranged from 2-12 µg/L, with lowest values towards the west. In August, peak values were almost twice early summer levels, ranging from 4-36 µg/L. At the mouth of the Red River, chlorophyll $a$ concentration was 12 µg/L and decreased through the basin to 4 µg/L. In the north basin chlorophyll $a$ was heterogeneous, ranging from a low of 4 µg/L at Grand Rapids and the narrows to a peak of 36 µg/L near Berens River. By fall, peak chlorophyll $a$ values had declined, with most of the south basin at 2 µg/L and the north basin ranging from 4 to peaks of 18 µg/L in a few locations.

Chlorophyll $a$ concentrations generally reached higher peak levels in 2003 than 2002. Salki (pers. com.) calculated the mean maximum open water chlorophyll $a$ concentration of eight stations in the north basin at approximately 40 µg/L based on data collected during the LWRC cruise.
Unpublished chlorophyll $a$ data for Lake Winnipeg for the period 1992-2005 is provided in Table 7 (Water Quality Management Section, Manitoba Water Stewardship, and Fisheries and Oceans Canada 2006).

Table 7. Mean chlorophyll $a$ values between 1992 and 2005 for Lake Winnipeg.

<table>
<thead>
<tr>
<th>Year</th>
<th>South Basin</th>
<th>Narrows</th>
<th>North Basin</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>n</td>
<td>mean chl $a$ (µg/L)</td>
<td>n</td>
</tr>
<tr>
<td>1992</td>
<td>26</td>
<td>3.92</td>
<td>18</td>
</tr>
<tr>
<td>1993</td>
<td>11</td>
<td>4.27</td>
<td>7</td>
</tr>
<tr>
<td>1994</td>
<td>28</td>
<td>14.84</td>
<td>10</td>
</tr>
<tr>
<td>1995</td>
<td>2</td>
<td>4.50</td>
<td>-</td>
</tr>
<tr>
<td>1996</td>
<td>14</td>
<td>8.00</td>
<td>3</td>
</tr>
<tr>
<td>1998</td>
<td>18</td>
<td>4.16</td>
<td>-</td>
</tr>
<tr>
<td>1999</td>
<td>18</td>
<td>4.22</td>
<td>15</td>
</tr>
<tr>
<td>2000</td>
<td>13</td>
<td>4.46</td>
<td>10</td>
</tr>
<tr>
<td>2001</td>
<td>11</td>
<td>6.61</td>
<td>4</td>
</tr>
<tr>
<td>2002</td>
<td>53</td>
<td>7.13</td>
<td>34</td>
</tr>
<tr>
<td>2003</td>
<td>19</td>
<td>10.92</td>
<td>7</td>
</tr>
<tr>
<td>2004</td>
<td>10</td>
<td>3.90</td>
<td>5</td>
</tr>
<tr>
<td>2005</td>
<td>20</td>
<td>5.94</td>
<td>30</td>
</tr>
</tbody>
</table>

Given the differences in sampling effort and locations among years, observed trends need to be interpreted with caution. However, average values in the south basin ranged from approximately 4-15 µg/L while in the north values ranged from 3-22 µg/L; no temporal trend was evident in the south basin, but in the north the highest values tended to occur in the latter part of the period.

Kling (2006) reported on algal species composition and biomass from samples collected in 1969 (as part of the lake-wide survey conducted by the Freshwater Institute), in 1992 (samples collected during monthly cruises over the lake by the Dept. of Environment, Province of Manitoba, and Canadian Coast Guard) and in August 1994 (as part of the multidisciplinary lake-wide survey conducted by the Canadian Geological Survey and the Freshwater Institute). Results of these analyses are discussed in Section 3.4.1.2.

Blooms of diatoms have historically been an issue for the winter fishery (see following section); Kling (pers. com.) measured under ice algae blooms and found them to be dominated by the diatom *Aulacoseira islandica*, which has been common through the lake's history.
Satellite Imagery

Satellite imagery has recently been used to provide a series of "snap shots" of the north basin of Lake Winnipeg (G. McCullough, pers. com.). Simultaneous collection of chlorophyll \(a\) fluorescence data and satellite imagery in 2003 enabled the development of a relationship between chlorophyll \(a\) concentration in surface waters and colour spectra measured in MODIS images. The satellite recording these images has only been in orbit since 2001, so correlation between the MODIS images and an AVHRR image from a series of satellites in use since 1978 enabled acquisition of a longer record of satellite images, extending back to 1983. The satellite images record dense (>50 µg/L) surface accumulations of algae (i.e., cyanobacteria surface blooms). Analysis of at least 3 images per year found that there were widespread blooms in 8 of 9 years from 1995 to 2003, in comparison to 4 of 11 years for 1983-1994. Given that the lake could only be viewed on cloud free days and only a few days were cloud-free each summer, lack of a surface bloom observation was not proof that no bloom occurred; however, the sample was considered large enough (3 or more images/yr) that the frequency of blooms could be compared between the two decades.

3.4.1.2 Trends and Potential Causes

Phytoplankton community composition

Bajkov (1934) provided a qualitative description of the plankton of Lake Winnipeg; however, it is difficult to ascertain in which years sampling was conducted. He states that in late winter/early spring, diatoms are most common, and then numbers of diatoms and green algae increase. With respect to the blue green alga, \textit{Aphanizomenon flos-aquae}, he states "This is the commonest alga in Lake Winnipeg during the second portion of the summer. It reaches maximum abundance in August and, together with \textit{Anabaena}, in places it forms so thick a carpet on the surface, that many tourists and summer resorters refuse to go into the water to swim." Bajkov stated that as a rule, by the end of August, extensive areas are thickly covered, usually more so in the south basin where temperatures are higher, but also in the north basin if temperatures are warmer.

Anecdotal reports from fisheries studies also mention phytoplankton. Hewson (1953a, b) reported that during the 1951-52 winter fishery, the phytoplankton bloom that had been observed each year since 1949 re-appeared, leading to a decline in fish catch and the early closure of the winter fishing season. Limnological surveys conducted by Manitoba in the 1960s and early 1970s (e.g., Rybicki 1966, Kristofferson et al. 1975) provided limited information on the phytoplankton.
The first complete sampling of the phytoplankton community, including measurements of photosynthetic production, occurred during the limnological survey of 1969. These data were never published although some data are available (Brunskill 1969, unpublished data). Kling (2006) recently reported on results of microscopic enumeration of samples collected during this survey (see below). Brunskill et al. (1979a) reported that during dense algal blooms in calm weather in July and August, the surface of the water appeared to be covered with opaque light green or yellow green paint. The algal bloom was in the upper 30 cm of the water column, and often large clumps of cells and filaments floated on the surface.

A survey of attached algae on natural and artificial substrates conducted concurrently with the 1969 limnological survey found that cyanobacteria and green algae were relatively more abundant in the south than the north basin, and that algal distribution on the artificial substrates (marker buoys) appeared linked to light penetration (Evans and Stockner 1972).

Kling (2006) reported on algal species composition and biomass from samples collected in surface (0 – 0.5 m) waters in 1969, 1992 (samples collected during monthly cruises over the lake by the Dept. of Environment, Province of Manitoba, and Canadian Coast Guard) and in August 1994. The 1969 survey provided the most complete seasonal record, and indicated a spring and fall diatom growth, and one cyanobacteria bloom in the summer/fall. The spring bloom of diatoms occurred earlier in the north than south basin, and the cyanobacteria bloom was more pronounced in the north basin. Blooms were first found in the north basin in mid July and later in the south basin and narrows (late July, August, and September). The size of biomass and algal composition were rather similar for the north and south basin. Green algae were a larger part of the community in the south basin and narrows, while dinoflagellates and chrysophytes were more important in the north basin. The algal community in 1992 was quite different and dominated by diatoms and cryptophytes, which Kling ascribed to cool wet conditions in that year. During the August 1994 survey, a widespread bloom was present in the lake, comprised almost entirely of the cyanobacterium, *Aphanizomenon flos-aquae*. The maximum recorded biomass in 1994 was 5X higher than in 1969, but the mean basin values were only 2X higher. Kling (2006) noted that in a comparison of dominant species between 1969 and 1994, most of the same species were present but many were not recorded in the first scientific survey of the lake conducted in 1924. Several species characteristic of eutrophic waters were relatively more abundance in 1994 than in 1969.

**Chlorophyll a measurements**

No chlorophyll *a* data from the 1969 survey are provided in published reports. The Water Quality Management Section, Manitoba Water Stewardship provided unpublished data (from
Brunskill 1969 survey) which showed an average of 5.58 µg/L and 2.59 µg/L for the south and north basins, respectively. It should be noted that samples for the analysis of chlorophyll \( a \) were collected 1 m below the surface and so likely underrepresented the true abundance in the photic zone, in particular in the south basin. In addition, these samples would not be comparable to later studies, which collected surface samples.

In a study conducted to examine the effects of the 1997 Red River flood on Lake Winnipeg, Stewart et al. (2000) examined the historic record of chlorophyll \( a \) in the south basin using samples from surveys in 1969 (Freshwater Institute (FWI)), 1992 (Province of Manitoba), 1994 (FWI), 1996 (FWI) and 1998 (FWI). Summer mean concentrations for the south basin provided in the report are presented in Table 8\(^5\).

Table 8. Summer mean chlorophyll \( a \) concentrations from 1969 and 1992 to 1998 for the south basin of Lake Winnipeg.

<table>
<thead>
<tr>
<th>Year</th>
<th>Mean Chl ( a ) (µg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1969</td>
<td>6.6</td>
</tr>
<tr>
<td>1992</td>
<td>8.6</td>
</tr>
<tr>
<td>1994</td>
<td>19</td>
</tr>
<tr>
<td>1996</td>
<td>15</td>
</tr>
<tr>
<td>1998</td>
<td>5</td>
</tr>
</tbody>
</table>

The authors did not note a temporal trend. However, average chlorophyll \( a \) concentrations and average TN in the south basin were significantly correlated, which was interpreted as a demonstration of nitrogen limitation to algal growth in the south basin. Algal biomass in the 1998, post flood sample, was markedly lower than expected from the TN concentration; the authors considered a variety of explanations such as light limitation or inhibition from herbicides in the water but could not reach a definite conclusion. No relationship existed between TP and chlorophyll \( a \) in the south basin.

Evidence from sediment cores

Kling (1998) analyzed a long core from the north basin collected in 1994; as the intent of the study was to obtain rough background levels of the microfossils, a coarse sample resolution was chosen. The species composition of the fossil phytoplankton assemblage was interpreted as indicating that Lake Winnipeg has been oligo-mesotrophic since its beginnings post Lake Agassiz, with periodic blooms of cyanobacteria alternating with diatoms. Many of the same

\(^5\) As discussed for the nutrient data, results provided by Stewart et al. (2000) and Manitoba Water Stewardship are somewhat different.
species dominant today are present in the fossil record. In more recent times, there has been a progressive increase in the presence of cyanobacteria akinetes. Other evidence from the fossil record indicating an increase in the trophic status of the lake was a change in species composition from oligo-mesotrophic taxa to more eutrophic taxa. Kling (1998) notes that conditions under which algal remains are deposited can affect their preservation; in particular diatom frustules may dissolve under certain conditions.

A short core collected from the middle of the south basin was subjected to both lead 210 dating and analysis for phytoplankton in smaller layers to provide a more precise record of changes in the plankton in recent times. Diatoms (primarily *Aulacoseira*) and cyanobacteria akinetes exhibited pronounced peaks in abundance over the past 100 years, in particular in the last few decades.

### 3.4.1.3 Factors Affecting Algal Growth

Studies of factors affecting algal growth in Lake Winnipeg have primarily focussed on nutrient conditions. Healey and Hendzel (1980) collected samples from mid summer 1974 in Lake Winnipeg and, based on ratios of P/C, N/P, N/C, chlorophyll *a*/*C* and the protein/carbohydrate ratio in the particulate (phytoplankton) fraction, concluded that there was sufficient P and N for growth; this finding was supported by undetectable measures of phosphatase activity which is used to indicate P and N debts. The absence of nutrient limitation was attributed to growth limitation by other factors, in particular light. Further analysis by Hecky et al. (1993) comparing the ratios of particulate C, N and P to the Redfield ratio, found that growth in Lake Winnipeg, as well as some other large lakes, was either limited by physical factors or expressing a tendency to P limitation based on physiological indicators.

Hendzel et al. (2002) reported on measurements of four composition ratios (C:P, C:N, N:P, and C: Chl *a*) and three metabolic indicators (alkaline phosphatase activity, nitrogen fixation rates and nitrogen debts) in 2000/2001. Neither P nor N deficiency was significant during most of the open water period, suggesting that light limitation was the most important factor controlling phytoplankton, in particular in the south basin. Blooms of heterocystous cyanobacteria in the north basin were preceded by increased transparency coupled with adequate available P, and N deficiency. These species dominated the summer phytoplankton community and ultimately became P limited during bloom conditions. Nitrogen fixation was found to be significant in compensating for the low N:P supply ratio in the north basin. Preliminary results from investigations in 2002 found that phytoplankton under ice in April tended to be slightly phosphorus and nitrogen deficient, but during summer phytoplankton populations were neither P nor N limited. At a number of south basin stations and one north basin station, phytoplankton
were actively fixing atmospheric N; this would offset any apparent cellular N deficiency and on occasion P deficiency was seen to increase during periods of active N fixation (Hendzel 2005). In 2003, the APA and N debt data suggested that the phytoplankton community in spring was P sufficient in all basins while being strongly N deficient in the narrows and north basin, but not the south basin. The development of nitrogen-fixing algal blooms, which covered large area of the north basin and isolated blooms in the narrows and south basin during the summer relieved the N deficiency in areas where nitrogen fixation was high. Demand for P exceeded availability, as shown by a very high APA.

Algal toxins

The abundance of cyanobacteria in Lake Winnipeg has resulted in concerns related to the presence of toxins produced by these species. On September 9, 1996, a sample collected at Victoria Beach on Lake Winnipeg had a microcystin-LR concentration of 300 µg/L, whereas microcystin levels had declined to 0.2 µg/L in a follow-up sample taken later in the month (Gurney and Jones 1997). The majority of the microcystin-LR samples collected in Lake Winnipeg since 1999 have been in response to the development of extensive algal blooms along recreational beaches. These algal blooms have generated microcystin-LR levels in the 3 to 15 µg/L range with one exception at 331 µg/L (at Sandy Beach, Lake Winnipeg in August 2004) (A. Bourne, MWS, pers. com.). A number of algae blooms in Manitoba have generated a very thick algal scum that was unpleasant for swimming, but these blooms did not necessarily produce high levels of toxins. Severe algae blooms were observed at Alberta Beach and Gimli Beach, Lake Winnipeg in 2003, although microcystin-LR levels were relatively low (A. Bourne, MWS, pers. com.).

Herbert et al. (2003) reported on a sampling program to measure algal toxin levels in fish and waterfowl. Kling (pers. com.) reported that algal toxins had been recorded in several higher organisms, including fish.

3.4.2 Aquatic Macrophytes

No description of the macrophytes of Lake Winnipeg was located. Bajkov (1930) mentions that the areas of boulder and gravel bottoms were very rich in aquatic vegetation but does not provide further details. The limnological surveys conducted on Lake Winnipeg are generally limited to deeper, offshore waters.

Netley Marsh on the south side of Lake Winnipeg is an extensive area of rooted and submerged aquatic vegetation. Grosshans et al. (2004) provide a description of the marsh and changes between 1979 and 2001. The study noted that the loss of emergent vegetation and erosion of
uplands has increased the area of open water within the marsh from 35\% in 1979 to 51\% in 2001.

3.4.3  Fauna

The potential role of fauna and food web interactions in contributing to the eutrophication of Lake Winnipeg has been identified; however, during the course of this review, no published or unpublished analyses of this topic were located. Therefore, the following section provides a very brief overview of the faunal components of the Lake Winnipeg ecosystem and observations of changes than may be related to the trophic status of the lake, to the extent that such analyses have been completed.

3.4.3.1  Zooplankton

Zooplankton were sampled at stations throughout Lake Winnipeg as part of the recent studies by agencies participating in the LWRC. Pelagic zooplankton may be one group that has the longest record on Lake Winnipeg, with the potential for samples to be collected and analyzed in a comparable manner from the work conducted by Bajkov in 1929, and then in 1969, 1994, 1999 and during the recent surveys. Salki (1996) provided a comparative analysis of collections in 1929, 1969 and 1994. The 1969 survey did not reveal any clear change in the species composition since 1929; furthermore, community structure appeared to be relatively stable between 1969 and 1994. The settled plankton volume was higher in both basins in 1969 than in 1929. In 1994 abundance in the north basin was greater than in 1969 (though the abundance in the south basin had not changed).

Salki (2005) provided an updated analysis of changes in zooplankton when comparing samples collected in 1969, 1999, 2002 and 2003 throughout the open water season. Spring, summer and fall abundance of zooplankton has increased consistently since 1969; however temporal trends among regions were not consistent. The zooplankton community structure also changed, with shifts in the relative abundance of major groups (cladocerans in the north basin increased, while cyclopoids decreased), as well as changes in the abundance of individual species, some of which may be linked to habitat use. Larger cladocerans, which always dominated the community in the south basin, appeared to have increased in the north basin. The exotic invader *Eubosmina* had continued to spread.

Salki (2005) noted that many of the changes observed in the zooplankton community were consistent with the eutrophication of the lake; in addition, changes in zooplankton community structure could affect the phytoplankton community.
3.4.3.2  **Benthic Invertebrates**

Benthic invertebrates were sampled in Lake Winnipeg by Bajkov in the 1920s, throughout the 1960s as part of the Province of Manitoba's pollution survey of the lake, during the limnological survey of 1969 and in more recent studies, including work by agencies participating in the LWRC. Differences in sampling methodology and analytical methods confound attempts to identify temporal trends (Flannagan et al. 1994).

Stone and Cober (1965) reported on the results of their 1963-64 survey of the south basin (grid with 66 sampling sites) and compared results at stations at similar locations sampled by Bajkov in 1928 and 1929. The authors noted that the numbers of *Hexagenia* seem to have increased somewhat; which was at odds with local reports of the disappearance of huge emergences referred to in early reports. The Amphipoda had virtually disappeared from the south basin but were still present in the north basin but in reduced numbers. The Sphaeriidae, Gastropoda, Chironomidae, and in particular the Oligochaeta increased all over the lake.

Crowe (1973a) reported on results of sampling from 1965-1969 and concluded that there was no evidence that the rate of eutrophication increased during the 1960s. Five groups dominated the benthos: Chironomidae; Ephemeroptera; Oligochaeta; Gastropoda and Sphaeriidae.

Benthic invertebrates were sampled during the limnological survey of 1969. Flannagan et al. (1994) reviewed the results of benthic sampling on Lake Winnipeg and concluded that the most obvious changes between the 1920s and the 1969 survey were: the virtual elimination of the amphipod *Diporeia brevicornis* from the south basin; the elimination of two species of Trichoptera (*Phryganea cinerea* and *Molanna flavicornis*) from the south basin; the appearance of a new contaminant tolerant species of Trichoptera (*Oecetis inconspicua*) in the south basin and a loss of production and shift in the relative abundance of the two species of Ephemeroptera (*Hexagenia limbata* and *H. rigida*). In general, changes were the greatest in the south basin and moderated with latitude. The authors discussed various potential causes of these changes, but indicated that nutrient enrichment did not appear to be the primary cause.

Benthic invertebrates have most recently been sampled during the spring, summer and autumn cruises of the LWRC in 2002, 2003 and 2004 (Hann and Kowalchuk 2005). Sample analysis has not been completed. However, following the discovery of hypoxic conditions in the north basin, some stations were selected and analyzed. Tubicidae and Chironomidae were least affected by hypoxia, whereas Lumbriculidae, Sphaeriidae, Planorbidae and benthic crustaceans suffered drastic declines. The densities of many individual groups recovered by the following year, with some exceptions (i.e., benthic crustaceans).
### 3.4.3.3 Fish

There are currently 52 species in Lake Winnipeg (Franzin et al. 2003). In recent history, several warm-water species have successfully become established in Lake Winnipeg, including the bigmouth buffalo (*Ictiobus cyprinellus*), carp (*Cyprinus carpio*), rainbow smelt (*Osmerus mordax*), and white bass (*Morone chrysops*) (Franzin et al. 2003). The stonecat (*Noturus flavus*) and the black crappie (*Pomoxis nigromaculatus*) have both used Lake Winnipeg to disperse to tributary environments, but are caught rarely, if at all, in the lake itself (Franzin et al. 2003). No species have been lost from the community; however, both lake trout (*Salvelinus namaycush*) and lake sturgeon (*Acipenser fulvescens*) have declined to remnant status. Native species that are tolerant of turbid lacustrine environments are widespread through the lake. These include lake whitefish (*Coregonus clupeaformis*), cisco (*C. artedi*), goldeye (*Hiodon alosoides*), mooneye (*H. tergisus*), sauger (*Sander canadensis*) and walleye (*S. vitreus*). Invasive species that are likewise tolerant of these conditions are rainbow smelt, white bass and bigmouth buffalo.

Agencies participating in the LWRC conducted surveys of the pelagic small fish in Lake Winnipeg in 2002 and 2003 using a small mesh trawl net deployed from the CCGS *Namao* at the stations surveyed as part of the overall study (Franzin and Watkinson, no date). Data on larger predatory fish were collected in conjunction with the forage fish surveys in 2003. Emerald shiner (*Notropis atherinoides*) was by far the most frequently collected species, with smaller numbers of rainbow smelt, lake cisco, yellow perch (*Perca flavescens*), white bass, trout perch (*Percopsis omiscomaycus*), walleye, freshwater drum (*Aplodinotus grunniens*) and goldeye. There was considerable variability in the total catches in each year, but the number collected per trawl was consistently greater in the south basin, intermediate in the narrows, and lowest in the north basin. In 2003, rainbow smelt were proportionally more abundant in the north basin, comprising over half the catch. Given that this species was first recorded in the lake in the early 1990s (Campbell et al. 1991), its abundance in the north basin marks a rapid and large shift in the fish community in this basin. North basin predators and large grazers included walleye, sauger, yellow perch, cisco, longnose sucker, white bass, whitefish and goldeye. The same species were collected from the south basin with the exception of longnose sucker and whitefish and the addition of drum and black crappie.
Commercial Fishery

The commercial fishery began in the 1870s but the first records were not kept until 1873. Lake whitefish, walleye, and more recently sauger, have been the key species. Annual yields of these three species increased until the 1950s and then declined irregularly from 1950-1970 due to continued over exploitation. In 1971 the fishery was closed due to mercury contamination (Lysack 1986). This respite and optimal spawning conditions produced strong cohorts, which supported an increasing fishery through the 1970s (Lysack 1986). The major lake whitefish fishery occurs in the north basin and narrows in June, while sauger and walleye are harvested primarily in the south basin and narrows and in isolated areas in the north basin in summer, fall and to a lesser extent, winter. The annual landings of the commercial fishery have varied considerably over the period, with peaks of 10,000,000 kg annually in the early 1900s, the 1920s and late 1930s (Heuring 1993). Total commercially-marketed harvests from 1883 – 1989 are presented in Table 9.

In the last decade, catches of walleye have increased to record levels, while catches of sauger have declined. The whitefish catch varies, but has not displayed either a consistent increasing or decreasing trend. Lake Winnipeg comprises Manitoba's largest commercial fishery, with a total value of $20,380,350 in 2000-2001 (LWIC 2005).
Table 9. Total commercially-marketed harvests (kg) for Lake Winnipeg from 1883 – 1989 (Heuring 1993).

<table>
<thead>
<tr>
<th>Decade/Year</th>
<th>0</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
</tr>
</thead>
<tbody>
<tr>
<td>1880</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>72,867</td>
<td>359,000</td>
<td>790,821</td>
<td>800,000</td>
<td>1,270,261</td>
<td>1,779,660</td>
<td>1,815,446</td>
</tr>
<tr>
<td>1890</td>
<td>3,108,717</td>
<td>2,537,338</td>
<td>2,742,243</td>
<td>2,467,917</td>
<td>3,005,739</td>
<td>3,229,084</td>
<td>3,480,535</td>
<td>2,530,112</td>
<td>2,591,150</td>
<td>1,820,643</td>
</tr>
<tr>
<td>1900</td>
<td>3,258,816</td>
<td>6,422,545</td>
<td>8,018,180</td>
<td>9,727,273</td>
<td>10,159,090</td>
<td>9,806,818</td>
<td>8,125,000</td>
<td>4,683,636</td>
<td>2,996,726</td>
<td>4,661,544</td>
</tr>
<tr>
<td>1910</td>
<td>6,058,637</td>
<td>5,716,728</td>
<td>3,894,000</td>
<td>3,206,090</td>
<td>5,717,907</td>
<td>5,602,909</td>
<td>6,211,407</td>
<td>6,610,818</td>
<td>6,896,963</td>
<td>4,127,728</td>
</tr>
<tr>
<td>1920</td>
<td>4,473,317</td>
<td>6,756,682</td>
<td>5,654,173</td>
<td>4,586,627</td>
<td>4,016,091</td>
<td>5,183,591</td>
<td>8,333,927</td>
<td>8,894,093</td>
<td>10,110,164</td>
<td>10,519,983</td>
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<td>1930</td>
<td>5,471,681</td>
<td>3,791,728</td>
<td>4,679,364</td>
<td>5,043,226</td>
<td>6,300,045</td>
<td>4,717,090</td>
<td>5,469,136</td>
<td>6,551,680</td>
<td>8,429,135</td>
<td>7,102,819</td>
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<td>1940</td>
<td>9,952,092</td>
<td>9,164,364</td>
<td>8,040,955</td>
<td>8,264,910</td>
<td>6,251,589</td>
<td>7,422,864</td>
<td>6,504,182</td>
<td>7,471,864</td>
<td>7,476,137</td>
<td>7,868,545</td>
</tr>
<tr>
<td>1950</td>
<td>8,331,363</td>
<td>8,531,592</td>
<td>7,556,864</td>
<td>5,376,817</td>
<td>6,664,045</td>
<td>7,986,308</td>
<td>6,289,046</td>
<td>5,671,500</td>
<td>5,842,274</td>
<td>4,421,408</td>
</tr>
<tr>
<td>1980</td>
<td>5,037,212</td>
<td>5,330,453</td>
<td>4,955,546</td>
<td>5,150,324</td>
<td>5,595,563</td>
<td>5,204,807</td>
<td>5,104,892</td>
<td>5,518,273</td>
<td>5,766,186</td>
<td>5,634,482</td>
</tr>
</tbody>
</table>
4.0 NUTRIENT ENRICHMENT: LAKES

There is a vast array of scientific literature surrounding the occurrence, effects, and management of anthropogenic nutrient enrichment in lakes and reservoirs across the globe. Cultural eutrophication of lakes was identified decades ago as a potential issue of concern for ecosystem health and has been the subject of rigorous study and debate for many years. As Schindler (2006) pointed out, a majority of recent research has been conducted in Europe, with relatively less in North America since the initial pioneering work in the 1970s. However, as the science of eutrophication is now fairly advanced and lake restoration programs (primarily P loading reduction programs) have been ongoing for decades, there is a great deal of research emerging regarding re-oligotrophication of lakes and reservoirs. This ‘re-oligotrophication’ literature is a valuable contribution to our understanding of the effects of nutrient enrichment in lake ecosystems as well as the pathways of effects of reversing nutrient enrichment. As such, it contributes both to the areas of lake management and recovery as well as to refining our understanding of nutrient enrichment effects.

Chambers et al. (2000a) indicated that small lakes and ponds in the Canadian prairies are particularly prone to nutrient enrichment, most notably phosphorus, due to the dry climate. Additionally, they indicated that lake sediment analyses have shown that some prairie lakes were eutrophic prior to European settlement, due to the nutrient-rich soils of the region, long residence times, and high evaporation rates. The occurrence of toxic algal blooms have been reported in prairie lakes prior to the use of inorganic fertilizers and intensive livestock operations. Therefore, the aquatic ecosystems of the Canadian prairies may be particularly unique in their inherently high nutrient concentrations and trophic status. However, increasing nutrient enrichment is evident across Canada due primarily to municipal and industrial wastewater discharges and agriculture (Chambers et al. 2001). Surface waters of the southern prairie provinces have been identified as “areas of concern” respecting nutrient enrichment (Chambers et al. 2001).

The following is a review of selected primarily recent (and a few older) publications pertaining to the broad subject of “nutrients in lakes and reservoirs”. An attempt has been made to capture recent literature, literature reviews and synthesizes, and literature on key issues with respect to this topic. It is emphasized, however, that this review was not intended to be comprehensive.

4.1 RESPONSES OF LAKES TO NUTRIENT ENRICHMENT

Literature on the effects of nutrient enrichment on lake ecosystems is immense and dates back numerous decades. However, although the literature indicates some generalizations regarding trophic responses, it also indicates that responses may vary between systems. General effects of
nutrient enrichment on phytoplankton, macrophytes, zooplankton, fish, biodiversity, lake chemistry, and trophic cascades are discussed below.

### 4.1.1 Phytoplankton

The most commonly reported effect of nutrient enrichment on aquatic ecosystems is an increase in algal biomass, which is often expressed as chlorophyll $a$ concentration. Typically, there is a positive correlation between TP (and often TN) and phytoplankton biomass or chlorophyll $a$ in lakes and reservoirs, although the precise relationship may vary between lakes. For example, Sondergaard et al. (2005a) recently evaluated correlations between TP concentrations in hundreds of Danish lakes to various 'indicators', including phytoplankton biovolume. Not surprisingly, they reported a positive correlation between phytoplankton biovolume and TP, as well as to TN. Findlay and Kasian (1987) reported that addition of both N and C increased phytoplankton biomass by two to four times pre-treatment levels in Lake 226. A greater response (increases of 4 to 8 times) was observed in the half of Lake 226 receiving N, C, and P.

In general, it is the algal response to nutrient enrichment that is typically associated with eutrophication and the focus of concern and management.

In addition, the continuum of eutrophication is generally associated with a reduction in phytoplankton diversity and shifts in community composition, ultimately culminating in the dominance of a few or even single species (Dokulil and Teubner 2000 In Dokulil and Teubner 2005; Moss et al. 2004).

Nutrient enrichment may also result in an increased presence and abundance of cyanobacteria (Pick and Lean 1987). This generally manifests as a greater dominance of these organisms, more frequent blooms, and or blooms forming over a larger area. As some cyanobacteria may produce toxins, they may pose a significant risk to aquatic life, wildlife, and even human consumers (Chambers et al. 2001). The prevailing explanation for this shift in community structure is related to the common decline in nitrogen to phosphorus ratios that typically occurs during eutrophication; cyanobacteria that may fix atmospheric nitrogen are believed to hold a competitive edge when nitrogen-limiting conditions occur. A more detailed discussion of the factors affecting the occurrence of cyanobacteria blooms is provided in Section 4.4.

### 4.1.2 Macrophytes

Eutrophication typically results in the reduction or loss of macrophytes in lakes (Sondergaard et al. 2003). For example, Sondergaard et al. (2005a) reported that macrophyte maximum depth distribution and coverage was negatively correlated to TP and TN in Danish lakes. Macrophytes are considered to be pivotal for shallow lakes as they affect a lake's capacity to buffer nutrient
enrichment and are integrally involved in trophic cascades and transitions between clear water and turbid states (Hietala et al. 2004). Accordingly, the effect of nutrients on macrophytes has been the focus of considerable study in recent years, and it has been demonstrated that nitrogen has a negative effect on macrophyte species richness (James et al. 2005) and reduces the probability that macrophytes will remain dominant in lakes with moderately high TP (Gonzalez Sagrario et al. 2005). For example, nitrogen additions were required in conjunction with phosphorus to negatively affect macrophyte dominance in mesocosm experiments; macrophytes were not affected at TP 0.07-0.13 mg/L and TN \( \leq 1.2 \) mg/L (Gonzalez Sagrario et al. 2005). For further discussion of macrophytes, see Sections 4.3.5 and 4.8.

4.1.3 Zooplankton

Zooplankton biomass may increase (e.g., Hietala et al. 2004; Moss et al. 2004; Sondergaard et al. 2005a) or decrease (reviewed In O'Brien et al. 2005) due to eutrophication and increase (Jeppesen et al. 2002) or decrease during re-oligotrophication (e.g., Phillips et al. 2005). In addition, zooplankton species composition may shift with increasing nutrient concentrations. Sondergaard et al. (2005a) reported that cladoceran biomass was negatively correlated, while cyclopoid and calanoid copepod biomass was positively correlated, to TP and TN in Danish lakes. Zooplankton body size may also be affected by nutrient enrichment, generally decreasing with nutrient enrichment (e.g., Carpenter et al. 2001; Gophen et al. 1999). The precise relationship is highly dependent upon trophic cascades (changes to and interactions across the food web), as high rates of planktivory tend to reduce the mean body size of zooplankters (e.g., Carpenter et al. 2001). However, there is typically an inverse relationship reported between the size of zooplankters and chlorophyll \( a \) and primary production (e.g., Carpenter et al. 2001). Furthermore, Carpenter et al. (2001) speculated that when planktivory is low, the size of zooplankters is related to nutrient inputs, but under high planktivory, large-bodied zooplankters are effectively eliminated. The ratio of zooplankton:phytoplankton also typically declines as a result of eutrophication (e.g., Jeppesen et al. 2000; Moustaka-Gouni et al. 2006); Jeppesen et al. (1997) reported a rapid and sharp decrease occurs above a TP concentration of 0.1 mg/L.

4.1.4 Fish

In addition to increased plant and algal biomass and subsequent secondary effects on water quality, nutrient enrichment may lead to increased productivity (i.e., biomass) of higher trophic levels such as desirable fish species (EC 2004). However, eutrophication is generally accompanied by a decrease in piscivorous fish species concurrent with an overall increase in fish abundance and biomass (e.g., Romo et al. 2005; Sondergaard et al. 2005a). This effect has been shown to progress along a TP gradient in lakes (Sondergaard et al. 2005a), with increasing abundance of planktivorous fish occurring with increasing TP (Jeppesen et al. 2000). In Europe,
the fish community typically responds to increasing TP enrichment along a gradient in the order salmonids - coregonids - percids - cyprinids (reviewed in Jeppesen et al. 2005a). Ultimately, eutrophication may result in an increased dominance of planktivorous fish, which in turn may alter the top-down controls on phytoplankton (see Section 4.3.4). Similarly, re-oligotrophication is often accompanied by an increase in piscivorous fish (e.g., Jeppesen et al. 2002) and to some extent, shifts in community structure.

4.1.5 Biodiversity

Although an increase in productivity may be perceived as positive, nutrient over-enrichment may also lead to reduced biodiversity and a shift from more sensitive biota to more ‘tolerant’ species (EC 2004). As indicated in Section 4.1.1, phytoplankton diversity tends to decrease with trophic status (e.g., Moss et al. 2004). However, the response of different trophic levels in terms of diversity appears to vary. For example, Jeppesen et al. (2000) reported that zooplankton and submerged macrophytes species richness declined with increasing TP in shallow Danish lakes. Conversely, for fish, phytoplankton, and floating-leaved macrophytes, species richness unimodally related to TP, peaking at 0.1-0.4 mg/L.

4.1.6 Physical/Chemical Changes

The most widely cited and arguably most significant physical and chemical changes that typically accompany nutrient enrichment in lakes are reduced water clarity and dissolved oxygen. In general, with increasing trophic state, lakes become more turbid and light may become limiting to plants (i.e., macrophytes) and some algal species.

Shallow lakes shift from a clear-water state to a turbid state with increasing eutrophication, according to the alternative stable states theory (see Section 4.3.5 for further discussion). There are many mechanisms proposed to explain this shift, including loss of macrophytes, increased algal turbidity, and top-down effects (i.e., effects of higher trophic levels on primary production). Recent studies have shown that conversion from a turbid to a clear-water state does not always occur during lake re-oligotrophication (see Section 4.3.4 for further discussion).

Dissolved oxygen (DO) depletion may be localized and short-term/cyclical, in association with diurnal DO sags in phytoplankton blooms or attached algal mats, or more prolonged and widespread, as occurs during the decay of an algal bloom. DO depletion may cause fish kills in the most severe instances. Additionally, the promotion of anoxic conditions at the sediment-water interface due to death and decay of algal blooms that have settled on the lake bottom can create a positive feedback for eutrophication. The low DO promotes the release of phosphorus from sediments, thus contributing to internal nutrient loading.
4.1.7 Cascading Effects

Eutrophication often leads to a cascade of environmental effects, particularly across the food web (i.e., trophic cascades) and many of the effects exist in positive feedback loops. For example, increased biomass of phytoplankton can reduce water clarity, and thus adversely affect macrophytes. Similarly, reduced macrophyte biomass may also lead to increased phytoplankton. A typical trophic cascade response pathway to nutrient enrichment, as observed in Danish lakes, involves a shift to dominance of planktivorous and benthivorous fish species, increased predation pressure on zooplankton and a subsequent shift to smaller zooplankters, and reduced grazing pressure on phytoplankton. Similarly, it has been suggested that eutrophication led to an increase in planktivorous fish in Danish lakes, which in turn reduced grazing of zooplankton on phytoplankton and periphyton, ultimately leading to the loss of submerged macrophytes (reviewed In Batterbee et al. 2005). These ‘trophic cascades’ ultimately lead to increased turbidity and, in some systems, reduced macrophytes (Sondergaard et al. 2005a).

The structure of a food web, which may vary between lakes, may play a significant role in determining the response of a lake to eutrophication/nutrient enrichment. For example, as summarized in a recent review, Schindler (2006) indicated that lakes with even numbers of pelagic trophic levels exhibit lower algal abundance than lakes with an odd number of trophic levels. The premise is that lakes dominated by planktivorous fish exert a cascade of top-down effects through effects to phytoplankton grazers.

The particular significance of trophic cascades in eutrophication, however, lies in the fact that disturbances unrelated to eutrophication that disrupt the food web can exert highly significant effects on the response of a lake to nutrient enrichment. That is, stressors unrelated to nutrient enrichment per se, may affect lake responses to nutrients. As Schindler (2006) points out, overharvesting and habitat degradation leading to reductions in piscivorous fish may lead to shifts in low to high algal states in eutrophic lakes.

4.1.8 Initial Conditions

Numerous studies have shown that the particular response of a lake to eutrophication and to re-oligotrophication is affected by the initial conditions of the lake (i.e., the 'starting point', Hietala et al. 2004; Moss et al. 2004; 2005). For example, as Moss et al. (2004) indicate, starting conditions such as macrophyte density, nutrient concentrations, or chlorophyll a influence the response trajectory in a lake. Hietala et al. (2004) demonstrated through mesocosm experiments in a shallow Finnish lake that the initial community structure (i.e., presence/abundance of macrophytes and subsequent cascading effects on the food web structure) and weather in a given year determined the responses to nutrient enrichment and fish (roach) additions. That is, in a
year when macrophyte growth was poor (owing to a late, cold spring and summer), a turbid state was maintained with high phytoplankton growth. Conversely, the following year was warmer and macrophyte biomass was high, which in turn lead to the occurrence of a clear water state.

4.2 TROPHIC CATEGORIES

There are many trophic categorization schemes for lakes, a few of which are summarized in Table 10. As a general rule of thumb, lakes with TP concentrations less than 10 µg/L are oligotrophic and lakes with TP in excess of 100 µg/L are hypereutrophic (USEPA 2000a). Carvalho et al. (2002) presented the common classification scheme, which is based on ratings of bad to excellent, developed for European lakes (Table 11). Criteria used in European member states to define lakes and reservoirs “subject to eutrophication” are presented in Table 12.

Perhaps the most widely accepted trophic categorization scheme is that developed by the Organization for Economic Co-operation and Development (OECD 1982). This scheme was also adopted, with a minor modification, by the CCME (2004) in their phosphorus guidance framework (see Section 4.6.1 for discussion). Despite the wide acceptance of trophic categorizations, it is important to consider that trophic transitions generally occur as a continuum, rather than as abrupt changes associated with changes in nutrient concentrations (e.g., Sondergaard et al. 2005a). The OECD (1982) recognized this danger and strongly recommended consideration of probability distributions when determining lake trophic status. The OECD (1982) also recommended that consideration of all parameters should be granted when applying fixed boundary trophic status categorizations to lakes and the evaluation of trophic condition should consider the entire picture. However, it is also important to recognize that according to the current alternative stable states theory, transitions between clear water and turbid states can be very abrupt (Romo et al. 2005).

Although most trophic categorization schemes are based entirely or are dominated by TP as the variable of greatest concern, Sondergaard et al. (2005a) emphasized the need to consider nitrogen in the trophic categorization of lakes, due to the importance of nitrogen for macrophytes. Similarly, the USEPA nutrient criteria incorporate TP, TN, chlorophyll a and Secchi depth, and lakes and reservoirs are to comply with criteria for all of these variables (USEPA 2000a). Consideration of eutrophication response variables (e.g., chlorophyll a) is critical, as not all lakes and reservoirs exhibit the same relationship between nutrients and phytoplankton. For example, as reservoirs are typically more turbid than lakes, they exhibit lower chlorophyll a: TP ratios, due to light limitation (USEPA 2000a).
Table 10. Summary of selected trophic status classification schemes.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Lake Trophic Status</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.01-0.5</td>
<td>0.3-3</td>
</tr>
<tr>
<td>(µg/L)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 11. A common classification scheme for European lakes (Cardoso 2001 In Carvalho et al. 2002).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Excellent</th>
<th>Good</th>
<th>Fair</th>
<th>Poor</th>
<th>Bad</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP (mean)</td>
<td>(µg/L)</td>
<td>Natural levels</td>
<td>&lt;125% of Excellent</td>
<td>125-150% of Excellent</td>
<td>150-200% of Excellent</td>
<td>&gt;200% of Excellent</td>
</tr>
<tr>
<td>Chlorophyll a (mean)</td>
<td>(µg/L)</td>
<td>&lt;2</td>
<td>&lt;5</td>
<td>&lt;10</td>
<td>&lt;25</td>
<td>&gt;25</td>
</tr>
<tr>
<td>Chlorophyll a (maximum)</td>
<td>(µg/L)</td>
<td>&lt;5</td>
<td>&lt;10</td>
<td>&lt;20</td>
<td>&lt;50</td>
<td>&gt;50</td>
</tr>
<tr>
<td>Secchi Depth (mean)</td>
<td>(m)</td>
<td>&gt;5</td>
<td>2-5</td>
<td>1.5-2</td>
<td>1-1.5</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Secchi Depth (minimum)</td>
<td>(m)</td>
<td>&gt;3</td>
<td>1-3</td>
<td>0.7-1</td>
<td>&lt;0.7</td>
<td>&lt;0.7</td>
</tr>
</tbody>
</table>

Table 12. Threshold criteria used in European member states to designate lakes and reservoirs subject to eutrophication (Cardoso 2001 In Carvalho et al. 2002).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>United Kingdom</th>
<th>Ireland</th>
<th>France</th>
<th>Spain</th>
<th>Joint Research Centre Ispra (Italy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>(µg/L)</td>
<td>&gt;50</td>
<td>&gt;25</td>
<td>-</td>
<td>&gt;30</td>
<td>Reference conditions</td>
</tr>
<tr>
<td>Chl a (mean)</td>
<td>(µg/L)</td>
<td>-</td>
<td>-</td>
<td>&gt;60</td>
<td>&gt;8</td>
<td>&gt;10</td>
</tr>
<tr>
<td>Chl a (maximum)</td>
<td>(µg/L)</td>
<td>&gt;30</td>
<td>&gt;35</td>
<td>-</td>
<td>&gt;25</td>
<td>&gt;20</td>
</tr>
<tr>
<td>Secchi Depth</td>
<td>(m)</td>
<td>&lt;3</td>
<td>-</td>
<td>-</td>
<td>&lt;3</td>
<td>&lt;3</td>
</tr>
</tbody>
</table>
4.3 FACTORS AFFECTING EUTROPHICATION RESPONSES

There is on-going discussion and research involving the factors and, in particular, the dominant factors controlling eutrophication responses, where the 'responses' are generally defined on the basis of phytoplankton abundance and taxonomic composition. These factors range from chemical (i.e., nutrient relationships, or the "bottom-up" control pathway), physical (e.g., lake water levels, water residence times, turbidity/light, and temperature), to biological (i.e., the "top-down" controls of food webs). Additionally, recent research has identified the 'alternative stable states' theory for shallow lakes which postulates that a lake may exist in two alternative conditions at moderate nutrient concentrations. The following overview of these topics is based on a selection of recent publications and is not intended to comprise all of the enormous body of literature related to this topic.

4.3.1 Nutrient Thresholds

The literature pertaining to nutrient thresholds for eutrophication responses, some of which is summarized in Table 13, is extensive and relatively consistent. For TP, thresholds associated with various critical responses generally range from approximately 0.01 to 0.1 mg/L. Concentrations greater than 0.1 mg/L of TP are typically associated with significant adverse ecological effects such as complete dominance by cyanobacteria and/or significant changes to lake processes (see Table 4). Thresholds are less commonly identified for TN, likely due to the emphasis on phosphorus as the critical limiting nutrient. However, recent studies have shown that macrophyte abundance and the attainment of clear water states are associated with concentrations of TN <1.2-2 mg/L (Gonzalez Sagrario et al. 2005).

Concentrations of dissolved forms of N and P associated with nutrient limitation are more difficult to determine than total forms and there is less information on thresholds for inorganic forms of N and P. However, a concentration of soluble reactive phosphorus (SRP) of <0.01 mg/L has been associated with significant reductions in chlorophyll a (Sas 1989 In Phillips et al. 2005) and a threshold of <0.03 mg/L dissolved inorganic nitrogen (DIN) has been suggested for nitrogen limitation (Wilander and Persson 2001). Additionally, at DIN concentrations in excess of 0.1 mg/L non-heterocystous cyanobacteria reportedly dominate over heterocystous forms (Lawrence et al. 2000).
Table 13. Summary of ecological thresholds relating nutrient concentrations and other factors to eutrophication response variables.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Unit</th>
<th>Response</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>&lt; 0.01 (mg/L)</td>
<td></td>
<td>Minimal risk due to toxic algae</td>
<td>Acknowledged threshold may not be achievable in many lakes</td>
<td>Jacquet et al. 2005</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.01-0.015 (mg/L)</td>
<td></td>
<td>Cyanobacteria contributed significantly to phytoplankton biomass in deep Danish lakes undergoing oligotrophication</td>
<td></td>
<td>Reviewed In Sondergaard et al. 2001</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.03 (mg/L)</td>
<td></td>
<td>Minimal risk of cyanobacteria dominance</td>
<td></td>
<td>Downing et al. 2001</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.05 (mg/L)</td>
<td></td>
<td>Low seasonal variability of TP concentrations in shallow Danish lakes.</td>
<td></td>
<td>Jeppesen et al. 2005b</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.05 (mg/L)</td>
<td></td>
<td>Filamentous cyanobacteria disappear (shallow Dutch lakes)</td>
<td></td>
<td>Reeders et al. 1998</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.05-0.1 (mg/L)</td>
<td></td>
<td>Re-establishment of macrophytes</td>
<td></td>
<td>Jeppesen et al. 1999, 2000</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.05 (summer mean) (mg/L)</td>
<td></td>
<td>Dominance of filamentous cyanobacteria disappears</td>
<td>In conjunction with &lt; 1.35 mg N/L</td>
<td>Portielje and Van der Molen 1999</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.1 (mg/L)</td>
<td></td>
<td>Fish reductions increase Secchi depths by &gt; 1 m in shallow Danish lakes</td>
<td></td>
<td>Jeppesen et al. 1990 In Bergman 1999</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.1 (mg/L)</td>
<td></td>
<td>Secchi disk remained above 1 m in Lake Ringsjon, Sweden</td>
<td></td>
<td>Bergman 1999</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.1 (mg/L)</td>
<td></td>
<td>Internal P loading restricted to July and August in shallow Danish lakes</td>
<td></td>
<td>Sondergaard et al. 1999</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.1 (summer mean) (mg/L)</td>
<td></td>
<td>Likelihood of high zooplankton grazing in shallow lakes increases markedly</td>
<td></td>
<td>Jeppesen et al. 2000</td>
</tr>
<tr>
<td></td>
<td>&lt; 0.08-0.15 (mg/L)</td>
<td></td>
<td>Re-establishment of macrophytes</td>
<td></td>
<td>Jeppesen et al. 1990 In Gonzalez Sagrario et al. 2005 Jeppesen et al. 2005b</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.05 (mg/L)</td>
<td></td>
<td>Cyanobacteria contributed significantly to phytoplankton biomass in shallow Danish lakes undergoing oligotrophication</td>
<td></td>
<td>Jeppesen et al. 1999</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.05-0.1 (mg/L)</td>
<td></td>
<td>Effect of fish removal in large shallow lakes small or temporary</td>
<td></td>
<td>Jeppesen et al. 1999</td>
</tr>
<tr>
<td></td>
<td>0.05-0.20 (mg/L)</td>
<td></td>
<td>P is limiting and filamentous cyanobacteria decline rapidly with decreasing TP (shallow Dutch lakes)</td>
<td></td>
<td>Reeders et al. 1998</td>
</tr>
</tbody>
</table>
Table 13. Continued.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Unit</th>
<th>Response</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>zooplankton:phytoplankton decline sharply above 0.1 mg/L</td>
<td></td>
<td>Jeppesen et al. 1997</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>cyanobacteria exceed 70% of the total phytoplankton biomass</td>
<td></td>
<td>Watson et al. 1997 In Moustaka-Gouni et al. 2006</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>TN:TP ratios decrease at a declining rate above this threshold</td>
<td></td>
<td>Downing and McCauley 1992</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>High seasonal variability of TP concentrations in shallow Danish lakes. Summer concentrations 2x to 4x higher than winter.</td>
<td></td>
<td>Reviewed In Sondergaard et al. 2001</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.100</td>
<td>(mg/L)</td>
<td>Phosphorus inhibits nitrogen fixation</td>
<td></td>
<td>Jeppesen (pers. Comm. In Tonno and Noges 2003)</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.200</td>
<td>(mg/L)</td>
<td>Phosphorus inhibits nitrogen fixation</td>
<td></td>
<td>Tonno and Noges 2003</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.20</td>
<td>(mg/L)</td>
<td>P not limiting and cyanobacterial dominance approaches 100% (shallow Dutch lakes)</td>
<td></td>
<td>Reeders et al. 1998</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.2</td>
<td>(mg/L)</td>
<td>Internal P loading occurs from April to September in shallow Danish lakes</td>
<td></td>
<td>Sondergaard et al. 1999</td>
</tr>
<tr>
<td>SRP</td>
<td>&lt;0.010</td>
<td>(mg/L)</td>
<td>Chlorophyll a is reduced</td>
<td></td>
<td>Sas 1989 In Phillips et al. 2005</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.01</td>
<td>(mean summer in epilimnion) (mg/L) P not limiting to algal growth</td>
<td></td>
<td>Cooke et al. 1993 and Marsden 1989, both In Fortin and Gurney 1998</td>
<td></td>
</tr>
<tr>
<td>TN</td>
<td>1.2-2 (summer mean)</td>
<td>(mg/L)</td>
<td>shift to a more turbid state with low plant biomass</td>
<td>Mesocosm experiments with an accompanying TP &gt; 0.2 mg/L; threshold for Danish lakes irrespective of TP concentration (0.03-0.2 mg/L)</td>
<td>Gonzalez Sagrario et al. 2005</td>
</tr>
<tr>
<td></td>
<td>&lt;2 (summer mean)</td>
<td>(mg/L)</td>
<td>high coverage of macrophytes in shallow Danish lakes</td>
<td></td>
<td>Gonzalez Sagrario et al. 2005</td>
</tr>
<tr>
<td></td>
<td>&lt;1.35</td>
<td>(mg/L)</td>
<td>When TP &lt; 0.05 mg/L cyanobacteria dominance disappeared (231 lakes and ponds in Netherlands)</td>
<td></td>
<td>Portielje and Van der Molen 1999</td>
</tr>
</tbody>
</table>
Table 13. Continued.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Unit</th>
<th>Response</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>TN:TP</td>
<td>64</td>
<td>(molar ratio)</td>
<td>threshold for cyanobacterial dominance</td>
<td>29 by mass</td>
<td>Smith 1983</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Smith and Bennett 1999</td>
</tr>
<tr>
<td></td>
<td>49</td>
<td>(molar ratio)</td>
<td>threshold for cyanobacterial dominance in lakes</td>
<td>22 by mass</td>
<td>Smith et al. 1995 In Havens et al. 2003</td>
</tr>
<tr>
<td></td>
<td>33</td>
<td>(molar ratio)</td>
<td>threshold for nitrogen limitation and occurrence of nitrogen fixation in Swedish lakes</td>
<td>15 by mass</td>
<td>Reviewed In Wilander and Persson 2001</td>
</tr>
<tr>
<td></td>
<td>44</td>
<td>(molar ratio)</td>
<td>Initiation of nitrogen fixation in cyanobacteria (Lake Vortsjarv, Estonia)</td>
<td>20 by mass</td>
<td>Tonno and Noges 2003</td>
</tr>
<tr>
<td>DIN:SRP</td>
<td>22</td>
<td>(molar ratio)</td>
<td>strong nitrogen limitation; favour nitrogen-fixing cyanobacteria</td>
<td>10 by mass</td>
<td>Smith et al. 1995 In Havens et al. 2003</td>
</tr>
<tr>
<td>DIN</td>
<td>&lt; 0.03</td>
<td>(mg/L)</td>
<td>occurrence of nitrogen deficiency</td>
<td>Swedish lakes</td>
<td>Wilander and Persson 2001</td>
</tr>
<tr>
<td></td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>Non-heterocystous cyanobacteria (<em>Microcystis</em> and <em>Oscillatoria</em>) will dominate over heterocystous species where other conditions are conducive to either group</td>
<td>Lawrence et al. 2000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 0.1</td>
<td>(mg/L)</td>
<td>N fixation most likely to occur in Lake Okeechobee, FA</td>
<td>Smith et al. 1995 In James et al. 2005</td>
<td></td>
</tr>
<tr>
<td>Nitrate</td>
<td>&lt; 0.5</td>
<td>(mg/L)</td>
<td>Sediments of Lake Muggelsee, Germany undergoing oligotrophication began to release P</td>
<td>Kohler et al. 2005</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 1 (annual maximum)</td>
<td>(mg/L)</td>
<td>threshold for high plant diversity</td>
<td>James et al. 2005 In Moss et al. 2005</td>
<td></td>
</tr>
<tr>
<td>Temperature</td>
<td>&lt; 15</td>
<td>(°C)</td>
<td>Low cyanobacterial biomass in four Alberta lakes; high diatom biomass</td>
<td>Zhang and Prepas 1996</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 15</td>
<td>(°C)</td>
<td>Low diatom biomass in four Alberta lakes; high cyanobacterial biomass</td>
<td>Zhang and Prepas 1996</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 20</td>
<td>(°C)</td>
<td>Increased dominance of cyanobacteria</td>
<td>Pick and Lean 1987</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 21</td>
<td>(°C)</td>
<td>increased likelihood of cyanobacteria</td>
<td>In association with low N:P</td>
<td>McQueen and Lean 1987 In Stockner and Shortreed 1988</td>
</tr>
<tr>
<td></td>
<td>22-24</td>
<td>(°C)</td>
<td>Favours <em>Cylindrospermopsis raciborskii</em> growth</td>
<td>Gorzo 1991 In Istvanovics and Somlyody 2001</td>
<td></td>
</tr>
</tbody>
</table>
Table 13. Continued.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Unit</th>
<th>Response</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Macrophyte coverage</td>
<td>&gt; 5 (%)</td>
<td>Lower TN, TP, and chlorophyll a in Dutch lakes and ponds</td>
<td>Portielje and Van der Molen 1999</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sediment TP</td>
<td>1 (mg/g d.w.)</td>
<td>Saturation Level</td>
<td>Cooke et al. 1993 In Scharf 1999</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sediment TP</td>
<td>2.7 (mg/g d.w.)</td>
<td>P saturation of sediments in Lake Muggelsee, Germany and sediments became net source of P</td>
<td>Kohler et al. 2005</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zooplanktivorous Fish Density</td>
<td>&lt; 10 (g/m²)</td>
<td>For successful trophic cascades and biomanipulation of eutrophic lakes</td>
<td>Reviewed In Sondergaard et al. 2000</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Piscivore Stocking Density</td>
<td>&gt; 0.1 (individuals/m²/year)</td>
<td>For successful biomanipulation of eutrophic lakes sufficient to shift to a clear-water state</td>
<td>Reviewed In Sondergaard et al. 2000</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turbidity</td>
<td>76 (NTU)</td>
<td>Sediment resuspension stimulated primary productivity in Lake Waihola, New Zealand</td>
<td>Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tunbidity</td>
<td>&gt; 200 (NTU)</td>
<td>Light limitation in Lake Waihola, New Zealand</td>
<td>Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Light</td>
<td>1-5.2 (E/m²/day)</td>
<td>Light co-limiting to algae</td>
<td>Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Light</td>
<td>&lt; 5-7 (E/m²/day)</td>
<td>Light limitation in turbid reservoirs</td>
<td>Knowlton and Jones 2000 In Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Light</td>
<td>&gt; 6 (E/m²/day)</td>
<td>Nutrient limitation occurs in turbid reservoirs</td>
<td>Knowlton and Jones 2000 In Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Light</td>
<td>1-6 (E/m²/day)</td>
<td>Co-limitation of light and nutrients in turbid reservoirs</td>
<td>Knowlton and Jones 2000 In Schallenberg and Burns 2004</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
4.3.2 Limiting Nutrients

There is continued debate regarding the issue of limiting nutrients in aquatic ecosystems (Pick and Lean 1987). However, the overwhelming majority of published scientific literature states that phosphorus is the most limiting nutrient to primary production in lakes (and freshwater systems in general). The scientific rationale behind this is largely explained on the basis on nutrient stoichiometry.

The basis of the P limitation argument was largely entrenched through whole lake nutrient addition studies conducted in the 1970s at the Experiment Lakes Area (ELA) in the boreal shield of Ontario (Schindler 1975). These studies conclusively demonstrated that phosphorus was more limiting to algal growth than nitrogen in these systems. The first seminal experiment on this issue was conducted in Lake 226 in which one half was fertilized with N, P, and C and the other half was fertilized with only N and C. Findlay and Kasian (1987) reported that phytoplankton biomass in the basin receiving P, N, and C increased four to eight times and in the other basin which received only C and N, the biomass increased by two to four times pre-fertilization levels. Chlorophyll \( a \) concentrations were also found to correlate to TP concentrations, ultimately leading to the conclusion that phosphorus is the limiting nutrient in lakes (Schindler 1977).

The P limitation argument is related to the stoichiometric nutrient requirements of algae and the relative abundance of phosphorus and nitrogen in the environment. In general, nitrogen is more abundant in the aquatic environment and phosphorus is less soluble than nitrogen. Collectively, the supply-demand of nitrogen and phosphorus determine which nutrient is more limiting. This concept was first introduced many years ago and has been periodically tested, and largely supported, over the years.

Phytoplankton contain, on average, nitrogen and phosphorus at a molar ratio of 11:1 or 5:1 by mass (Hecky and Kilham 1988). Therefore, the resource-ratio hypothesis argues that nitrogen becomes limiting to phytoplankton at low ambient N:P ratios and phosphorus becomes limiting at high ambient N:P ratios. The precise thresholds (or ratio) vary in the literature but typically a nitrogen:phosphorus ratio less than 10 (by mass) is considered to indicate N limitation (Smith 2001). However, the nutrient stoichiometric requirements of algae can vary considerably between species and even within species. Table 14 provides a few examples of optimal N:P ratios for some phytoplankton species. Smith (1982) argued that due to these inherent differences, alteration of the N:P ratio would result in shifts in the phytoplankton communities. Bulgakov and Levich (1999) also found that the cellular N and P requirements of several species of Chlorococcales and Cyanophyta (determined experimentally) were similar to the “optimal” environmental concentrations for growth. As discussed in Section 4.4.1, there is a great deal of evidence that cyanobacteria are favoured at low N:P ratios.

<table>
<thead>
<tr>
<th>Species</th>
<th>TN:TP</th>
<th>Molar</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Melosira Binderana</td>
<td>7</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Microcystis sp.</td>
<td>9</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Synedra ulna</td>
<td>10</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Skeletonema costatus</td>
<td>12</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Asterionella formosa</td>
<td>12</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Anabaena flos-aquae</td>
<td>&lt; 18</td>
<td>&lt; 8</td>
<td></td>
</tr>
<tr>
<td>Microcystis aeruginosa</td>
<td>18-40</td>
<td>8.5-18</td>
<td></td>
</tr>
<tr>
<td>Ankistordesmus falcatus</td>
<td>21</td>
<td>9</td>
<td></td>
</tr>
<tr>
<td>Pseudoanabaena catenata</td>
<td>21</td>
<td>9</td>
<td></td>
</tr>
<tr>
<td>Selenastrum capricornutum</td>
<td>23</td>
<td>10</td>
<td></td>
</tr>
<tr>
<td>Chaetoceros affinis</td>
<td>24</td>
<td>11</td>
<td></td>
</tr>
<tr>
<td>Fragilaria crotonensis</td>
<td>25</td>
<td>11</td>
<td></td>
</tr>
<tr>
<td>Scenedesmus obliquus</td>
<td>30</td>
<td>14</td>
<td></td>
</tr>
<tr>
<td>Synechococcus linearis</td>
<td>24-44</td>
<td>11-20</td>
<td></td>
</tr>
<tr>
<td>Oscillatoria agardhii</td>
<td>27-46</td>
<td>12-21</td>
<td></td>
</tr>
<tr>
<td>Cryptomonas erosa</td>
<td>39</td>
<td>18</td>
<td></td>
</tr>
<tr>
<td>Synechococcus sp.</td>
<td>46</td>
<td>20</td>
<td></td>
</tr>
<tr>
<td>Scenedesmus quadricauda</td>
<td>87</td>
<td>39</td>
<td></td>
</tr>
</tbody>
</table>

N:P ratios have been widely examined in lakes of varying trophic condition and it is has been shown that the ratios generally decline with increasing trophic condition (e.g., Downing and McCauley 1992). The OECD (1982) notes that the N:P ratio generally decreases from more than 100 in oligotrophic systems to less than 10 in eutrophic systems. As a corollary, lakes tend to shift from phosphorus to nitrogen limitation with increasing eutrophication (Downing and McCauley 1992).

N:P ratios tend to decline in lakes with increasing eutrophication because N:P ratios tend to be low in the inflows to lakes receiving high nutrient loading (i.e., the "resource supply ratio" is low). For example, sewage and runoff from urban lands have low N:P ratios, whereas runoff from forested lands and precipitation tend to have high ratios (Downing and McCauley 1992). Similarly, Downing and McCauley (1992) indicate that naturally eutrophic systems tend to have low N:P ratios because the dominant geology in such basins is typically limestone or apatite, which have low N:P ratios.
Additionally, nutrient cycling is altered by eutrophication, which tends to reinforce the occurrence of nitrogen limitation (e.g., increased denitrification reduces nitrogen loads in lakes), particularly at high TP concentrations (Downing and McCauley 1992). Interestingly, there may also be biological feedback loops that contribute to the low N:P ratio in eutrophic systems. For example, Gophen et al. (1999) suggested that the reduction in size of zooplankton due to changes in the fish community in Lake Kinneret may enhance the occurrence of low N:P ratios because smaller zooplankton excrete low N:P ratios. Large *Daphnia* contain lower N:P ratios than the phytoplankton on which they graze and, consequently, excrete excess nitrogen, thereby enhancing or maintaining phosphorus limitation (Schindler 2006).

Although the bulk of the scientific literature indicates that lakes are generally phosphorus limited, there are numerous examples of nitrogen limitation and nitrogen and phosphorus co-limitation. A discussion of the significance of nitrogen in eutrophication and consideration of N controls in lake management are provided in Section 4.8.

### 4.3.3 Physical Factors Affecting Eutrophication/Algal Biomass

In general, the concepts of trophic status and nutrient criteria relate nutrient concentrations to algal growth and to the secondary effects of nutrient enrichment. However, the precise applicability of a trophic categorization scheme to a given waterbody must consider local conditions, local water usages, and factors other than nutrients that may limit plant and algal growth (e.g., temperature and light). For example, at a given nutrient concentration, algal biomass may be lower in highly turbid systems, such as reservoirs, relative to more clear lakes (USEPA 2000a). Therefore, it is critical to consider site-specific factors pertinent to assessing the effects of nutrient enrichment on aquatic ecosystems when establishing nutrient criteria. The USEPA indicates that further investigation is warranted where a lake is found to exceed nutrient criteria but chlorophyll *a* and Secchi depth (i.e., 'response indicators') are lower than would be expected at those nutrient concentrations (USEPA 2000a). Inorganic turbidity, colour, or flushing rates may all affect nutrient-algae relationships. The inclusion of response indicators in the USEPA nutrient criteria provides a mechanism for accounting for site-specific factors other than nutrients in lake eutrophication management.

Algal biomass does not increase linearly in all lakes and major factors other than phosphorus have also been found to affect algal biomass including nitrogen, light, lake morphometry, and zooplankton grazing (USEPA 2000a). As the USEPA (2000a) stated, if only one factor (i.e., phosphorus) was always limiting to algal growth, then the derivation of nutrient criteria would be a simple task. In a review of nutrient limitation, Hecky and Kilham (1988) emphasized that a variety of factors, one of which is nutrients, may limit phytoplankton growth and that "it is
simplistic to assume that all the species in a community are limited by a single factor”. The following sections discuss the effects of various physical factors on algal growth.

### 4.3.3.1 Lake Depth

Many studies have discussed the differences in the responses of deep and shallow lakes\(^6\) to nutrient enrichment. The studies discussing differences in deep and shallow lakes are too numerous to discuss here, however, key differences that have been noted may be summarized as follows:

- Deep lakes may have higher water clarity than shallow lakes at similar TP concentrations, possibly due to the occurrence of sediment resuspension in shallow systems (e.g., Sondergaard et al. 2005a);
- Shallow lakes are more vulnerable to nutrient enrichment due to small water volumes and lower dilution (Sondergaard et al. 2003);
- Mixed lakes, which are typically shallow, exhibit higher chlorophyll \(a\): TP ratios (Riley and Prepas 1985);
- Shallow lakes may have higher chlorophyll \(a\) or phytoplankton biomass at high TP concentrations (i.e., > 100 µg/L) relative to deep lakes; however, deep and shallow lakes appear to have similar chlorophyll \(a\) concentrations at lower TP concentrations (Sondergaard et al. 2005a);
- Effects to macrophytes are generally greater in shallow lakes as macrophytes are more significant in these systems (e.g., Sondergaard et al. 2005a). Shallow lakes also exhibit alternative stable states, with abrupt transitions between the two;
- Effects of grazing and trophic cascades (i.e., top-down effects) are hypothesized to be lower in shallow lake ecosystems due to the presence of macrophytes, which increase the habitat complexity (i.e., providing refuges for zooplankton grazers, reviewed in Hietala et al. 2004);
- Deep lakes typically stratify and are generally dominated by nitrogen-fixing cyanobacteria, whereas shallow eutrophic lakes are typically dominated by non-nitrogen-fixing cyanobacteria such as *Oscillatoria* (Havens et al. 2003);
- Shallow lakes typically recover more slowly than deep lakes (e.g., Wilander and Persson 2001);

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\(^6\) It should be cautioned that the definition of "shallow" lakes varies in the scientific literature although in a conceptual sense the distinction between deep and shallow lakes is intended to separate lakes that do not stratify from those that do.
• Shallow lakes appear to be less responsive to external nutrient loading reductions, relative to deep lakes (e.g., Wilander and Persson 2001), at least partially due to the prevalence of internal loading (Sondergaard et al. 2005a);

• Nutrient concentrations in shallow lakes may be significantly increased by sediment resuspension (reviewed in Schallenberg and Burns 2004); and

• Seasonal responses to external loading reductions may vary between deep and shallow lakes. Shallow lakes tend to respond with reduced TP in spring, early summer, and fall and no change in late summer, whereas, deep lakes show declines from May to August (Sondergaard et al. 2005a). This may reflect differences in internal loading.

4.3.3.2 Water Level and Water Level Changes

As the effects of water level changes are influenced by lake morphometrics and watershed characteristics, it is perhaps not surprising that the scientific literature regarding the effects of water level changes on nutrient concentrations and phytoplankton biomass is inconsistent and varies between lakes. Some studies report no effect, others report nutrients increase, and still others report they decrease with changes in lake levels. For many years, it was believed that TP was affected by water levels in Lake Okeechobee (e.g., Canfield and Hoyer 1988, Havens et al. 2003), but more recently no significant correlation was found using an extended data set (Havens and James 2005). Conversely, Kangur et al. (2003) concluded that water level has a significant effect on dissolved nutrient concentrations in Lake Peipsi, Estonia. Nitrate and phosphate concentrations were positively and negatively correlated to lake water level, respectively; as a corollary, nutrient ratios were lowest during low water years. The authors speculated that at low water levels, internal P loading is likely more significant, and at high water levels, there is greater loading of nitrogen from the catchment. In other instances, decreases in water levels cause increases in sediment resuspension, nutrients, and chlorophyll a (reviewed in Hoyer et al. 2005). Conversely, studies have also reported that reduced water levels cause nutrients and chlorophyll a to decline due to increased macrophyte biomass and reduced sediment resuspension.

Hoyer et al. (2005) examined the relationships between nutrients, chlorophyll a, and water level fluctuations in 84 Florida lakes. No pattern associated with fluctuations could be discerned; in some cases the relationships between the variables were positive, some negative, and still others, not significant. Hoyer et al. (2005) also showed that water level was significantly positively, negatively, or insignificantly correlated to nutrients and chlorophyll a in selected Florida lakes.
Turner et al. (2005) evaluated the effect of water level changes in a whole lake experiment emulating typical reservoir water level fluctuations at the ELA, northwestern Ontario. The study found no effect of water level fluctuations on nutrients and that phytoplankton were largely unaffected. The greatest effect to periphyton was a physical one - loss of suitable substrate. They concluded that effects of water level fluctuations would be greatest in the littoral zone.

### 4.3.3.3 Water Residence Times/Flushing Rates

Water renewal is very significant in controlling the effects of nutrient enrichment, however the importance and direction of the relationship varies according to the magnitude of residence times of lakes. Schindler (2006) indicated that a doubling of water renewal time of a lake is approximately the equivalent of doubling the nutrient input. The most profound effects of fluctuations in lake residence times on lake trophic conditions are observed for lakes with relatively short residence times (i.e., days). However, the relationship between water residence time and nutrients or algae in rapidly flushed lakes is variable.

In some cases, decreased water residence times have resulted in lower nutrients in lakes. For example, Grasmere Lake, England, appears to have experienced a period of reduced eutrophication over the period of 1945-1965 and the most likely explanation is the coincidental occurrence of a wet cycle (Barker et al. 2005). Although this would have increased nutrient loading, it was thought that the increased flushing rate (the lake is characterized by a rapid flushing rate) would more than compensate.

Similarly, Rippey et al. (1997) suggested that a period of accelerated eutrophication in an Irish lake, inferred from sedimentary records, was due to a drought which subsequently increased the lake water residence time (reduced flushing rate), thereby increasing P retention, and increasing P release from sediments due to enhanced stratification. The authors suggested that lakes with residence times of approximately one year may be susceptible to this type of response to climatic variability. Rippey et al. (1997) also suggested that more eutrophic lakes are more susceptible to changes in hydraulic residence times than oligotrophic lakes, because the latter release little P from sediments and alterations to flushing rates have little overall effect on P retention.

Moustaka-Gouni et al. (2006) demonstrated the effectiveness of increasing lake flushing rates as a means of controlling biomass of cyanobacteria (and phytoplankton in general) in Lake Kastoria (Greece), as has been reported by others (reviewed In Moustaka-Gouni et al. 2006). Rapid flushing pulses have also been an effective method of disrupting the development of cyanobacteria blooms (e.g., Padisak et al. 1999 In Moustaka-Gouni et al. 2006).
Conversely, there is some evidence that reductions in water residence times increase nutrients and algae. For example, Rennella and Quiros (2006) reported that in two interconnected lakes with short residence times (days), the flushing rate was positively correlated to TP and chlorophyll $a$, presumably due to an effect on zooplankton. Several hypotheses for the increased phytoplankton were suggested, including a bottom-up effect (i.e., increased TP) and a top-down effect related to a reduction of large cladocerans. The authors postulated that the decrease in *Daphnia* may have been caused by their slow reproductive rate in conjunction with the rapid flushing.

### 4.3.3.4 Water Column Stability

Water column stability may affect phytoplankton community composition and abundance, as reduced stability may create unfavourable light conditions and/or favour species that can control buoyancy. For example, numerous authors have noted that cyanobacteria dominance and the stability of blooms is related to water column stability (Elser 1999). For example, Zhang and Prepas (1996) in a study of Alberta lakes reported that cyanobacteria were more dominant under conditions of high water stability and diatoms under low water column stability.

### 4.3.3.5 Turbidity/Water Clarity/Light Limitation

Other than nutrients, light is frequently one of the most significant limiting variables for phytoplankton growth (Wang et al. 2005). Light limitation has been reported in Manitoba lakes, including Southern Indian Lake, where neither N nor P appeared limiting to phytoplankton (Healey and Hendzel 1980). Light limitation is also more common in reservoirs than lakes because the former are typically more turbid due to erosion and their location in depositional areas (USEPA 2000a). The precise light conditions that limit phytoplankton growth clearly vary between systems, however, turbidities of $> 200$ NTU have been associated with light limitation and thresholds of $< 5-7$ E/m$^2$/day have been reported for turbid reservoirs (Table 13). Additionally, as the light requirements of phytoplankton species vary, light conditions can affect the community composition of phytoplankton. As indicated in Section 4.4.6, some cyanobacteria species prefer shady conditions and tend to dominate in turbid systems.

### 4.3.3.6 Sediment Resuspension

Sediment resuspension is important in some shallow lakes and is related to a variety of internal and external variables including: wind speed and direction; fetch; water depth; macrophyte biomass; sediment structure and cohesiveness; and, biological factors, such as the presence of certain fish species (e.g., bream, roach, and carp, Schallenberg and Burns 2004). Sediment resuspension may have complex and inconsistent effects on phytoplankton biomass (reviewed in
Schallenberg and Burns 2004). On the one hand, resuspension may increase total nutrients in the water column, reduce light penetration, and increase phytoplankton (Schallenberg and Burns 2004). This pathway may be sufficient to maintain the 'turbid water state' in shallow lakes. Furthermore, photoinhibition of phytoplankton may be reduced in lakes with high water clarity following a resuspension event, thereby increasing phytoplankton biomass.

Conversely, where the depth of the euphotic zone is less than lake depth, sediment resuspension can result in light limiting conditions for phytoplankton (Schallenburg and Burns 2004). Additionally, resuspension may actually decrease the amount of available nutrients to algae, thereby causing nutrient limitation. Schallenberg and Burns (2004) found that while sediment resuspension in a New Zealand Lake generally lead to increased phytoplankton biomass, resuspension did not always alleviate nutrient limitation. Clearly, the relative importance as well as the effect of sediment resuspension is highly site-specific and should be considered on an individual lake basis.

4.3.3.7 Temperature

Temperature has a well established effect on phytoplankton biomass in lakes, most notably in north temperate ecosystems. In a general sense, low temperatures limit biological processes, including phytoplankton growth, which are typically temperature-dependent. More specifically, temperature can alter phytoplankton species composition because phytoplankton species exhibit varying temperature optima and tolerance limits. In some systems, temperature may play a dominant role in controlling phytoplankton biomass (e.g., Abdul-Hussein and Mason 1988 In Lau and Lane 2002; Wang et al. 2005). The significance of temperature is particularly notable with respect to the relative abundance (or dominance) and species composition of cyanobacteria. Zhang and Prepas (1996) found that the most significant factors relating to the dominance of cyanobacteria were high water temperature (>15 °C) and to some extent, water column stabilization.

4.3.4 Top Down vs. Bottom-Up Controls

There has been a great deal of debate of the relative importance of bottom up vs. top down controls on lake trophic condition. As discussed in the preceding sections, nutrients clearly affect lake trophic condition. However, top-down control of lake trophic condition (e.g., phytoplankton biomass) can be highly significant in lakes, and in some cases will be the dominant pathway controlling phytoplankton biomass. This is often true of oligotrophic systems and the prevailing paradigm indicates that top-down controls (i.e., grazing) are most significant in systems with low nutrient concentrations (e.g., Benndorf 1987, 1995 In Lau and Lane 2002).
Conversely, some recent studies have refuted this paradigm demonstrating that top-down controls can be equally important in nutrient rich systems, including deep stratified lakes (e.g., Carpenter et al. 2001). In general, nutrient enrichment results in a reduction in top-down control through the trophic cascade of: increased nutrients; increased phytoplankton and planktivorous fish; increased grazing of zooplankton, notably of larger-bodied daphnids; and reduced grazing pressure, ultimately reinforcing or allowing proliferation and dominance of phytoplankton (Hietala et al. 2004).

In a multi-year whole lake nutrient enrichment study, Carpenter et al. (2001) demonstrated that piscivore-dominated lakes may have lower chlorophyll \( a \) and primary production than planktivore-dominated lakes at the same nutrient concentrations. In fact, an inverse relationship was observed between piscivore biomass and primary production in these lakes. However, the authors noted that grazing was never able to eliminate cyanobacteria blooms in one of the lakes (Carpenter et al. 2001). The occurrence of cyanobacteria blooms is particularly interesting in this case as the nutrient enrichment was conducted using a high N:P ratio, thereby maintaining the pre-enrichment high N:P ratios (i.e., nitrogen was not limiting). Similarly, other researchers have reported that grazing by zooplankton can be a significant limiting factor for phytoplankton in eutrophic lakes (e.g., Lau and Lane 2002). A widely studied example of the significance of top-down controls on phytoplankton is that of Lake Erie. The introduction of zebra mussel (*Dreissena polymorpha*), a filter-feeder, is believed to have significantly reduced phytoplankton biomass in some areas, but not all, of Lake Erie, in conjunction with P loading reductions (e.g., Nicholls and Hopkins 1993, Makarewicz et al. 1999, and Dahl et al. 1995). Zebra mussels are also believed to have significantly altered the surface nutrient chemistry of Lake Erie (Makarewicz et al. 2000).

Overall, it is believed that bottom-up controls remain highly significant in shallow lakes but that the effects of nutrients are modified by top-down controls, including food web structure (Moss et al. 2004). That is, in shallow lakes, bottom-up controls are considered to be more important than top-down controls on grazing (Hietala et al. 2004). Moss et al. (2004) recently demonstrated that nutrients had a larger effect on phytoplankton and zooplankton than fish through an analysis of 11 mesocosm experiments in shallow lakes.

The concept of top-down control on phytoplankton biomass and composition in lakes has been used quite extensively and with relatively good success through lake biomanipulation programs in recent years; that this method is commonly, and successfully, applied attests to the significance of trophic cascades on primary productivity (see Section 4.7 for further discussion).
4.3.5 Alternative Stable States and Shallow Lakes

Many recent studies have demonstrated that shallow lakes may exist in two alternative stable states (reviewed in Schroeder et al. 2005). These states are commonly referred to as the "clear-water state" or "clear state", in which water clarity is high, phytoplankton is low, and macrophytes are significant (typically), and the "turbid state" in which phytoplankton dominate over macrophytes and water clarity is reduced. According to the theory, at high nutrient concentrations (i.e., high TP), a turbid state is maintained, whereas at low nutrients, the clear state is maintained. However, at moderate nutrient concentrations a lake may exist in either ‘alternative’ state (reviewed in Hietala et al. 2004). The transition between the two states is considered to be relatively sudden and stochastic, triggered by various, often catastrophic, events such as weather, shifts in food webs, fish kills, or shifts in lake depth (reviewed in Gonzalez Sagrario et al. 2005, Moss et al. 2005, Hietala et al. 2004).

Recovery of shallow lakes has often found to be delayed such that, although nutrient concentrations may be reduced to target levels, chlorophyll \(a\) and turbidity often lag behind (e.g., Coveney et al. 2005); incomplete recovery (i.e., failure of macrophyte reestablishment) is also commonly observed (e.g., Romo et al. 2005, reviewed in Sondergaard et al. 2000). Many studies have shown that lakes subject to external nutrient reductions have responded with lower nutrient concentrations but the conversion to the clear water state did not occur (e.g., Lake Eymir, Turkey, Beklioglu et al. 2003). That is, lake recovery was incomplete and in this case, as in many others, recovery to the clear water state did not occur until biomanipulation techniques were employed. According to the alternative stable states theory, this is an indication that some factor(s) has prevented the lake from making the transition to a clear water state. The mechanisms for this occurrence are not known but are likely complex and highly variable. An example of this transition is Lake Ringsjön, Sweden where a combination of a fish kill, biomanipulations (i.e., fish removal), low external TP loading, and a decline of TP concentrations below 0.1 mg/L culminated in a shift from a turbid to a clear water state (Bergman 1999).

Macrophytes are commonly cited as one of the requirements for the maintenance of a clear-water state and, according to the prevailing literature, are the key to the transition (e.g., Hietala et al. 2004, Sondergaard et al. 2003). Portielje and Van der Molen (1999) reported that at macrophyte coverage of 5% or more, TN, TP, and chlorophyll \(a\) were significantly reduced in Dutch lakes and ponds. There are several possible explanations for the significance of macrophytes:
• Plants reduce sediment resuspension, and therefore reduce introduction of nutrients;7
• Plants enhance denitrification;
• Plants remove nutrients from the water column and sediments;
• Macrophytes may provide suitable spawning habitat for piscivorous fish; and
• Plants provide refugia for zooplankton and benthic filter feeders, including large effective grazers such as *Daphnia*, thus enhancing the grazer community (Gonzalez Sagrario et al. 2005, Beklioglu 1999).

The presence of planktivorous and benthivorous fish is also critical to the alternative stable states model. High densities of these organisms maintain turbid states in at least two manners: (1) predation on zooplankton reduces grazing pressures, favouring phytoplankton abundance; and, (2) high densities increase sediment resuspension (e.g., Jeppesen et al. 1999) which increases nutrients, turbidity, and chlorophyll *a*, in at least some instances (see Section 4.5 for discussion of sediment resuspension). Additionally, some species may significantly adversely affect macrophytes through physical destruction. Shifts to dominance of planktivorous and benthivorous fish in eutrophic systems tends to favour the turbid state, thereby reducing the resilience of the lake to recovery (Sondergaard et al. 2000). Ultimately, this trophic structure maintains the turbid state (Jeppesen et al. 1999) and may lower the nutrient threshold required to convert to the clear-water state. Biomanipulation has been particularly successful for assisting in the transition from turbid to clear water states in shallow lakes in Europe (e.g., Jeppesen et al. 1999), indicating the significance of fish communities on top-down controls of phytoplankton growth.

Although factors other than nutrients evidently play a significant role in determining which alternative state occurs in a lake, the ability of a lake to convert to a clear-water state is also linked to nutrient concentrations. Several authors have suggested thresholds for the transition; Jeppesen et al. (1990, cited In Gonzalez Sagrario et al. 2005), indicated TP must not exceed 0.08 to 0.15 mg/L to maintain a clear state. More recently, Jeppesen et al. (1999, 2000) indicated a similar threshold of 0.05-0.1 mg/L, which was supported by a survey of 65 Danish lakes. Sondergaard et al. (2001) indicate that there is a strong correlation between phosphorus retention in lakes and the occurrence of the clear water state. That is, P retention increases as shallow lakes shift from the turbid to the clear water state. Proposed mechanisms for this include the positive role of macrophytes in increasing water clarity and retention of nutrients in sediments and benthic algae and the reduced biomass of planktivorous fish which promote sediment

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7 Conversely, some researchers have reported that macrophytes enhance P release from sediments (e.g., Andersen and Ring 1999).
resuspension. Similarly, a concentration of 0.1 mg/L of TP has been suggested as the threshold that facilitates successful fish reductions (i.e., biomanipulation) and increased water clarity (Bergman 1999, Jeppesen et al. 1990 In Bergman 1999).

The role of nitrogen in the transition between turbid and clear water states has been discussed recently by a number of researchers (e.g., Gonzalez Sagrario et al. 2005). It has been suggested that the pivotal point (i.e., the TP concentration) at which the transition occurs during re-oligotrophication may be related to nitrogen concentrations. The adverse effect of nitrogen on macrophytes is one mechanism through which lake recovery may be delayed and/or prevented, as macrophytes are pivotal in the conversion to, and maintenance of, clear water states. Gonzalez Sagrario et al. (2005) for example demonstrated that moderate or high nitrogen concentrations were required in mesocosms to decrease macrophytes. Similarly, Moss (2001 In Gonzalez Sagrario et al. 2005) indicated that macrophyte species diversity is reduced at high TN. Reduced diversity may increase the vulnerability of a lake and therefore, predispose it to the turbid state. Effects of nitrogen are further related to the concentration of TP; effects of nitrogen will be greatest at moderately high TP and lower at low TP (Gonzalez Sagrario et al. 2005).

The TN threshold for conversion to a clear state was estimated as 1.2 - 2 mg/L for Danish lakes (Gonzalez Sagrario et al. 2005). It was cautioned by the authors that this TN threshold is not intended to be universally applicable, as factors such as food web structure may significantly affect shifts between clear and turbid states. If phytoplankton biomass is affected by TN, then TN may affect the presence of macrophytes through promotion of algal growth and subsequent shading effects. Conversely, macrophytes may also reduce TN concentrations, thus presenting what Gonzalez Sagrario et al. (2005) called the "chicken and egg problem". Regardless, the evidence indicates that high N loading reduces the probability that a clear water state is reached.

4.3.6 Reservoirs vs. Lakes

The USEPA (2000a) recommends consideration of lakes and reservoirs separately when developing nutrient criteria, as reservoirs inherently vary in a variety of ways from natural lakes. Reservoirs vary from natural lakes in terms of shape, size, and hydrology and often, in terms of chemistry. Reservoirs formed from damming of streams typically exhibit three ‘zones’: (1) flowing, riverine conditions; (2) a transitional zone intermediate between lotic and lentic conditions; and (3) a zone characterized by 'lake-like' conditions. Many reservoirs are characterized by very short hydraulic residence times and high (non-algal) turbidity. Given these differences, it is not surprising that reservoirs may respond differently to a given nutrient load than a natural lake.
In general, the USEPA (2000a) indicates that nutrient loading models that perform well for natural lakes tend to overestimate the effects of external nutrient loads on nutrient concentrations in reservoirs. This is believed to reflect, at least in some instances, the greater significance of erosion and erosional topography for reservoirs. It is suggested that the larger sediment input typical of most reservoirs leads to increased sedimentation rates and, therefore, increased P settling in sediments. Where reservoirs exhibit long residence times (i.e., years), they are believed to behave more similarly to natural lakes in terms of trophic responses. Therefore, reservoirs with short residence times should be considered distinctly from natural and man-made lakes.

4.4 Factors Affecting Cyanobacteria Abundance

One of the least desirable and often most detrimental responses of eutrophication in lakes is the development of cyanobacteria blooms. In general, as lakes increase in trophic status there is a shift in phytoplankton community composition, with an increased dominance of cyanobacteria. Additionally, cyanobacteria blooms may develop more frequently, earlier in the year, for more prolonged periods, and over larger areas as nutrient enrichment increases. Cyanobacteria blooms are not only associated with the adverse secondary effects typical of algal blooms in general (e.g., aesthetics problems), but also may produce hepatotoxins or neurotoxins that can be harmful to aquatic life, wildlife, and humans. Therefore, there is a general emphasis in the scientific literature on determining the factors favouring cyanobacteria and the identification of potential management options for their control.

Although the correlation between cyanobacteria dominance and eutrophication is clear, the mechanisms behind this competitive edge are not. In general, bloom-forming cyanobacteria are unique as they are relatively large, slow growing, may produce toxins, are generally not controlled by grazing, and can regulate buoyancy (Ferber et al. 2004). There are several current theories postulated to explain this association, namely the low N:P ratio hypothesis, the low DIN hypothesis, water temperature, low light/high turbidity, water column stability, food web effects, and/or some combination of these factors. The precise conditions favouring cyanobacteria blooms are contentious and may vary considerably between lakes and over time in a lake. Several recent reviews or re-examinations of current theories have been published, including Ferber et al. (2004), Levine and Schindler (1999), Elser (1999), Gophen et al. (1999), Hecky and Kilham (1988), and Smith and Bennett (1999) among others. The following is a brief synthesis of the current state of understanding on this issue.
4.4.1 Low N:P Hypothesis

In general, the low N:P hypothesis is predicated on the concept of nutrient stoichiometry and the ability of some bloom-forming cyanobacteria species to fix atmospheric nitrogen. Many studies have related the occurrence and dominance of cyanobacteria to low N:P ratios in lakes (e.g., Smith 1983, Schindler 1977, Scheffer et al. 1997, Bulgakov and Levich 1999, reviewed in Elser 1999). As indicated in Section 4.3.2, this is explained by the competitive edge gained by nitrogen-fixing cyanobacteria under nitrogen-limiting conditions. Evidence for this theory is provided by laboratory, mesocosm, and whole-lake experiments as well as through statistical analysis of natural lakes (e.g., Gophen et al. 1999, reviewed in Pick and Lean 1987, reviewed in Elser 1999, Barica et al. 1980). The precise ratio of N:P that favours cyanobacteria is likely to vary between lakes (Pick and Lean 1987), but common ratios are in the range of 22 to 29 by mass (Smith 1983, Smith and Bennett 1999).

The original theory was first proposed over 70 years ago by Pearseall (1932 in Ferber et al. 2004) but likely the most widely cited studies in support of the low N:P hypothesis are those of the whole lake nutrient enrichment experiments conducted at ELA in the boreal shield of Ontario. Nutrient additions and reduction of lake TN:TP ratios caused a shift in the phytoplankton communities to cyanobacteria dominance and nitrogen fixation in Lake 227 (reviewed in Hecky and Kilham 1988 and Elser 1999). Hendzel et al. (1994) also demonstrated that the elimination of experimental N loading (but not P loading) to Lake 227, reduced the N:P ratio and resulted in consistent dominance of nitrogen-fixing cyanobacteria. Findlay et al. (1994) indicated that blooms of nitrogen-fixing *Aphanizomenon* occurred annually in Lake 227 after the nutrient loading ratio was reduced to 5:1. Since that time, many studies have supported the low N:P hypothesis (e.g., Burgi and Stadelmann 2002; Tonno and Noges 2003).

As a corollary, several authors have attempted to directly relate the abundance/biomass of cyanobacteria in lakes to TP concentrations (reviewed in Pick and Lean 1987). These attempts have been met with varying success. Watson et al. (1997 in Moustaka-Gouni et al. 2006) reported that cyanophyta comprise a minimum of 70% of the total phytoplankton biomass in lakes when the TP concentration exceeds 0.1 mg/L. Trimbee and Prepas (in Pick and Lean 1987) found that TP was the best predictor of cyanobacteria biomass, surpassing even the N:P ratio. A number of studies have also demonstrated that the relative abundance of cyanobacteria increases with increasing total phytoplankton biomass (reviewed in Pick and Lean 1987) and that heterocystous cyanobacteria abundance increases with TP and decreases with TN:TP (Tonno and Noges 2003).
The low N:P ratio hypothesis has been criticized for a number of reasons, including:

- the occurrence of cyanobacterial dominance at high ratios;
- increases in N:P ratios have failed to prevent blooms of cyanobacteria;
- N-limitation does not always correlate to cyanobacterial abundance;
- the relationship between cyanobacterial dominance and N:P ratio is incidental (i.e., is only correlative and does not show cause and effect); and
- dominance of other species of algae at low N:P ratios may occur.

Not all cyanobacteria species prefer or are restricted to low ratios and some species of cyanobacteria have been found to occur over a wide range of TN:TP ratios (reviewed in Pick and Lean 1987). Pick and Lean (1987) concluded that based on laboratory studies there is "no conclusive evidence that N:P ratios play a major role in cyanobacterial dominance. Low N:P ratios are found for algae of other classes (e.g., Melosira) and there is considerable variability in the N:P ratios reported for well-studied cyanobacterial species". For example, Microcystis may occur at varying TN:TP ratios and still other species may occur more frequently at higher TN:TP ratios (e.g., Oscillatoria and Lyngbya, reviewed in Pick and Lean 1987). Other studies have shown that alterations to the N:P ratio may eliminate N-fixing blooms but only to have them replaced by non-nitrogen fixing cyanobacteria bloom-forming species. For example, Barica et al. (1980) eliminated the occurrence of Aphanizomenon blooms in a prairie pond by the addition of nitrogen, but these blooms were replaced by Microcystis. Similarly, Stockner and Shortreed (1988) eliminated blooms of Anabaena through increasing TN:TP ratios and nutrient enrichment in coastal BC lakes but the Anabaena blooms were replaced by Synechococcus. It was suggested that Synechococcus responds strongly to both N and P enrichment, but tends to dominate at higher N:P ratios. Similarly, Gonzalez Sagrario et al. (2005) found that non N-fixing cyanobacteria were most abundant in mesocosms containing high nitrogen and high phosphorus/moderate nitrogen, but not in cells enriched only with P. As an extension to this, no relationship between the nutrient ratio and cyanobacterial dominance was observed.

Additionally, numerous studies have reported that high N:P ratios have not prevented cyanobacteria blooms and/or that low N:P ratios do not always correlate to cyanobacterial dominance. Scheffer et al. (1997) reported a lack of correlation between TN:TP ratios and the dominance of a non-nitrogen fixing cyanobacteria (i.e., Oscillatoria) in 55 shallow Dutch lakes. In recent whole-lake nutrient enrichment experiments, Carpenter et al. (2001) demonstrated that high N:P ratios and the occurrence of phosphorus limiting conditions in enriched lakes did not prevent the occurrence of cyanobacteria.
Studies also refute nitrogen-limiting conditions as the main driver for cyanobacteria dominance. For example, Jensen et al. (1994 In Scheffer et al. 1997) reported no relationship between the abundance of heterocystous cyanobacteria and nitrogen availability in 210 shallow Danish lakes. Similarly, Smith (1985) found that TN was an equally good predictor of cyanobacteria biomass as TP and low TN:TP ratios.

Several studies have shown that changes to nutrient loading and subsequently N:P ratios in lakes, which should, according to the prevailing theory, reduce cyanobacteria have failed to affect cyanobacteria abundance. For example, in the Bay of Quinte, Lake Ontario, reductions in TP loading did not alter the relative abundance of cyanobacteria, although it did cause a reduction in the total biomass of all phytoplankton (reviewed In Pick and Lean 1987). A similar effect was reported in Kootenay Lake, BC (reviewed In Pick and Lean 1987). In other cases, cyanobacteria biomass has been reported to increase with TN:TP ratios; this occurred in Lake St. George, ON, where the relative abundance of cyanobacteria increased in association with an increase in TN:TP from <20 to >80 (reviewed In Pick and Lean 1987). In the latter case, temperature was found to be a factor.

Because low N:P ratios typically occur in nutrient-enriched systems, it has been suggested that this observed correlation is incidental and not mechanistic (e.g., Downing et al. 2001). In other words, could some factor(s) other than a low ratio that is also associated with nutrient enrichment and eutrophication be the ultimate cause of cyanobacterial dominance? One such factor that has been suggested is turbidity. For example, water clarity typically declines in eutrophic systems and cyanobacteria prefer low light environments. Dominance of cyanobacteria has been attributed to turbid conditions in some lakes (see Section 4.4.6 for further discussion). Furthermore, cyanobacteria create a positive feedback loop by further reducing light levels through self-shading, thereby reinforcing their competitive advantage in eutrophic systems (Scheffer et al. 1997). Downing et al. (2001) analysed data from 99 temperate zone lakes and found that the “risk of degradation by Cyanobacteria blooms” is more strongly correlated with variability in TP, TN, or the standing biomass of algae than the N:P ratio. TN was more strongly correlated to cyanobacteria dominance than TP.

There is evidence from some studies that phytoplankton other than cyanobacteria have been dominant in lakes with low TN:TP ratios (reviewed In Pick and Lean 1987). For example, diatoms and chlorophytes dominated at TN:TP ratios below 12 (by weight) in the widely cited study of Japanese lakes by Sakamoto (1966). Similarly, Jensen et al. (1994 In Gonzalez Sagrario et al. 2005) found that green algae dominated in shallow, north temperate lakes under high nutrient loading.
4.4.2 Modified Low N:P Hypothesis

Much of the evidence opposing the long-held low N:P hypothesis, which is discussed above, can be explained through a slight modification to the current theory. Specifically, the modified theory specifies that low N:P ratios favour nitrogen-fixing cyanobacteria whereas non-nitrogen-fixing cyanobacteria are not favoured under a low ratio due to their inherent inability to sequester atmospheric nitrogen. This modification is consistent with much of the literature, including the ELA whole-lake experiments and algal stoichiometry (e.g., Hendzel et al. 1994). It has been suggested that much of the trouble applying the conventional low N:P hypothesis is that many researchers have applied it to all cyanobacteria and not solely to those species that can actually fix nitrogen. Levine and Schindler (1999) recommended that the original low N:P hypothesis be adjusted in this manner, such that it its application be restricted to nitrogen-fixing species. They demonstrated that non-nitrogen fixing cyanobacteria (Oscillatoria, Pseudoanabaena and Lyngbya) increased along an N:P gradient in mesocosm experiments in Ontario and they concluded that "clearly, the low-N:P theory of cyanobacterial dominance is appropriate only for N2-fixing species".

Conceptually, this modified hypothesis explains a great deal of the discrepancy in the literature. For example, studies have shown that low N:P ratios favoured nitrogen fixers but as the ratio is increased, blooms of these species may be replaced by non-nitrogen fixers such as Microcystis. Similarly, the lack of correlations found between cyanobacterial dominance and TN:TP ratios in some studies, may also be explained by the consideration of both nitrogen and non-nitrogen fixing species collectively (reviewed in Ferber et al. 2004). This modification to the theory is highly significant from a management perspective as it emphasizes the need to consider nitrogen as a factor controlling the occurrence of nuisance and most importantly, potentially toxin-forming, blooms of cyanobacteria. That is, sole manipulation of phosphorus loading and controls and/or manipulation of N:P ratios may not be adequate to prevent the occurrence of noxious cyanobacterial blooms in lakes. Conversely, Levine and Schindler (1999) indicate that reductions of P loads remain the best approach for controlling cyanobacteria because both heterocystous and non-heterocystous species are favoured by high P. Lawrence et al. (2000) suggested that non-heterocystous cyanobacteria would dominate over heterocystous species where the inorganic nitrogen concentration were greater than 0.1 mg/L, which is consistent with this theory.

Deep and shallow lakes may require distinct considerations in this regard. Deep lakes typically stratify and are generally dominated by nitrogen-fixing cyanobacteria, whereas shallow eutrophic lakes are typically dominated by non-nitrogen-fixing cyanobacteria such as Oscillatoria (Havens et al. 2003). As a corollary, it can be suggested that shallow lakes are either less prone to
development of nitrogen limitation and/or that nitrogen-fixing species are not as competitive in shallow lakes. One suggestion is the high tolerance for low light levels by the non-nitrogen fixing species, as shallow lakes are typically more turbid (Scheffer et al. 1997). Regardless, of the explanation, this occurrence suggests that the modified low N:P ratio requires consideration in lake management.

### 4.4.3 Low DIN Hypothesis

A relatively newer and less commonly discussed hypothesis is the low DIN hypothesis. In this instance, the relative dominance of heterocystous vs. non-heterocystous cyanobacteria is determined by the presence of a benthic ammonia source when DIN is depleted in the epilimnion, but the central key to the hypothesis is the dominance of cyanobacteria at low DIN (Blomqvist et al. 1994 and Hyenstreand et al. 1998a,b all In Ferber et al. 2004). Vacuolated filamentous or colonial genus' such as *Microcystis* and *Oscillatoria* are said to be at a competitive advantage relative to flagellated algae as they can rapidly and relatively 'cheaply' vertically migrate down to depths to take advantage of benthic ammonia sources (Ferber et al. 2004). These species may then grow more rapidly than their heterocystous counterparts that remain at the water surface. Ultimately, according to this theory, heterocystous species would dominate only when DIN sources are inadequate both at the surface and in benthic supplies. However, as Ferber et al. (2004) point out, many heterocystous species may also migrate vertically.

Ferber et al. (2004) found a strong dominance by non-heterocystous cyanobacteria in a hypertrophic lake (Shelburne Pond, VT), despite high TN:TP ratios. They found overall, that the occurrence of nitrogen fixation was small (< 2%) and that the principle source of nitrogen was ammonia. This relationship held even when the community was dominated by heterocystous cyanobacteria. Ferber et al. (2004) suggested that their study supported the low DIN hypothesis as the presence and abundance of cyanobacteria taxa were consistent with the relative sources of DIN but recommended modifying the hypothesis to include all migrating cyanobacteria and inclusion of consideration of P as well as nitrogen. Ferber et al. (2004) suggest that although significant in some cases, nitrogen fixation typically represents a small fraction of the overall nitrogen cycling in lakes (< 10%). They argue that failure to express nitrogen fixation loading in the context of total nitrogen accumulated by algae results in a skewed representation of atmospheric nitrogen sources (i.e., need to consider nitrogen cycling).
4.4.4 Nutrient Thresholds

Several authors have related the probability or actual occurrence of cyanobacterial dominance to nutrient thresholds (Table 4). TP thresholds vary from 0.01 to 0.1 mg/L. Nitrogen thresholds are not as commonly identified but levels of TN of < 1.35 mg/L in conjunction with TP < 0.05 mg/L were reported to be associated with the disappearance of cyanobacterial dominance in 231 lakes and ponds in the Netherlands (Portielje and Van der Molen 1999).

4.4.5 Temperature

Temperature is clearly an important parameter governing the relative abundance of cyanobacteria as they generally prefer warmer temperatures than other taxa (e.g., Scheffer et al. 1997, Pick and Lean 1987, Smith and Bennett 1999, Zhang and Prepas 1996); several authors have identified critical temperatures that are determinants for cyanobacteria abundance (Table 4). For example, cyanobacteria have been reported to dominate when the temperature exceeds 20 ºC (reviewed in Pick and Lean 1987). Temperature appears to be one of the critical factors determining the likelihood of bloom development; it has been reported that cyanobacteria dominate in lakes with low TN:TP ratios when the temperature is high. It is often reported that cyanobacteria blooms correspond to either a particularly warm, calm period of summer and/or during a particularly warm year (e.g., Soininen et al. 2005). The thermotolerance of cyanobacteria also appears to be modified by nutrients, thus further securing their dominance under nutrient enrichment. For example, Scheffer et al. (1997) reported that cyanobacterial dominance is less sensitive to low temperatures at high nutrient concentrations; thus, cyanobacterial dominance may become more frequent and/or prolonged under high nutrient concentrations.

4.4.6 Turbidity/Light Levels

There is a fair amount of evidence to support the 'high turbidity theory' for cyanobacterial dominance in lakes. Cyanobacteria are favoured by low light levels and can have a competitive advantage in lakes with high turbidity. As eutrophic systems and shallow lakes tend to have reduced water clarity, it is suggested that cyanobacteria tend to have an increased edge in these environments. Furthermore, cyanobacteria that form blooms are also able to control buoyancy through the presence of gas vacuoles, thus allowing them to remain near the surface and maximize light exposure. The high tolerance of cyanobacteria for low light creates a further advantage for bloom-forming species as they are able to tolerate self-shading while reducing light availability for species lower in the water column.

There is mounting evidence that shallow, turbid lakes may be predisposed to dominance by non-heterocystous cyanobacteria taxa, relative to deeper, clearer lakes. Dominance of certain species
of cyanobacteria has been related to turbidity (e.g., Scheffer et al. 1997) and turbidity is generally higher in shallow lakes. Scheffer et al. (1997) reported a lack of correlation between TP, TN, or TN:TP ratios and the relative abundance of the dominant cyanobacteria *Oscillatoria* in 55 shallow Dutch lakes. Rather, the authors reported a correlation between shady conditions and the abundance of this species. Scheffer et al. (1997) suggested that this 'turbidity theory' was supported by the results of Smith (1986) and Jensen et al. (1994) in studies of lakes worldwide and in Denmark, respectively. Jensen et al. (1994), for example, reported a much stronger correlation between shady conditions and cyanobacteria than TP. Other evidence supporting this theory is provided by lakes Schlachtensee and Veluwemeer in which responses to nutrient reductions occurred at different nutrient concentrations but the decline in *Oscillatoria* occurred at similar light conditions in both lakes (reviewed in Scheffer et al. 1997). Smith (1986) similarly indicated that when TN:TP are constant, cyanobacteria will be favoured by reduced light availability. Scheffer et al. (1997) went as far as to suggest that elimination of *Oscillatoria* from lakes with high background turbidity will be difficult. Havens et al. (2003) suggested that phytoplankton growth in Lake Okeechobee (FA) is light limited and that reduced water clarity is due primarily to mineral turbidity (possibly related to sediment resuspension). Additionally, they suggested that turbidity affects the relative dominance of nitrogen-fixing vs. non-fixing cyanobacteria, with lower water clarity favouring *Oscillatoria*. Berger (1989 in Havens et al. 2003) suggested that *Oscillatoria* would dominate in lakes with a low Secchi depth and a mean depth < 3 m.

### 4.4.7 Hydrology and Lake Morphometry

Lake morphometry, which is related to lake depth, flushing rates, stratification/mixing, may play a role in determining the relative abundance of cyanobacteria. In general, there is evidence that cyanobacteria are favoured by shallow lake depths, low flushing rates, and low water column stability.

Shallow lakes may be predisposed to cyanobacterial dominance. Smith (1985), for example, reported that cyanobacteria biomass was inversely related to lake depth. High flushing rates may also limit cyanobacteria; sufficiently high flushing rates will prevent bloom development if the slow-growing cyanobacteria do not have sufficient time to grow (Scheffer et al. 1997). In a recent review, Smith and Bennett (1999) presented the work of Reckhow (1987) in which an equation was developed that positively related hydraulic retention time to the probability of cyanobacterial dominance.

Cyanobacteria also thrive under high water column stability, with some inter-specific variability regarding the precise preferred conditions (e.g., Elser 1999, Havens et al. 2003, Jacquet et al. 2005, reviewed in Pick and Lean 1987, Smith 2001). Zhang and Prepas (1996) found that the
most significant factors relating to the dominance of cyanobacteria were high water temperature (> 15 °C) and water column stabilization in mesotrophic to hypereutrophic Alberta lakes. They noted, however, that cyanobacteria dominated in four Alberta lakes under a range of water column stabilities. Wang et al. (2005) reported that the occurrence of cyanobacterial blooms coincided with a brief period of stratification in Lake Clinton (a reservoir), KA, during a period of relatively constant water level and near zero discharge from the reservoir. When outflow and water level were rapidly increased, blooms were disrupted. The significance of water column stability was effectively demonstrated by Jacquet et al. (2005) in a comparison of Lake Geneva and Lac du Bourget. Both have similar nutrient concentrations and phytoplankton species richness but Lac du Bourget is dominated by a noxious species of cyanobacteria (*Planktothrix rubescens*) whereas Lake Geneva is not. This discrepancy was explained by the higher water column stability of Lac du Bourget. In general, strong wind can disrupt cyanobacteria blooms in lakes (Jacquet et al. 2005).

### 4.4.8 Chemical Conditions

Chemical conditions other than nutrients may also affect cyanobacterial abundance including pH and ionic composition of lakes. Numerous studies have reported that high pH favours cyanobacterial dominance in lakes. However, pH is not the favoured hypothesis for cyanobacterial dominance in the scientific literature. Recently, Willame et al. (2005) reported that the taxonomic composition of cyanobacterial blooms in Belgium and Luxembourg were related (primarily) to the local geology and water chemistry (excluding nutrients) of the lakes. For example, *Aphanizomenon* reportedly thrives in waters of high ionic strength. Furthermore, Willame et al. (2005) indicated that there is a general north-south pattern across Europe, with heterocystous species typically dominating cyanobacterial blooms in northern Europe, with *Planktothrix* dominating in northern Germany, and *Microcystis* dominating in central and southern Europe. This pattern may reflect differences in geology and water chemistry.

### 4.4.9 Cyanobacteria Blooms: Multi-factor and Food Web Effects

The conflicting literature summarized above may be made somewhat more congruous if one accepts that cyanobacteria blooms are a result of the convergence of multiple factors (one of which is high nutrients) that are conducive to bloom development. Elser (1999) indicated that the pathway leading to blooms can be described in the following relationship:

1. **nutrient loading**: levels of nutrient loading and subsequent nutrient concentrations in lakes will affect the ability of blooms of any kind to develop (i.e., low nutrients will prevent blooms);
2. **nutrient loading N:P ratio**: if nutrient loading is sufficiently high, and the N:P ratio in the inflows is low, cyanobacteria will be favoured;

3. **hydrodynamic effects**: when loading is high and N:P is low, other factors may determine whether a bloom will develop. These include temperature, water column stability, and light intensity; and

4. **foodweb structure and stoichiometric feedbacks**: even when all aforementioned conditions are favourable to cyanobacteria, trophic cascades can affect bloom formations. Food web structure that supports the dominance of large-bodied zooplankton leads to reductions in algal biomass through grazing.

There has also been some interesting work demonstrating the effects of zooplankton on the N:P ratio and cyanobacteria. As reviewed in Elser (1999), studies have shown that *Daphnia* may shift the dominance of cyanobacteria through grazing and shifts in the epilimnetic N:P ratio. Larger *Daphnia*, that are typically more abundant where there are substantial number of piscivorous fish, have a lower N:P ratio than their food source and subsequently excrete excess nitrogen, shifting the external N:P ratio upwards. This can influence the local pool of bioavailable nutrients and the N:P ratio in favour of phosphorus limitation. These studies emphasize the need to consider the food web when assessing eutrophication and potential management options.

Overall, cyanobacteria bloom species are not grazed significantly by zooplankton due to the general low abundance of large zooplankters in eutrophic systems (e.g., Carpenter et al. 2001), the general low quality of cyanobacteria as a food source (i.e., the inedibility of cyanobacteria), and/or the production of algal microtoxins, all of which reduce grazing (Smith 2001). Cyanobacteria blooms may be correlated to a reduction in grazing pressure by *Daphnia* (e.g., Portielje and Van der Molen 1999). Smith (2001) indicated that colonial cyanobacteria, such as *Aphanizomenon*, are too large for *Daphnia* to graze, and dominance of zooplankton communities by *Daphnia* can lead to dominance of this cyanobacteria.

Macrophytes have been shown to produce growth-inhibiting substances that affect primarily cyanobacteria (reviewed in Scheffer et al. 1997). Thus, macrophytes may add an increased resistance to the effects of nutrient enrichment and development of nuisance cyanobacteria blooms.

Jacquet et al. (2005) provide a multi-factor explanation for the proliferation of *Planktothrix rubescens* in Lac du Bourget, the largest, and relatively deep lake in France, following
substantial reductions in TP concentrations (from 0.12 to 0.026 mg/L). They postulate that dominance was the combined result of the following, all of which favour *P. rubescens*:

- Reduced TP which favoured *P. rubescens* as it can use organic P;
- Warmer springs, leading to establishment of early stratification; and
- Reduced surface irradiance.

It is also noteworthy that the external nutrient reductions resulted in an increased TN:TP as the program focused on P reductions. Despite the high TN:TP, noxious cyanobacterial blooms persisted. In fact, Jacquet et al. (2005) found that a bloom of *P. rubescens* was preceded by PO₄ depletion; this is explained by the ability of *P. rubescens* to utilize organic P. This is a highly significant finding from the perspective of lake management and nutrient reduction strategies as this case indicates that P loading reduction may be inadequate to prevent the occurrence of cyanobacteria blooms in some lakes. Furthermore, the low light preference of this species (and other cyanobacteria) suggests that turbid systems may be predisposed to blooms.

### 4.4.10 Synthesis

Overall, the literature indicates that a host of factors may affect cyanobacteria dominance, some of which are either a direct result of or correlated to nutrient enrichment, while others are uncoupled from nutrient enrichment (e.g., temperature), and that the controlling variables vary in time, space, and between lakes. Furthermore, the predictability of the occurrence of cyanobacteria blooms is not an exact science, but rather can be best described as a probabilistic relationship whereby blooms are more likely to form under certain scenarios. A single factor does not consistently explain the dominance of cyanobacteria (e.g., Smith 1983), and Pick and Lean (1987) caution that controlling phosphorus and the TN:TP ratio may not be an effective means of controlling some cyanobacteria species (e.g., *Microcystis*).

### 4.5 INTERNAL LOADING AND NUTRIENT RETENTION

Sediments typically serve as ‘sinks’ for nutrients and other substances, including contaminants, and lakes therefore retain nutrients. Phosphorus settles, most notably in particulate forms, to lake sediments and is precipitated as insoluble iron, calcium, or aluminum phosphates. However, under certain conditions, sediments may become net sources of nutrients. Internal loading refers to the release of nutrients (typically phosphorus) from the sediment to the overlying water column, thereby effectively behaving as an ‘internal’ nutrient load. Physicochemical and biological conditions at the sediment-water interface that affect the occurrence of internal loading are:
• phosphorus saturation of sediments;
• low DO conditions (e.g., hypolimnetic oxygen depletion that occurs in some stratified lakes);
• elevated temperatures;
• reducing conditions;
• turbulence;
• pH and temperature;
• biological activities of sediment biota (i.e., bioturbation); and
• iron availability.

The relative importance of the above factors in controlling internal loading and/or the extent of internal loading is highly site-specific. For example, lakes with an abundant benthic fauna may be relatively more affected by bioturbation of sediments than deeper lakes with a less abundant community. However, some ‘generalizations’ can be made regarding the relative vulnerability of lake types to the occurrence of internal loading.

The conventional view once was that internal loading was primarily associated with deep lakes that naturally stratify and that develop anoxic or hypoxic hypolimnions (reviewed in Sondergaard et al. 2001; 2003). However, we now know that internal loading is quite significant in shallow lakes and in some systems can be the largest source of nutrients (reviewed in Sondergaard et al. 2001). P release from sediments appears to occur through two primary mechanisms in shallow lakes: (1) diffusion; and, (2) sediment resuspension. It has been shown that fully mixed shallow lakes can experience significant internal loading even under aerobic conditions (Sondergaard et al. 2003). Decomposition of organic materials at the sediment water interface reduces the depth of the oxic sediment layer (i.e., to a few mm), thus compromising the ability of sediments to retain phosphorus and it is released through diffusion. Furthermore, there is some evidence that internal P loading is greater in littoral than profundal zones (Andersen and Ring 1999), which may predispose shallow lakes to greater overall internal loading. In the boreal plains of Canada, P release has been shown to be greater in lakes during periods of anoxia (e.g., Burley et al. 2001).

Additionally, shallow lakes and or the nearshore areas of lakes may also experience wind-driven sediment resuspension that can also introduce significant quantities of nutrients into the water column (e.g., Sondergard et al. 2001, James et al. 2005, reviewed in Sondergaard et al. 2001). Conversely, there have also been reports that sediment resuspension is not significant enough to
delay lake recovery following external loading reductions (e.g., Coveney et al. 2005, Jeppesen et al. 2003).

Various factors at the sediment water interface may affect the release and binding of nutrients in sediments. One of the most significant factors affecting the sorption/desorption of phosphorus in lake sediments is the oxido-reduction potential (i.e., “redox” conditions), in conjunction with low dissolved oxygen. The critical threshold below which phosphorus that is normally bound in the sediments is released to the overlying water column (i.e., “internal loading”) is 200 mV (Kalff 2002). Additionally, nitrate may significantly affect the sorption and release of P in lakes sediments, although the relationship is not linear. At moderate concentrations nitrate can affect redox conditions and enhance P retention whereas at high concentrations, nitrate may promote release (Gonzalez Sagrario et al. 2005).

The effects of bioturbation on sediment resuspension and release of nutrients from sediments may also be significant in some systems (reviewed In Sondergaard et al. 2001, Kelton and Chow-Fraser 2005). In particular, a high density of benthic and planktivorous fish may be a significant conduit for sediment resuspension and subsequent effects to nutrients, water clarity, and phytoplankton (see Section 4.3.5 for further discussion). Shallow lakes may be further predisposed to sediment influences due to their inherently high surface area: volume ratios (Salomons et al. 1987 In EC 2004) with high sediment surface areas.

Internal loading may be a highly significant source of nutrients to lakes and can result in the maintenance of high nutrients in the water column long after external loading has been reduced (e.g., Lau and Lane 2002, Annadotter et al. 1999, Wilander and Perssson 2001, Sondergaard et al. 2003). There are many reports that internal loading may significantly delay the effects of reduced external nutrient loading in lakes worldwide (e.g., EEA 2005, Marsden 1989, Lau and Lane 2002, Annadotter et al. 1999, Herodek 2002) and shallow lakes appear to be the most sensitive to this occurrence (Jeppesen et al. 1991 In Phillips et al. 2005, Welch and Cooke 1995, Sondergard et al. 2001). Jeppesen et al. (2005b) hypothesized that the delay in reductions of in-lake TP when phosphorus loading is reduced are related to lake retention times; they suggested that a minimum of three times the lake retention period is required to remove 95% of the excess P pool in mixed lakes. Delays are often 10-15 years, although delays of decades have also been observed (Phillips et al. 2005, Wilander and Persson 2001, reviewed In Jeppesen et al. 1999).

In extreme cases, a lake's capacity to assimilate phosphorus (i.e., through sedimentation and retention) may be affected such that the lake shifts from a net sink of phosphorus to a net source (e.g., Lake Hjarlmar, Sweden, Wilander and Persson 2001). For example, Lake Sobygard, Denmark has exhibited a very low resiliency to a high (90%) external TP load reduction such
that 13 years later, internal loading exceeded external loading. It has been suggested that 10 more years would be required before an equilibrium were reached (Sondergaard et al. 2003). This is particularly noteworthy because the lake is characterized by a short retention time (15-25 days).

There are strong indications that the assimilative capacity of Lake Okeechobee, FA has been overwhelmed, resulting in the lake serving as a net P source. Despite significant reductions in external P loads, P concentrations actually increased in the lake (Havens and James 2005). Two signs of the loss of assimilative capacity were the convergence of lake and inflow P concentrations and the decline in the net sedimentation of P in the lake. It was suggested that the reduced external loading may have caused the observed increase in internal loading (i.e., reduced sedimentation coefficient). Alternatively, the lake may have reached a critical threshold where the sediments had reached saturation levels (i.e., a critical tipping point was reached). Kohler et al. (2005) also suggested that external nutrient reductions lead to conversion of Lake Muggelsee, Germany from a net P sink to a net source. They postulated that reductions in nitrate loading favoured release of P from sediments.

In general, the ability of a lake to retain phosphorus is positively related to its retention time, the abundance of iron, and its biological structure (Sondergaard et al. 2003). The occurrence of a clear water state in a lake also enhances nutrient retention and Sondergaard et al. (2003) stated that "establishment of clearwater conditions is the most important factor for obtaining a high phosphorus retention capacity." Phosphorus retention capacities also change with shifts in external loading. For example, Sondergaard et al. (2003) indicated that after external loading reduction and a period of decline in lake nutrient concentrations, the equilibrium between sediments and water is disrupted and the lake switches from its role as a sink to a source of phosphorus. The speed with which a lake establishes a new equilibrium is proportional to the duration of the period of elevated external nutrient loading. Furthermore, predicting effects of internal loading following external loading reductions, based on current rates is not useful (Van der Molen and Boers 1994).

Internal loading appears to vary over the year, typically being greatest in summer in shallow temperate lakes (reviewed in Sondergaard et al. 2001 and Welch and Cooke 2005, Sondergaard et al. 2003). Sondergaard et al. (2003) indicated that the capacity of lake sediments to retain phosphorus is reduced in summer due to changes in the mineralization process and the depth of the oxic sediment layer. As a result, the highest TP concentrations often occur in August in Danish lakes. Similarly, delays in lake recovery are often reported to be seasonal. TP may rapidly decline following external loading reductions in spring, but summer TP concentrations
often remain unchanged (Kohler et al. 2005). This is most often explained as a consequence of internal loading which is typically greatest in summer.

The period of internal loading also appears to vary according to the nutrient status of the lake. Sondergaard et al. (1999) found that the period of sediment phosphorus release is a function of TP in shallow Danish lakes; at TP < 0.1 mg/L release occurs in July and August but where TP >0.2 mg/L release occurs from April through September. Similarly, Sondergaard et al. (2002) found that the period of internal loading decreased in shallow Danish lakes following external loading reductions. As a corollary, during lake recovery, rapid changes often occur in spring but internal loading continues to supply P in summer until a new equilibrium is reached between the sediment and water (Phillips et al. 2005).

Unlike the various forms of external loading to a lake, which are more readily manageable, internal phosphorus loading is not. Mitigation to reduce internal loading can be undertaken but is more challenging and efforts have met with varying levels of success. Remediation methods can include withdrawal of hypolimnetic water (e.g., Pine Lake, AB, Sosiak 1997), lake aeration (e.g., Amisk Lake, AB, Prepas and Burke 1997, Lake Sempach, Switzerland, Burgi and Stadelmann 2002, Lake Hald, Denmark, Jeppesen et al. 1999), sediment dredging (e.g., Lake Finjasjon, Sweden, Annadotter et al. 1999, Barton Broad, England, Phillips et al. 2005, Lake Braband, Denmark, Jeppesen et al. 1999,), chemical controls, such as nitrate injections (e.g., Lake Lyng, Denmark, Jeppesen et al. 1999) or aluminum (e.g., Vadnais Lake, Minnesota, reviewed In Welch and Cooke 2005) or iron compound treatments (Engstrom 2005), sediment capping (e.g., Lake Arendsee, Germany, Hupfer and Lewandowski 2005), and/or biomanipulation (e.g., Lake Finjasjon, Sweden, Annadotter et al. 1999).

Most discussions of internal loading in the scientific literature deal mostly or exclusively with the issue of phosphorus release from sediments. However, sediments may also be a significant source of nitrogen to the overlying water. Organic nitrogen, present for example in detritus in sediments, is mineralized to ammonia which can result in accumulations of ammonia in pore waters and bottom waters of lakes and reservoirs (e.g., Lawrence et al. 2000). Conversely, nitrate can be consumed by denitrifying bacteria, thereby reducing N in lakes (Lawrence et al. 2000). Nowlin et al. (2005) found that ammonia fluxes from sediments did not differ between lakes with differing mixing characteristics and oxygenation.

Sediment fluxes of nitrogen are important because they release readily bioavailable forms of nitrogen and in some lakes this source is very significant, particularly in terms of seasonality. However, the precise contribution and role of regeneration of inorganic nitrogen in sediments varies between lakes. According to Jensen et al. (1992 In Kozerski et al. 1999), the overall
contribution of internal nitrogen loading to the nitrogen budget of a lake is small because
denitrification losses balance releases of nitrogen to the water column. Conversely, James et al. (2005) reported that although Lake Okeechobee sediments were a net sink for TN, sediment fluxes of inorganic nitrogen were very important, comprising approximately 47% of the total inorganic nitrogen flux to the water column, or 63% of external loads. A similar conclusion was reached by Fisher et al. (2005) for Lake Okeechobee; the estimated internal load of DIN in the lake in 1999 was greater than the external loads for the period of 1989-1998. O'Brien et al. (2005) found that fluxes of inorganic nitrogen from sediments increased rapidly in an artificially enriched arctic lake following nutrient addition and continued to release significant quantities after cessation of treatment.

4.6 NUTRIENT CRITERIA

Various jurisdictions in Canada, including national and provincial agencies, as well as worldwide organizations have suggested or adopted nutrient criteria for lakes for the prevention of nuisance plant or algal growth. In many instances, the criteria are generic and have not been derived for site-specific waterbodies or regions, beyond the provincial or national scales. A summary of current provincial and CCME guidelines are provided in Table 15. As is evident from the summary of available nutrient criteria in Canada, most guidelines are restricted to phosphorus and no guidelines have been adopted for other indicators of trophic condition such as Secchi disk depth or chlorophyll \(a\).

Site-specific nutrient criteria (i.e., for a portion of a lake, a whole lake, or lakes in an ecoregion or ecozone) have been proposed or adopted by various agencies. It is generally advocated to develop site-specific nutrient criteria for a given lake, area of a lake, or a particular region (e.g., ecoregion). This advocation is based on decades of experience and accumulation of scientific information on nutrient enrichment and eutrophication in lakes, as well as the general acknowledgement that ‘baseline’ or ‘pristine’ nutrient conditions and trophic status varies considerably between lakes. There is general agreement that lakes do not respond consistently to nutrient enrichment and the ‘natural’ nutrient conditions in a lake vary considerably according to internal (e.g., lake morphologies) and external characteristics (e.g., soils, geology, climate, hydrology, drainage basin characteristics). Therefore, ‘natural’ nutrient concentrations for lakes should be defined on a case-by-case basis. The provinces of British Columbia and Ontario have reportedly established ‘reference conditions’ for phosphorus (EC 2004). Similarly, Quebec has recently evaluated the applicability and feasibility of developing and using ecoregional phosphorus criteria (Berryman 2006).
Table 15. Summary of provincial and national nutrient guidelines for lakes in Canada.

<table>
<thead>
<tr>
<th>Province</th>
<th>Usage</th>
<th>TP (mg/L)</th>
<th>TN (mg/L)</th>
<th>Chlorophyll $a$ (mg/L)</th>
<th>Phytoplankton (mg chlorophyll $a/m^2$)</th>
<th>Notes</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia&lt;sup&gt;1&lt;/sup&gt;</td>
<td>Aquatic Life</td>
<td>0.005-0.015 (inclusive)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Where salmonids are the predominant fish species</td>
<td>BCMWLAP 1998</td>
</tr>
<tr>
<td></td>
<td>Recreation</td>
<td>0.010 (maximum)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Drinking Water</td>
<td>0.010 (maximum)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Alberta</td>
<td>Aquatic Life</td>
<td>0.050</td>
<td>1.0</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>AENV 1999</td>
</tr>
<tr>
<td>Saskatchewan</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>SERM 1997</td>
</tr>
<tr>
<td>Manitoba</td>
<td>All (see note)</td>
<td>0.025 (reservoirs, lakes, or ponds)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>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</td>
<td>Williamson 2002</td>
</tr>
<tr>
<td>Ontario</td>
<td>Aquatic Life</td>
<td>0.020 (applicable to average of ice-free period)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Interim; considered to be “general guidelines”</td>
<td>Ontario MOEE 1994</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.010 (average for the ice-free period where lakes naturally below 0.10 µg/L)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>
Table 15. Continued.

<table>
<thead>
<tr>
<th>Province</th>
<th>Usage</th>
<th>TP (mg/L)</th>
<th>TN (mg/L)</th>
<th>Chlorophyll (a) Phytoplankton (mg chlorophyll (a)/m(^2))</th>
<th>Periphyton (mg chlorophyll (a)/m(^2))</th>
<th>Notes</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quebec</td>
<td>Aquatic Life and Recreation</td>
<td>0.01 maximum; 50% above natural background (ice-free season average)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>For lakes with average TP in ice-free season of (\leq 0.01) mg/L</td>
<td>Ministere de L’Environnement du Quebec 2004</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.02 maximum; 50% above natural background (ice-free season average)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>For lakes with average TP in ice-free season of 0.01-0.02 mg/L</td>
<td></td>
</tr>
<tr>
<td>National (CCME)</td>
<td>Aquatic Life</td>
<td>Not to exceed trigger ranges; and not to increase by more than 50% above “baseline” level</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Trigger Ranges: (&lt; 0.004) mg/L: ultra-oligotrophic; (0.004-0.01) mg/L: oligotrophic (0.01-0.02) mg/L: mesotrophic; (0.02-0.035) mg/L: meso-eutrophic; (0.035-0.100) mg/L: eutrophic; (&gt; 0.100) mg/L: hyper-eutrophic</td>
<td>CCME 2004</td>
</tr>
</tbody>
</table>

\(^1\) TP lakes is either the spring overturn concentration, if the residence time of the epilimnetic water is greater than 6 months, or the mean epilimnetic growing-season concentration, if the residence time of the epilimnetic water is less than 6 months.
4.6.1 CCME Phosphorus Guidance Framework

The CCME (2004) recently issued a phosphorus guidance framework for management of Canadian freshwater ecosystems. The framework incorporates the OECD (1982) trophic categorization scheme for TP with a slight adjustment (Table 16); the OECD (1982) mesotrophic category has been split into two categories due to the large variation in community composition and biomass that reportedly occurs over the range of 10-35 \( \mu \text{g/L} \) in Canadian freshwater systems. Under this framework, ecosystem objectives are identified (enhance, protect, or restore), the reference condition is established (see Section 4.6.2 for a discussion of common approaches) and the current TP concentration is compared to the associated trigger range defined on the basis of reference conditions. Additionally, the framework also includes an additional ‘clause’ indicating that increases in TP greater than 50% above the baseline are construed as “increased risk”.

The supporting scientific document generated for the production of the phosphorus guidance framework (EC 2004), recommends that TP concentrations for lakes should be defined as an average for the open-water season, based on several observations, or as the concentration measured during spring overturn.

Table 16. CCME trophic categories and associated trigger ranges for TP concentrations in Canadian freshwater ecosystems (CCME 2004).

<table>
<thead>
<tr>
<th>Trophic Status</th>
<th>TP (( \mu \text{g/L} ))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ultra-oligotrophic</td>
<td>&lt; 4</td>
</tr>
<tr>
<td>Oligotrophic</td>
<td>4-10</td>
</tr>
<tr>
<td>Mesotrophic</td>
<td>10-20</td>
</tr>
<tr>
<td>Meso-eutrophic</td>
<td>20-35</td>
</tr>
<tr>
<td>Eutrophic</td>
<td>35-100</td>
</tr>
<tr>
<td>Hyper-eutrophic</td>
<td>&gt; 100</td>
</tr>
</tbody>
</table>
4.6.2 Defining Baseline Nutrient Conditions: Reference Conditions

There are a variety of methods that have been employed to derive site-specific nutrient criteria for freshwater ecosystems. These methods generally fall into the following approaches:

- Regional reference sites: ‘pristine’ or reasonably unaltered conditions for a given lake are defined as a statistical condition derived from reference lakes. Reference lakes are often defined on an ecoregional basis. The 'reference condition approach' can be used in which the 'best available' condition is defined on the basis of conditions measured in relatively 'pristine' sites with similar characteristics.

- Use of historical data: criteria for a given lake are based on historical conditions (e.g., pre-development nutrient concentrations). This approach is rarely feasible given the limited historical records for most sites.

- Paleolimnological data: nutrient concentrations can be estimated through examination of physical, chemical, and algal remains (generally diatoms) in sediments. Historical conditions are estimated for various sediment depth strata in a sediment core. ‘Diatom-based inference models’ have been successfully applied for estimating TP concentrations as well as the timing, rate, extent, and probable causes of eutrophication in lakes. This approach has also been used to identify naturally productive lakes in some areas of Canada (see EC 2004 for review).

- Modeling: water quality models may be constructed and calibrated based on existing conditions and then used to estimate historical conditions through manipulation of the model parameters such as nutrient loading. Nutrient export coefficients may also be used to estimate loading associated with various land uses and ‘hindcasting’ of historical watershed nutrient loads can be estimated based on historical records of land use.

- Scientific/ecological studies: criteria may be derived and/or refined based on site-specific studies of nutrient – algal growth relationships.

Where nutrient criteria are developed from reference lakes, use of ecoregions for defining baseline nutrient conditions is recommended because it accounts for natural variations associated with differences in topography, climate, chemistry, geology etc. Therefore, defining a natural state for a given ecoregion assists in differentiating between ‘natural’ and anthropogenic nutrient enrichment (EC 2004). This is particularly important in the prairie ecozone where nutrients are typically naturally higher than other areas of Canada (Chambers et al. 2001). The CCME (2004) recommends defining reference conditions as the “best available condition” using relatively clean sites that are biologically, chemically, and physically similar.
Historical data for a lake are often limited in quantity and typically do not extend to pre-development periods. Additionally, the quality of historical data may be limited for a variety of reasons including exclusion of certain parameters (e.g., TKN was frequently not measured in some historical monitoring programs), changes in analytical methods, lack of QA/QC programs historically, and lack of reporting on sampling and analysis methods. Additionally, data may not have been collected at appropriate sampling locations within a lake and/or in sufficient frequency to capture seasonal variability. For these reasons, nutrient criteria are rarely derived from historical conditions in a particular waterbody. However, management targets are often established based on conditions measured at a particular period of time. Nonetheless, consideration of historical conditions on a specific waterbody is important when developing nutrient criteria. For example, it has been shown that cyanobacteria blooms have occurred for at least hundreds of years in some lakes (Andersen 1995 in Van der Molen and Boers 1999).

Paleolimnological approaches have been very effectively and widely used to estimate past nutrient conditions, and even for the identification of nutrient sources. The approach utilizes the micro and macro-fossil remains in lake sediments as indicators of nutrient conditions (e.g., prior to the onset of eutrophication). In the most general sense, nutrient conditions can be estimated by either reconstructing the biological communities in the past, a method which is best suited to shallow lakes, or the transfer function approach. The latter has been generally accomplished using diatom-TP relationships (reviewed in Batterbee et al. 2005). The use of polyphosphate as an indicator of nutrient enrichment has been recently discussed by Kenney et al. (2001), in which it was suggested that P enrichment can be linked to increasing polyphosphates.

Despite widespread use and acceptance, the diatom-TP model approach is not appropriate for all situations, namely at shallow sites, in nitrogen-limited systems, or systems experiencing additional, concurrent perturbations (Battarbee et al. 2005). Additionally, studies comparing paleolimnological estimates of nutrient concentrations to actual monitoring data have reported significant disparities in some systems. For example, Dokulil and Teubner (2005) found that diatom-inferred TP concentrations from sediment cores underestimated actual TP concentrations in Lake Mondsee (Austria) during the phase of highest TP concentrations. It is speculated that this underestimation was a result of the high abundance of *Planktothrix rubescens*, a cyanobacteria characteristic of meso- to eutrophic lakes that produces microcystin, at that time. Additionally, changes in diatom taxa in sediments over long time frames may reflect effects of non-nutrient related stressors and/or interactions of multiple factors. For example, Keatley et al. (2006) reported that shifts in diatom taxa in a small pond on Melville Island (NWT) likely reflected a shift in climate.
As N has generally received less attention in eutrophication studies, the majority of paleolimnological work has focused on P. However, Christie and Smol (1993) produced TN transfer functions for diatoms, based on data collected in southern Ontario lakes. They found that TN was the strongest predictor of diatom taxa distribution. Additionally, Schelske et al. (2006) recently reported that sedimentary biogenic silica served as a sensitive proxy for low-level phosphorus enrichment in the Great Lakes.

There are numerous water quality models that have been used to either predict future nutrient concentrations or for hindcasting past nutrient conditions in lakes. Models range in complexity from the relatively simple BATHTUB model to the more complex WASP and AQUATOX models. Empirical models have also been developed for a variety of lakes (e.g., Jensen et al. 2006). Modeling is a very useful approach for estimating historical trophic conditions in a waterbody. BATHTUB is particularly applicable for situations where land use has changed significantly and there is a need to simulate the effects of past land use patterns on nutrients in a lake or reservoir. The use of the nutrient export coefficient model approach is useful for evaluating past and potential future land use changes but is limited in its inability to account for the occurrence of internal loading, most notably for shallow lakes (Bennion et al. 2005).

4.6.3 USEPA Nutrient Guidance Framework

As a component of the national strategy for the development of nutrient criteria, the USEPA has issued a guidance document for the derivation of site-specific nutrient criteria for lakes and reservoirs (USEPA 2000a). The document is intended to provide guidance and a review of the science pertaining to eutrophication in lakes and reservoirs to assist state agencies in developing nutrient standards. Typically, the USEPA produces numerical criteria which are then adopted by state agencies as water quality standards across the nation (i.e., a single criterion is developed to be applicable to all waterbodies across the country). However, because there is a great deal of variability in nutrient concentrations and responses of aquatic ecosystems to nutrient enrichment, a single numerical criterion could not be generated for the entire nation. Rather, the guidance document describes methods and approaches through which site-specific nutrient criteria can be developed and provides suggested nutrient criteria as a 'starting point'.

The USEPA developed 'nutrient criteria guidance' for nitrogen, phosphorus, (i.e., the "causal indicators") and 'other nutrient parameters' including chlorophyll a and Secchi depth (i.e., the "response indicators") under Section 304(a) of the Clean Water Act (Table 17). The criteria were derived for given waterbody types (i.e., streams, lakes/reservoirs, and estuaries) and aggregate ecoregion. The intent of these criteria is to provide states with a starting point for development of state or region-specific nutrient standards defined to support designated uses of waters. States would examine the candidate criteria and further sub-divide lake for derivation of more site-
specific nutrient criteria. Sub-divisions could be done based on more finer divisions of ecoregions, watershed size, or on the basis of lake morphological characteristics such as mean depth.

USEPA criteria for lakes in 12 of the 14 aggregate ecoregions are presented in Table 17. These criteria are intended as a starting point for the development of state standards and were derived from actual conditions measured in lakes within each of the aggregate ecoregions. As is immediately evident, reference lake conditions vary considerably across the ecoregions. TP concentrations range from 0.008 to 0.0375 mg/L, TN ranges from 0.12 to 1.27 mg/L, chlorophyll a ranges from 1.08 to 12.35 µg/L and Secchi depth ranges from 0.79 m to 4.93 m. With the exception of TP (which is highest for ecoregion VI), the most extreme parameter values occur in aggregate ecoregion XIII. The latter ecoregion is the southern Florida coastal plain and is tropical.
Table 17. USEPA nutrient criteria based on aggregate ecoregions: lakes and reservoirs. NA = criteria not available (USEPA 2002).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>I</th>
<th>II</th>
<th>III</th>
<th>IV</th>
<th>V</th>
<th>VI</th>
<th>VII</th>
<th>VIII</th>
<th>IX</th>
<th>X</th>
<th>XI</th>
<th>XII</th>
<th>XIII</th>
<th>XIV</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>(mg/L)</td>
<td>NA</td>
<td>0.00875</td>
<td>0.017</td>
<td>0.020</td>
<td>0.033</td>
<td>0.0375</td>
<td>0.008</td>
<td>0.008</td>
<td>0.020</td>
<td>NA</td>
<td>0.008</td>
<td>0.010</td>
<td>0.0175</td>
<td>0.008</td>
</tr>
<tr>
<td>TN</td>
<td>(mg/L)</td>
<td>NA</td>
<td>0.12</td>
<td>0.40</td>
<td>0.44</td>
<td>0.56</td>
<td>0.78</td>
<td>0.24</td>
<td>0.24</td>
<td>0.36</td>
<td>NA</td>
<td>0.46</td>
<td>0.52</td>
<td>1.27</td>
<td>0.32</td>
</tr>
<tr>
<td>Chl a</td>
<td>(µg/L)</td>
<td>NA</td>
<td>1.08</td>
<td>3.40</td>
<td>2.00</td>
<td>2.30</td>
<td>8.59</td>
<td>2.43</td>
<td>2.43</td>
<td>4.93</td>
<td>NA</td>
<td>2.79</td>
<td>2.60</td>
<td>12.35</td>
<td>2.90</td>
</tr>
<tr>
<td>Secchi Depth</td>
<td>(FTU/NTU)</td>
<td>NA</td>
<td>1.30</td>
<td>2.70</td>
<td>2.00</td>
<td>1.30</td>
<td>1.36</td>
<td>4.93</td>
<td>4.93</td>
<td>1.53</td>
<td>NA</td>
<td>2.86</td>
<td>2.10</td>
<td>0.79</td>
<td>4.50</td>
</tr>
</tbody>
</table>

I = Willamette and central valleys;  
II = western forested mountains;  
III = xeric west;  
IV = great plains grass and shrubland;  
V = south central cultivated great plains;  
VI = corn belt and northern great plains;  
VII = mostly glaciated dairy region;  
VIII = nutrient-poor largely glaciated upper Midwest and northeast;  
IX = southeaster temperate forested plains and hills;  
X = Texas-Louisiana coastal and Mississippi alluvial plains;  
XI = central and eastern forest uplands;  
XII = southern coastal plain;  
XIII = southern Florida coastal plain; and  
XIV = eastern coastal plain.
Aggregate ecoregion VI is characterized by the highest TP criteria, the second highest TN and chlorophyll $a$, and the third lowest Secchi depth. Ecoregion VI, the corn belt and northern great plains, includes parts of North and South Dakota, Minnesota, Iowa, Illinois, Michigan, Ohio, Nebraska, Indiana, and a small portion of Wisconsin, and is characterized by nutrient-rich soils (USEPA 2000b). Reference conditions for the two ecoregions bordering the province of Manitoba are presented in Table 18. Reference TN and TP concentrations for ecoregion 46, the northern glaciated plains, are considerably higher than for the associated aggregate ecoregion VI or any of the other aggregate ecoregions (USEPA 2000b).

Table 18. Reference conditions for aggregate ecoregion VI Level III ecoregions 46 and 48 lakes (USEPA 2000b).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Ecoregion 46</th>
<th>Ecoregion 48</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>(mg/L)</td>
<td>0.090</td>
<td>0.05125</td>
</tr>
<tr>
<td>TN</td>
<td>(mg/L)</td>
<td>1.43</td>
<td>0.63</td>
</tr>
<tr>
<td>Chlorophyll $a$</td>
<td>(µg/L)</td>
<td>6.5</td>
<td>4</td>
</tr>
<tr>
<td>Secchi Depth</td>
<td>(m)</td>
<td>1.46</td>
<td>1.98</td>
</tr>
</tbody>
</table>

The technical guidance manual for lakes and reservoirs describes a process for developing nutrient criteria that entails consideration of five factors (USEPA 2000a). These factors or steps are:

- Establish a Regional Technical Assistance Group (RTAG);
- Assemble and examine historical information for a given waterbody or region, generally based on information collected over the last 25 years;
- Establish a reference condition using one or more of several approaches;
- Develop and use theoretical or empirical models of the historical and reference condition data to assist in understanding the condition of the lake; and
- Consider downstream effects.8

RTAGs are intended to evaluate information and data describing nutrient conditions and identify nutrient criteria. An RTAG would consist of regional specialists in such areas as biology, limnology, natural resources management, chemistry, and ecology.

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8 “Downstream receiving waters are considered to be those immediately below the lake or reservoir and within a few miles of it.”
Examination of historical information is used to assist in identifying past conditions as well as
temporal changes in nutrient or trophic condition and other important factors relevant to
eutrophication in the lake or reservoir.

As indicated in Section 4.6.2, there are a variety of methods that can be used to define a
reference condition for a waterbody. The two primary methods discussed by the USEPA are
based on the derivation of conditions for lakes/reservoirs within an eco-region or sub-region,
using measurements of nutrients and related variables in lakes and reservoirs in that region. The
first method involves deriving a numerical criteria from conditions in relatively pristine lakes,
where these lakes are considered to represent 'natural' conditions in the absence of anthropogenic
nutrient enrichment. In this instance, the 75th percentiles and nutrient or related variables for the
reference lakes form the criteria.

The other method of defining reference conditions is the use of a 25th percentile measure for a
representative sample of lakes in a given region. In this case, the lakes may be affected by
anthropogenic nutrient enrichment and the use of a lower percentile considers this factor. This
method also inherently allows for some nutrient enrichment (i.e., up to the 25th percentile) but
not enough to cause adverse effects to the lake or downstream environment. The former
approach (the 75th percentile) is the preferred method but for regions where 'pristine' lakes are
rare, the latter method can be employed. In either case, the reference condition, or best
attainable, most natural condition of the resource base is estimated from lake data. Typically, in
practice, data are 'reduced' through derivation of seasonal medians for a given lake, calculation
of the overall 25th (or 75th) percentile for a given parameter and season for all lakes combined,
and finally calculation of a 25th (or 75th) percentile for the entire year or period (e.g., open-water
season).

The USEPA (2000a) acknowledges that although nutrient criteria may differ between
waterbodies and waterbody types, criteria should be developed in consideration of effects to
other interconnecting systems. For example, it is critical to consider effects to estuaries when
developing criteria for coastal streams. This consideration is explicitly incorporated into the
framework for nutrient criteria development (i.e., the final step is to consider downstream
effects). However, the technical guidance manual for lakes and reservoirs defines downstream
environment as the “receiving waters…immediately below the lake or reservoir and within a few
miles of it”.

The USEPA also recommends establishing criteria for reservoirs independently from lakes. This
is suggested because of differing physical conditions that occur in lakes and reservoirs, including
the high turbidity of reservoirs which typically modifies nutrient-algal relationships (USEPA 2000a).

### 4.6.4 Nutrient Criteria in Other Jurisdictions

The European Union adopted the European Water Framework Directive (WFD) in 2000, the goal of which is to protect and improve the quality of all surface water resources. The WFD requires that all European waterbodies are classified into one of five ecological classes determined primarily on the basis of biological indicators. It further specifies that all waterbodies attain a status of "good ecological quality" or better by 2015. Classification schemes and, hence, assignation of the 'status' of ecological quality are not, however, defined in the WFD. Sondergaard et al. (2005a) recently proposed a preliminary lake classification scheme for Danish lakes based on the relationships between TP and responses to eutrophication. Their preliminary classification scheme, based on TP, is presented in Table 19. In general, the authors related eutrophication response variables to TP concentrations to form the basis of these categories.

**Table 19. Proposed preliminary classification scheme for deep and shallow Danish lakes based on TP (Sondergaard et al. 2005a).**

<table>
<thead>
<tr>
<th>Ecological Quality</th>
<th>TP (µg/L)</th>
<th>Shallow Lakes</th>
<th>Deep Lakes</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>0-25</td>
<td>0-12.5</td>
<td></td>
</tr>
<tr>
<td>Good</td>
<td>25-50</td>
<td>12.5-25</td>
<td></td>
</tr>
<tr>
<td>Moderate</td>
<td>50-100</td>
<td>25-50</td>
<td></td>
</tr>
<tr>
<td>Bad</td>
<td>100-200</td>
<td>50-100</td>
<td></td>
</tr>
<tr>
<td>Poor</td>
<td>&gt; 200</td>
<td>&gt; 100</td>
<td></td>
</tr>
</tbody>
</table>

The Swedish National Water Quality Criteria (cited In Wilander and Persson 2001) specify nutrient criteria both as a departure from "reference conditions" as well as relative to a scale of phosphorus concentration. Site-specific criteria are then derived on this basis, which include nitrogen criteria, particularly to protect downstream marine environments. Examples of site-specific criteria for four Swedish lakes are presented in Table 20. The lack of targeted nitrogen reduction programs has resulted in nitrogen excess in three of the four largest Swedish lakes. Nitrogen criteria have been proposed to reduce lake concentrations for the purposes of protecting the downstream marine environment.
Table 20. Water quality targets for the four largest lakes in Sweden (In Wilander and Persson 2001).

<table>
<thead>
<tr>
<th>Lake</th>
<th>Target</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Vanern</td>
<td>&lt; 0.008-0.010 mg/L 50% reduction of anthropogenic inputs (to cause a change from 0.8 to 0.6 mg/L)</td>
</tr>
<tr>
<td>Lake Vattern</td>
<td>&lt; 0.06 mg/L &lt; 0.450 mg/L</td>
</tr>
<tr>
<td>Lake Malaren</td>
<td>&lt; 2 times background concentration &lt; 2 times the outlet background transport</td>
</tr>
<tr>
<td>Lake Hjalmaren</td>
<td>&lt; 0.025 mg/L -</td>
</tr>
</tbody>
</table>

Denmark has adopted nutrient loading reduction targets of 50% and 80% for TN and TP, respectively, for aquatic systems through the 1987 "Action Plan on the Aquatic Environment" (Jeppesen et al. 1999). Requirements for nitrogen and phosphorus removals from sewage works are specified on the basis of plant capacity, which is in turn, defined on the basis of person equivalents.

4.7 Nutrient/Eutrophication Management

There are numerous options to consider for the management of nutrient overenrichment in lakes and reservoirs including (USEPA 1998):

- Use of vegetative buffer zones;
- Altering land use practices in the watershed;
- Restoration of aquatic habitat;
- Biomanipulation (e.g., fish removal or stocking);
- Sediment sealing or dredging;
- Hypolimnetic aeration and precipitation of phosphorus;
- Hypolimnetic nutrient removal;
- Macrophyte harvesting;
- Winter or other episodic drawdowns to augment sediment removal or consolidation;
- Relocation of sewage discharges;
- Restoration and protection of wetlands;
• Point source nutrient reductions or elimination; and

• Storm water management.

The applicability, feasibility, and effectiveness of each of the above options for nutrient management vary from lake to lake. The success of nutrient management strategies has varied considerably and the lessons learned from these initiatives have contributed substantially to the science of eutrophication. In general, it has been shown that effective nutrient management typically requires the use of a variety of approaches and methods (e.g., Annadotter et al. 1999). Most management strategies incorporate some level of external nutrient loading reduction initiatives.

One of the more successful methods that has been applied rather extensively in recent years, in conjunction with reductions in external nutrient loadings, is biomanipulation. Numerous programs have been undertaken to reduce the dominance and biomass of planktivorous fish. This results in enhanced grazing which reduces phytoplankton biomass and in the most successful cases, reverses the effects of eutrophication, and triggers a transition to the clear water states. Biomanipulation (i.e., cyprinid reduction) was successfully used, in conjunction with numerous other nutrient management methods, in eutrophic Lake Finjasjon, Sweden (Annadotter et al. 1999). Effects included an increase in daphnids, macrophytes, phytoplankton diversity and water clarity, elimination of cyanobacteria blooms, and a reduction of phytoplankton biomass, chlorophyll $a$, and phosphorus concentrations. Most interestingly, internal phosphorus loading was virtually eliminated. Although employed less frequently, biomanipulation may also be accomplished through stocking of piscivores such as pike.

Biomanipulation is quite commonly coupled with external load reduction programs to accelerate lake recoveries (Jeppesen et al. 1999). Many examples exist demonstrating the success of this approach in reducing eutrophication response variables (i.e., phytoplankton biomass and turbidity); the removal of either planktivores and benthivores (most commonly), or in some instances the stocking of piscivores, has caused the transition of shallow lakes to the clear water state (Jeppesen et al. 1999). In some instances, biomanipulations have lead to reductions in nutrient concentrations in shallow lakes, which is beneficial not only from a self-sustaining lake-centric perspective, but also concerning effects to downstream marine environments. That is, biomanipulation may lead to increased nutrient retention in lakes which greatly benefits downstream environments (Jeppesen et al. 1999). Similarly, Olin et al. (2006) evaluated the effect of cyprinid reductions in shallow (polymictic) and deep (stratified) Finnish lakes, some of which also contained high non-algal turbidity. Observed effects included an increase in piscivorous fish, reductions in the duration and size of cyanobacterial blooms, and increased P retention in the lake. The authors suggest that the biomanipulation effectively changed the role
of the fish community from “nutrient recycler” to “nutrient storage”. Interestingly, Olin et al. (2006) indicated that should the removal of fish be equivalent to or greater than the equivalent of 100 µg/L P in the lake, the removal of P bound in fish biomass could successfully reduce TP in the lake.

Although quite a successful approach in Europe, biomanipulation appears to require substantive effort to be successful. In a recent review of Danish lake restoration experience, Sondergaard et al. (2000) indicated that a minimum of 80% of the zooplanktivorous fish stock should be removed for biomanipulation to be successful. Similarly, other researchers have reported that biomass reductions must be >75% in the first year to be successful (Olin et al. 2006, Meijer et al. 1999). In Denmark and reportedly elsewhere, stocking must be 0.1 individuals/m²/year to be successful (reviewed In Sondergaard et al. 2000). A threshold of approximately 10 g/m² of fish biomass has been identified for cascades to be successful at the lower trophic levels in eutrophic shallow lakes. Additionally, there is some indication that the success of biomanipulation techniques may vary between deep and shallow lakes. Beklioglu (1999) indicated that biomanipulation is generally more successful in shallow lakes due to the significance of macrophytes in these systems.

The success of biomanipulation, at least in European lakes, in the short-term seems clear. However, what remains to be determined is the long-term effects of biomanipulation and in particular, effects of fish removals. Sondergaard et al. (2000) indicated that less intensive biomanipulation interventions may be negated in the long-term by compensatory growth and reproduction in the remnant fish populations. Bergman et al. (1999) found that zooplankton biomass was unaltered by cyprinid removal in Lake Ringsjon, Sweden, possibly owing to compensatory growth of young of the year fish after the reduction program. Olin et al. (2006) in a study of cyprinid reductions in Finnish lakes indicated that removal of 100 kg fish/ha/year could be compensated for within one or two years in these lakes and they questioned the sustainability of the effects of biomanipulation in these lakes. Beklioglu (1999), in a recent review of lake restoration, indicated examples of shallow lakes that have failed to re-establish macrophyte biomass and where water clarity had begun to decrease after biomanipulations. In a study of Finnish lakes, some of which had high non-algal turbidity, Olin et al. (2006) reported that the effectiveness of biomanipulation could be reduced by high inorganic turbidity, that between lake responses can be highly variable, and that the appropriateness of biomanipulation should be carefully evaluated on a lake-by-lake basis.
4.8 MANAGEMENT: NEED FOR NITROGEN CONTROLS?

The low N:P hypothesis and the N limitation/P excess theory in eutrophic systems have collectively led to, and widely supported, the paradigm of phosphorus control for aquatic ecosystems. Most nutrient loading controls have focussed on phosphorus and in some cases have only incorporated phosphorus reductions (e.g., Jeppesen et al. 1999). While this has led to many success stories (e.g., Lake Erie, Nicholls and Hopkins 1993; Lake Balaton, Herodek 2002), there is some recent evidence to suggest that ecosystem recovery may not be adequate without concomitant consideration of nitrogen reductions. Several limnologists have, in fact, recently advocated consideration of nitrogen controls in lake restoration initiatives (e.g., Gonzalez Sagrario et al. 2005, Moss et al. 2005, Barnese and Schelske 1994, Pers 2005).

Since the development of the phosphorus limitation paradigm decades ago, a number of studies have indicated that nitrogen may be more limiting than phosphorus in some environments. Specifically, nitrogen limitation appears to be more common in prairie ecosystems in Canada (CCME 2004, EC 2004), presumably due to the natural occurrence of phosphorus-rich soils. Similarly, lakes in other regions of the world are believed to be 'naturally' nitrogen limited, for example in: England (Moss et al. 2005); mountains of western North America (Schindler 2006); South American lakes (Soto 2002 In Schindler 2006); softwater, oligotrophic Florida lakes (Barnese and Schelske 1994); New Zealand lakes (e.g., Burns et al. 2005); and Arctic lakes (O'Brien et al. 2005). Consideration of nitrogen controls has been advocated in lake management programs for these systems (e.g., Barnese and Schelske 1994). It is often difficult to determine which of the two nutrients (nitrogen or phosphorus) is limiting or more important to plant and algal growth, largely because both nutrients tend to increase concurrently in lakes. For this reason, the OECD (1982) indicates that "it is impossible to speculate solely on the basis of nutrient conditions found in a lake, which one of the two factors is limiting production". Resolution of this issue, therefore, requires careful analysis on a site-specific basis.

The significance of nitrogen as a limiting plant nutrient is not in itself unusual. The interest and attention paid to phosphorus and its role in phytoplankton growth in lakes owes its roots to the "phosphorus paradigm" that emerged several decades ago (Kalff 2002). However, even in the formative eutrophication studies the importance of nitrogen was acknowledged and it has been demonstrated in fertilization and limnocorrall studies that for the majority of studies, both nitrogen and phosphorus had to be added together to cause a substantive stimulation of algal production (e.g., Elser et al. 1990, Gonzalez Sagrario et al. 2005). It is also important to acknowledge that because the nutritional requirements of phytoplankton vary between species, varying nutrient limitation can co-exist in a community (Hecky and Kilham 1988).
The significance of nitrogen was even noted in studies of Lake 226 – the site of the whole-lake initial nutrient enrichment studies that led to the P limitation paradigm. Findlay and Kasian (1987) reported that phytoplankton biomass increased in both of the basins of Lake 226, where one basin received N, P, and C and the other basin only received N and C, although the increase was greater in the basin receiving P additions. That N and C additions alone stimulated phytoplankton growth was not expected and several possible explanations were provided by the authors. Additionally, phytoplankton biomass was reduced in both basins following the cessation of both N and P additions.

Other lines of evidence that argue for nitrogen controls include the modified low N:P ratio hypothesis. As indicated in Section 4.4.1, cyanobacteria blooms of non-nitrogen fixing species may be supported at high (or low) N:P ratios, and programs aimed at P reductions leading to increases in the N:P ratio may not be adequate to prevent cyanobacterial blooms, including species capable of producing toxins. Furthermore, there is evidence indicating that lowering the N:P ratio or TN does not necessarily promote cyanobacteria (Jeppesen et al. 1998). These postulations have been fairly widely documented and the recommendation to modify the low N:P hypothesis by Levine and Schindler (1999) recognizes this fact. Although Levine and Schindler (1999) suggest that P reductions are the most effective management approach, there is some literature discussing the need for reductions in N. For example, Pers (2005) concluded, through the use of a biogeochemical lake model (BIOLA) of the eutrophic Lake Ostra Ringsjon, Sweden, that reductions of both N and P would have the greatest effect on the lake ecological status, including nutrients, phytoplankton, and cyanobacteria, than controlling either N or P alone. Smith (2001) advised strongly against nitrogen reductions and/or initiatives that would lower the N:P ratio below 10 (by mass) but makes an important distinction that nitrogen reductions could be considered in conjunction with "strong restrictions on phosphorus as well". Similarly, as indicated in Section 4.4.1, Hendzel et al. (1994) reported that the elimination of experimental N loading (but not P loading) to Lake 227 caused a consistent dominance of nitrogen-fixing cyanobacteria. Fixed nitrogen was deposited to and then recycled from the sediments. Findlay et al. (1994) also reported that elimination of experimental N loading, but maintenance of P fertilization, increased the predictability of the occurrence of nitrogen-fixing cyanobacteria dominance in Lake 227 and increased N fixation in the lake. These studies indicate that when P concentrations/supplies are constant, that cyanobacteria can compensate for reduced N loading through N fixation.

Some recent research has indicated that high concentrations of N may adversely affect macrophyte biomass and subsequent recovery to a clear water state in shallow lakes (Gonzalez Sagrario et al. 2005). Moss et al. (2005) suggested that both nitrogen and phosphorus controls are required to reach the desired recovery phase for Rostherne Mere and Little Mere (England).
was suggested that high N loading may have caused the observed low plant diversity and subsequent instability of the macrophyte community. The authors further suggest that the slow recovery of shallow lakes following the wide-spread P reduction programs implemented world-wide may reflect the significance of nitrogen in eutrophication.

The role of nitrogen in the form of nitrate is complex in terms of effects to P cycling in lakes. For example, at high concentrations nitrate may inhibit the release of phosphorus from sediments. However, at sufficiently high nitrate concentrations, P release from sediments can be promoted (Gonzalez Sagrario et al. 2005). Accumulations of nitrate have been observed in Swedish lakes following years of external phosphorus loading reductions (Wilander and Persson 2001).

In a recent review of the responses of lakes to re-oligotrophication Jeppesen et al. (2005b) cautioned that nitrogen controls require careful consideration in nutrient management strategies, at least for shallow lakes. Studies published in recent years have shown that nitrogen has a negative effect on macrophyte species richness (James et al. 2005) and reduces the probability that macrophytes will remain dominant in lakes with moderately high TP (Gonzalez Sagrario et al. 2005). For example, submerged plants tend to disappear in shallow Danish lakes with TP concentrations between 30 and 150 µg/L when the summer TN concentrations are above 1 to 2 mg/L (Gonzalez Sagrario et al. 2005). Nitrogen additions were required in conjunction with phosphorus to negatively affect macrophyte dominance in mesocosm experiments; macrophytes were not affected at TP 0.07-0.13 mg/L and TN ≤ 1.2 mg/L (Gonzalez Sagrario et al. 2005). Jeppesen et al. (2005b), and others (e.g., Gonzalez Sagrario et al. 2005, Sondergaard et al. 2005a), emphasize the importance of nitrogen for macrophytes and the need to consider nitrogen loading reductions in lake management.

Collectively, the literature indicates that the question of whether to remove nitrogen, phosphorus, or both, needs to be considered on a case-by-case basis. There is some indication that high or moderate levels of nitrogen may exert adverse effects in aquatic ecosystems (e.g., effects to macrophytes, effects of nitrate on P mobilization in sediments), that some cyanobacteria may flourish in environments low in P and high in N:P ratios, and that downstream environments may require nitrogen reductions from upstream systems. The precise targets and approaches need to be developed in consideration of the objectives of the program, the spatial areas/waterbodies to be considered in the program, and the precise conditions of the environment of concern.
4.9 OLIGOTROPHICATION

While the process of eutrophication has been widely studied in lakes for decades, the science of 're-oligotrophication' in lakes is comparatively new. However, with the wide-spread advent of numerous nutrient management initiatives across the globe in the 1970s, there have been a number of recent studies published in which lake re-oligotrophication has been examined based on decades of research and observations (largely European studies). The major issue to be resolved with respect to re-oligotrophication is the question of whether lakes will respond in a manner consistent with eutrophication responses, but simply in the reverse direction (i.e., following a similar but opposite trajectory). A number of studies have questioned this paradigm (e.g., Battarbee et al. 2005, Sondergaard et al. 2005b) and have reported some rather surprising lake responses. Some researchers have even postulated that some lakes may not return to pre-enrichment conditions (e.g., Scheffer et al. 1993 In Battarbee et al. 2005, Burgi and Stadelmann 2002, Kohler et al. 2005). In a long-term study of recovery of the four largest lakes in Sweden, Wilander and Persson (2001) reported that "it is obvious that immediate and complete reversibility are rarely seen". The response and predictability of lakes to nutrient control measures is, of course, critical to effective nutrient management. The following is a brief overview of a selection of recent publications on this subject.

Jeppesen et al. (2005b) published a review of re-oligotrophication based on 35 lake case studies. Numerous hypotheses for the recovery phase were examined including:

- Notable delay in reductions in TP;
- Rapid decline in TN;
- Increase in the TN:TP ratio;
- Decrease in the SRP:TP ratio;
- Increase in the chlorophyll a:TP ratio;
- Reduced phytoplankton biomass;
- An inconsistent response in the fish community;
- Reduced zooplankton biomass; and
- Increased Secchi disk depth.
Many of the above hypotheses were supported in the review. However, there were notable exceptions and it was further found that deep and shallow lakes did not always respond consistently. Responses that were observed in both deep and shallow lakes were reductions in TP concentrations, chlorophyll \( a \) concentrations, and phytoplankton biomass and an increase in Secchi depth. The first hypothesis indicated above (i.e., delays in TP reductions due to internal loading) was supported by the case studies which indicated approximately 10-15 years was required before a new equilibrium between sediments and water was reached. Although there was no clear indication that lake depth affected the rate of recovery, lakes with shorter retention times tended to recover more rapidly.

Rapid recovery of in-lake TN (i.e., <5 years) was observed following N loading reductions, as has been observed in other studies (e.g., Kohler et al. 2005, Jeppesen et al. 1998), but not all (Sondergaard et al. 2005b). Jeppesen et al. (2005b) attributed this rapid reduction to the occurrence of denitrification which effectively removes nitrogen from lakes in a relatively short time frame. However, DIN:TN ratios also increased indicating that denitrification was not adequate to fully compensate for reduced use of DIN by phytoplankton as nutrient loading is reduced.

TN:TP ratios generally increased in lakes undergoing recovery, however, the extent to which this is an artifact of the greater emphasis on phosphorus load reductions relative to nitrogen is not known (i.e., this may reflect larger external reductions in P than N). Ratios were also positively correlated to lake depth. Similarly, DIN:SRP ratios increased in the majority of shallow and deep lakes in conjunction with reduced TP loading; in-lake TP concentrations and the ratio was positively related to mean lake depth and negatively correlated to water residence time. In some cases, however, low ratios were observed in lakes with low TP concentrations (i.e., 20 µg/L).

Also as predicted, SRP:TP ratios decreased in most lakes with decreasing TP concentrations and SRP concentrations were positively correlated to lake depth. Exceptions were observed for two lakes, which was suggested to be a result of the relatively greater TN loading reductions relative to TP.

Chlorophyll \( a \):TP ratios generally increased with external loading reductions but decreased at low TP concentrations. This is believed to reflect an increase in the significance of top-down control by grazing at low TP concentrations. Overall, chlorophyll \( a \) declined during the recovery period, even at relatively high TP concentrations in deep and shallow lakes. Chlorophyll \( a \) was also positively and linearly related to the summer TP and TN concentrations, indicating both nutrients were significant factors.
An interesting finding of this study was a relatively rapid and significant response of fish to reduced nutrient loading. There was an overall reduction in fish catches and an increased proportion of potential piscivores, with responses occurring in less than 10 years. This finding has implications for nutrient management options, particularly the usefulness of biomanipulation (i.e., fish removals).

Changes in the phytoplankton community structure varied between lake types (i.e., deep vs. shallow) with diatoms also becoming more abundant in shallow lakes and chrysophytes in deep lakes. In most lakes cyanobacteria also became less dominant with reduced nutrient loading. Cyanobacteria contributed significantly to phytoplankton biomass down to 50 µg/L and 10-15 µg/L in shallow and deep lakes, respectively. Two lakes exhibited atypical responses to reduced nutrient loading with respect to the phytoplankton community structure. Lakes Gundsomagle and Sogard (shallow hypereutrophic lakes) underwent a shift from chlorophytes to cyanobacteria even though the TN:TP ratio of inflowing water and DIN:SRP ratios in the lake increased substantially. The authors suggested this occurrence indicates that N:P ratios may not be of particular importance in determining dominance of cyanobacteria in these systems. This is a significant observation as it is the general paradigm that cyanobacteria dominance is significantly affected by nutrient ratios (i.e., that cyanobacteria hold a competitive edge at low N:P ratios) and nutrient management is most often focused on removal of P in consideration of this phenomenon (Jeppesen et al. 2005b).

Other studies have reported similar results as Jeppesen et al. (2005b). However, some rather surprising observations have also been made. Burgi and Stadelmann (2002) reported a substantive increase in the species richness of phytoplankton in Lake Sempach, Switzerland, following external P reductions and lake aeration. However, phytoplankton biomass actually increased with nutrient reductions and the cyanobacteria Planktothrix dominated in the lake, despite increases in N:P ratios. Dokulil and Teubner (2005) demonstrated that phytoplankton biovolume and species composition shifts lagged behind nutrient reductions in Lake Mondsee (Austria). Species that characterized the community prior to nutrient reductions persisted during the early phases of oligotrophication. This effect has been reported by a number of researchers, most importantly, the persistence or even increase in cyanobacteria after loading reductions (e.g., Jacquet et al. 2005, Burgi and Stadelmann 2002). Jeppesen et al. (2005a) suggested that TP rather than the ratio of TN:TP was more important for determining the shift from non-heterocystous to heterocystous cyanobacterial dominance observed in shallow Danish lakes after nutrient loading reductions. They observed, as have others (reviewed in Jeppesen et al. 2005a,b), that the relative abundance of heterocystous cyanobacteria increased following external loading reductions (and increasing TN:TP).
Soininen et al. (2005) found that the phytoplankton community structure was most unpredictable in the most productive basin of a lake, suggesting that enriched ecosystems may be inherently more variable. This further complicates the ability to predict, and therefore manage, the effects of nutrient reductions on aquatic ecosystems. Similarly, as Battarbee et al. (2005) pointed out, there is considerable variability in the responses of lakes to re-oligotrophication. This is also consistent with the findings of Moss et al. (2004) who demonstrated that response processes in eutrophic shallow lakes are more unpredictable at higher latitudes.

Lags in responses to nutrient reduction programs have also been noted. Jeppesen et al. (2005b) reported that macrophytes appeared to exhibit a delayed response to increased water clarity and there was considerable variability in macrophyte responses among lakes. Kohler et al. (2005) found similar results as Jeppesen et al. (2005b) in a study of Lake Muggelsee, Germany following external nutrient load reductions. As observed by Jeppesen et al. (2005b), the lake rapidly responded to external nitrogen reductions by reductions in TN concentrations. Also consistent with results of Jeppesen et al. (2005b), TP recovery was slower, with maintenance of high TP in summer owing to the occurrence of internal loading. Interestingly, Kohler et al. (2005) speculated that the lake experienced an increase in internal loading due to reductions in nitrate. Additionally, Kohler et al. (2005) and Phillips et al. (2005) emphasized the importance of seasonality on the response of TP in lakes to external load reductions. That is, TP concentrations may decline in winter and spring but remain elevated or even increase in summer during oligotrophication. These changes may be masked when TP is expressed as an annual average.

Finally, external factors other than those discussed in the preceding sections may affect the responses of aquatic ecosystems to nutrient management initiatives. These 'other factors' include contaminants, which may for example exert an adverse effect on grazers, or climate change, which may offset the effects of restoration initiatives (Battarbee et al. 2005).
5.0 RIVERS AND STREAMS

Streams generally respond quite differently to nutrient enrichment than lakes (generally streams are less responsive than lakes) and effects to primary production are heavily influenced by physical factors such as substrate types, water depth, water clarity, shading, water velocities and travel times (EC 2004). Nutrient availability is also typically higher in streams than lakes (Wetzel 1983) and concentrations may fluctuate widely on a seasonal basis as well as in the short-term in association with storm events. Nutrients also behave differently in streams than lakes, as they are typically repeatedly taken up and then released by biota while at the same time being transported downstream. This process, termed "spiralling", describes the cycling of nutrients between the water, sediments, and biota (Chambers et al. 2001; Hauer and Lamberti 1996).

Nutrient enrichment of streams is a widespread, global phenomenon and is arguably one of the most significant issues respecting water quality in lotic systems. In the United States, nutrients have been identified as the leading cause of impairment in lakes and coastal waters and the second leading cause of impairment to rivers and streams (the leading cause is siltation, USEPA 1998). In Manitoba, most streams where monitoring has occurred from the 1970s onward showed either an increase or no significant change in TN or TP concentrations over the last several decades (Jones and Armstrong 2001). Although there is no single source describing nutrient data and related variables for all stream sites monitored in Manitoba, several publications have provided information most notably for southern Manitoba streams (e.g., Armstrong 2002, Hughes 1999, Jones 1999); a summary of data for long-term monitoring sites, provided by Manitoba Water Stewardship, is presented in Table 21.

Despite the widespread occurrence of eutrophication of streams, less is known of the response of streams than lakes to nutrient enrichment as stream eutrophication has received comparatively less attention. The following is a brief overview based on a selection of recent reports describing the current understanding on this issue.

<table>
<thead>
<tr>
<th>Location</th>
<th>Mean TN (mg/L)</th>
<th>Mean TP (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Assiniboine River</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>At Kamsack</td>
<td>1.51</td>
<td>0.14</td>
</tr>
<tr>
<td>At Brandon</td>
<td>1.30</td>
<td>0.17</td>
</tr>
<tr>
<td>At Treesbank</td>
<td>1.53</td>
<td>0.19</td>
</tr>
<tr>
<td>At Portage Spillway</td>
<td>1.56</td>
<td>0.22</td>
</tr>
<tr>
<td>East of Portage</td>
<td>1.70</td>
<td>0.26</td>
</tr>
<tr>
<td>At Headingley</td>
<td>1.70</td>
<td>0.27</td>
</tr>
<tr>
<td><strong>Assiniboine River Tributaries</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Qu’Appelle River</td>
<td>1.25</td>
<td>0.17</td>
</tr>
<tr>
<td>Little Saskatchewan</td>
<td>2.06</td>
<td>0.15</td>
</tr>
<tr>
<td>Souris River, Westhope</td>
<td>1.83</td>
<td>0.25</td>
</tr>
<tr>
<td>Souris River, Treesbank</td>
<td>2.23</td>
<td>0.33</td>
</tr>
<tr>
<td>Cypress River</td>
<td>1.61</td>
<td>0.26</td>
</tr>
<tr>
<td><strong>Red River</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>At Emerson</td>
<td>1.86</td>
<td>0.24</td>
</tr>
<tr>
<td>At St. Norbert</td>
<td>2.02</td>
<td>0.25</td>
</tr>
<tr>
<td>At Selkirk</td>
<td>2.35</td>
<td>0.30</td>
</tr>
<tr>
<td><strong>Red River Tributaries</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LaSalle River</td>
<td>2.00</td>
<td>0.46</td>
</tr>
<tr>
<td>Roseau River</td>
<td>1.18</td>
<td>0.09</td>
</tr>
<tr>
<td>Rat River</td>
<td>1.35</td>
<td>0.15</td>
</tr>
<tr>
<td>Boyne River</td>
<td>1.52</td>
<td>0.15</td>
</tr>
<tr>
<td>Cooks Creek, RM boundary</td>
<td>2.05</td>
<td>0.39</td>
</tr>
<tr>
<td>Pembina River</td>
<td>1.49</td>
<td>0.28</td>
</tr>
<tr>
<td>Seine River</td>
<td>1.53</td>
<td>0.23</td>
</tr>
<tr>
<td><strong>Other Streams</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brokenhead River</td>
<td>1.10</td>
<td>0.06</td>
</tr>
<tr>
<td>Burntwood River</td>
<td>0.41</td>
<td>0.04</td>
</tr>
<tr>
<td>Carrot River</td>
<td>1.01</td>
<td>0.08</td>
</tr>
<tr>
<td>Dauphin River</td>
<td>1.40</td>
<td>0.02</td>
</tr>
<tr>
<td>Mossy River</td>
<td>1.00</td>
<td>0.09</td>
</tr>
<tr>
<td>Nelson River</td>
<td>0.52</td>
<td>0.03</td>
</tr>
<tr>
<td>North Duck River</td>
<td>0.60</td>
<td>0.05</td>
</tr>
<tr>
<td>Ochre River</td>
<td>0.67</td>
<td>0.06</td>
</tr>
<tr>
<td>Red Deer River</td>
<td>1.03</td>
<td>0.04</td>
</tr>
<tr>
<td>Saskatchewan River, above Carrot River</td>
<td>0.59</td>
<td>0.05</td>
</tr>
<tr>
<td>Saskatchewan River, at Grand Rapids</td>
<td>0.42</td>
<td>0.02</td>
</tr>
<tr>
<td>Swan River</td>
<td>1.28</td>
<td>0.18</td>
</tr>
<tr>
<td>Turtle River</td>
<td>0.73</td>
<td>0.06</td>
</tr>
</tbody>
</table>

Table 21. Continued.
<table>
<thead>
<tr>
<th>Location</th>
<th>Mean TN (mg/L)</th>
<th>Mean TP (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Valley River</td>
<td>1.19</td>
<td>0.08</td>
</tr>
<tr>
<td>Vermillion River</td>
<td>1.16</td>
<td>0.21</td>
</tr>
<tr>
<td>Waterhen River</td>
<td>1.02</td>
<td>0.03</td>
</tr>
<tr>
<td>Whitemud River</td>
<td>1.50</td>
<td>0.12</td>
</tr>
<tr>
<td>Wilson River</td>
<td>1.15</td>
<td>0.17</td>
</tr>
<tr>
<td>Winnipeg River</td>
<td>0.51</td>
<td>0.02</td>
</tr>
<tr>
<td>Woody River</td>
<td>1.22</td>
<td>0.13</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.41</td>
<td>0.02</td>
</tr>
<tr>
<td>Maximum</td>
<td>2.35</td>
<td>0.46</td>
</tr>
<tr>
<td>Mean</td>
<td>1.32</td>
<td>0.16</td>
</tr>
<tr>
<td>Median</td>
<td>1.30</td>
<td>0.15</td>
</tr>
</tbody>
</table>
5.1 EFFECTS OF NUTRIENT ENRICHMENT IN STREAMS

Although the sensitivity of streams to nutrient enrichment is typically less than for lakes (EC 2004), the effects of nutrient enrichment in streams also typically includes increased productivity, shifts in community composition, biodiversity and relative dominance of primary producer groups, alterations to water chemistry such as reduced DO, increased pH and turbidity, and in extreme instances, elimination or severe reduction of invertebrate and fish species. The following is a brief synopsis of the key effects of nutrient enrichment in lotic systems.

5.1.1 Phytoplankton, Periphyton, and Aquatic Plants

Nutrient enrichment can increase phytoplankton, periphyton (i.e., benthic algae), and/or plant biomass in streams, although the relative importance of each component varies between sites. The responses of streams dominated by periphyton or phytoplankton to nutrient enrichment may vary for several reasons. Periphyton growth is saturated at much lower levels of nutrients than phytoplankton (USEPA 1998, Dodds and Welch 2000, reviewed in Chambers et al. 2001) and phytoplankton may develop blooms of nitrogen-fixing cyanobacteria. Therefore, streams should be separated into periphyton-dominated and phytoplankton-dominated systems when evaluating eutrophication and developing nutrient criteria. Physical and biological factors affecting the relative abundance of phytoplankton and periphyton in streams are summarized in Table 22 and discussed in Section 5.2. In general, the primary response variable in fast-flowing, grave/cobble bed streams is periphyton biomass, whereas in slow-moving, depositional rivers, phytoplankton biomass is dominant. The following is a brief discussion of the effects of nutrient enrichment on periphyton, phytoplankton, and aquatic plants.
Table 22. Physical and biological factors that affect periphyton and phytoplankton biomass levels in streams, given adequate to high nutrient supply and non-toxic conditions (USEPA 2000c).

<table>
<thead>
<tr>
<th>Phytoplankton-Dominated Systems</th>
<th>Periphyton-Dominated Systems</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>High Biomass</strong></td>
<td><strong>High Biomass</strong></td>
</tr>
<tr>
<td>Low current velocity (&lt;0.1 m/s)/long detention time (&gt;10 days); and</td>
<td>High current velocity (&gt;0.1 m/s); and</td>
</tr>
<tr>
<td>Low turbidity/colour; and</td>
<td>Low turbidity/colour; and</td>
</tr>
<tr>
<td>Open canopy; and</td>
<td>Open canopy; and</td>
</tr>
<tr>
<td>Greater stream depth; and</td>
<td>Shallow stream depth; and</td>
</tr>
<tr>
<td>Greater depth to width ratio.</td>
<td>Minimal scouring; and</td>
</tr>
<tr>
<td></td>
<td>Limited macroinvertebrate grazing; and</td>
</tr>
<tr>
<td></td>
<td>Gravel or larger substrata; and</td>
</tr>
<tr>
<td></td>
<td>Smaller depth to width ratio.</td>
</tr>
<tr>
<td><strong>Low Biomass</strong></td>
<td><strong>Low Biomass</strong></td>
</tr>
<tr>
<td>High current velocity (&gt;0.1 m/s)/short detention time (&lt;10 days); and/or</td>
<td>Low current velocity (&lt; 0.1m/s); and/or</td>
</tr>
<tr>
<td>High turbidity/colour; and/or</td>
<td>High turbidity; and</td>
</tr>
<tr>
<td>Closed canopy; and/or</td>
<td>Closed canopy; and</td>
</tr>
<tr>
<td>Shallow stream depth.</td>
<td>Greater stream depth; and/or</td>
</tr>
<tr>
<td></td>
<td>High scouring; and</td>
</tr>
<tr>
<td></td>
<td>High macroinvertebrate grazing; and/or</td>
</tr>
<tr>
<td></td>
<td>Sand or smaller substrata.</td>
</tr>
</tbody>
</table>

Increased biomass of periphyton is a commonly reported response to nutrient enrichment in streams, including those of the prairie region of Canada (e.g., Chambers et al. 2000b). Effects are frequently observed downstream of point sources, such as municipal and industrial wastewater treatment facilities (WWTFs). For example, an eightfold increase in periphyton biomass was observed in the Bow River downstream of the City of Calgary (reviewed in Chambers et al. 1997). Chambers (1996), as a component of the Northern River Basins Study, reported that periphyton biomass was elevated in the Athabasca and Wapiti rivers, AB, due to nutrient enrichment. Similarly, pulp mill effluent discharge caused a "massive accumulation of algae" in the Thompson River, BC (Bothwell 1992). Although a common response, the precise response of periphyton biomass to nutrient enrichment varies considerably between systems and according to other factors that may limit or modify periphyton growth and loss (see Section 5.2 for further discussion).
Similarly, phytoplankton biomass can be significantly increased as a result of nutrient enrichment in streams and in extreme cases, can manifest as blooms of toxic cyanobacteria. For example, a toxic cyanobacterial bloom developed due to nutrient enrichment in Australia (Bowling and Baker 1996 In Dodds and Welch 2000). High concentrations of phytoplankton chlorophyll $a$ (10-50 $\mu$g/L) also occur in the eutrophic Yamaska River, Quebec (reviewed In Chambers et al. 2001).

Nutrient enrichment has a similar effect on the species composition of algal communities in streams as it does in lacustrine systems. With increasing nutrients, there is shift in the dominant taxa towards green and blue-green algae and the period of dominance by these taxa can be lengthened (Welch 1992 In USEPA 2000c). State agencies in the US that monitor periphyton consistently measure species composition (USEPA 2000c) because periphyton biomass can be highly variable (spatially and temporally), whereas shifts in species composition can be a more reliable tool for monitoring eutrophication effects. Vis et al. (1998) reached a similar conclusion for the St. Lawrence River periphyton community, where the taxonomic composition responded more strongly to municipal wastewaters than did biomass. Specifically, Vis et al. (1998) suggested that the abundance of one species, *Plectonema notatum*, in summer is a useful indicator of wastewater effects. The USEPA (2000c) indicated there is some evidence that the algae species composition of streams may shift to grazer-resistant filamentous algae, which is analogous to increased dominance of grazer-resistant cyanobacteria in lakes. Biggs (2000a) indicated that diatoms are the 'best' food source for most invertebrates and that there is evidence that some invertebrates avoid large, green filamentous algae, possibly due to the large size of the filaments.

The presence and abundance of certain algal taxa can be indicative of nutrient enrichment and trophic status in streams. For example, *Cladophora* and *Melosira* prefer nutrient-rich conditions (Chetalat et al. 1999 In Dodds and Welch 2000) and *Cladophora* is generally associated with eutrophication (Dodds 2006). Vis et al. (1998) reported that the relative abundance of cyanobacteria increased in periphyton communities in the St. Lawrence River, QB, in relation to discharge of municipal wastewaters. Biggs (2000a) summarized the benthic taxa characteristic of various trophic states of New Zealand streams; overall, he indicated that oligotrophic streams are characterized by diatoms, and mesotrophic systems are more dominated by moderately tall growing filamentous green algae, stalked diatoms, and cyanobacteria. In eutrophic New Zealand streams, periphyton is dominated by tall-growing filamentous green algae, such as *Cladophora* and/or the filamentous diatom, *Melosira varians*. However, Chetalat et al. (2006) suggested that increased cyanobacterial biomass occurs at much higher nutrient concentrations in streams than in lakes. Despite these generalizations, there is insufficient information at this time to enable us
to predict at what point nutrient enrichment causes a shift to the 'nuisance' forms of algae in streams (Dodds and Welch 2000, Dodds 2006, Biggs 2000a).

Nutrient enrichment may also increase the biomass of rooted aquatic plants, particularly in Canada (Chambers et al. 1999). For example, macrophyte biomass was found to greatly increase in the Saskatchewan and Bow rivers, downstream of the cities of Saskatoon and Calgary, respectively (Sosiak 2002, Chambers and Prepas 1994). These increases coincided with increased TP concentrations in the rivers (reviewed in Chambers et al. 1997, Sosiak 2002, Chambers and Prepas 1994). Additionally, increased biomass of aquatic plants occur in the backwater zones and reservoirs along the Yamaska River, Quebec, which contains high concentrations of TN and TP (reviewed in Chambers et al. 2001). As Chambers et al. (1999) point out, macrophytes often pose a problem associated with excess biomass in Canada which is unlike the European situation where the issue is generally one of macrophytes loss. However, because aquatic plants obtain most of their nutrients from sediments, macrophyte growth may be uncoupled from TN and TP concentrations in water. Additionally, the effects of eutrophication on macrophytes in streams has been poorly studied. Consequently, it is very difficult to predict the effects of nutrient reductions on macrophyte biomass at this time (Chambers et al. 1999). However, Carvalho et al. (2002) have defined links between macrophytes and nutrients for British rivers as follows:

- Nutrient-poor streams: relatively sparse biomass but a diverse range of higher plants, bryophytes or algae;
- Medium nutrient streams: elevated biomass, but fewer species, of higher plants;
- Nutrient-rich streams: dominated by few species of higher plants or macroalgae; and
- Streams with very high levels of nutrients: dominated by colonies of a few species of filamentous algae or microbial films and most higher plants are lost.

Long-term effects of nutrient enrichment on primary production are rare and indicate that short- and long-term effects may differ significantly. A 16-year study of nutrient enrichment of the Alaskan Kuparik River is one such example. Initially, nutrient enrichment stimulated algal growth but biomass was later (after 2 years) controlled by grazing. After 8 years of enrichment, bryophyte cover increased markedly, thereby reducing epilithic algal biomass (reviewed in Greenwood and Rosemond 2005).
5.1.2 Other Biota

According to the USEPA (2000c) nutrient enrichment is expected to cause an increase in the production and biomass of consumers up to a theoretical maximum nutrient concentration beyond which increases will either stop, or be greatly reduced, relative to incremental increases in nutrients. Eutrophication has been associated with both increased production of benthic invertebrates and fish, as well as shifts in species composition and dominance. However, this area has been poorly studied in comparison to lakes and other effects of nutrient enrichment in streams.

Debruyn et al. (2003) demonstrated that long-term nutrient enrichment in the St. Lawrence River, downstream of Montreal, increased macroinvertebrate and fish production. Similarly growth rates of the most abundant large insects, arctic grayling, and periphyton increased in the Kuparuk River, Alaska, due to P additions (Peterson et al. 1993 In Chambers et al. 2001). Chambers (1996) also reported increased length and weight of spoonhead sculpin and increased periphyton biomass and benthic invertebrate densities on the Athabasca River, AB downstream of Hinton. Longnose sucker exhibited increased energy storage on the Wapiti River, AB, downstream from a pulp mill (reviewed In Chambers et al. 2000b). Perrin and Richardson (1997) observed increased benthic invertebrate densities following nutrient enrichment in stream mesocosms in the Nechako River, BC. Benstead et al. (2005) reported a rapid, four- to seven-fold increase in the abundance and a two- to four-fold increase in the biomass of macroinvertebrates in an oligotrophic Arctic stream following nutrient enrichment. Although only a qualitative assessment, there was some indication that growth rate of Arctic grayling also increased following enrichment.

Concentrations of nutrients that are associated with adverse effects to fish or macroinvertebrates have only been identified in a few instances (reviewed In Biggs 2000a, Table 23) and information on thresholds for streams is limited. In two recent reviews, Dodds and Welch (2000) and Dodds (2006) indicated that nutrient enrichment causes an increase in invertebrate biomass and shifts in community structure. Miltner and Rankin (1998 In Dodds and Welch 2000) were able to link changes in community structure to concentrations of phosphorus. In a recent review, Biggs (2000a) indicated that the contribution of Ephemeroptera/Plecoptera/Trichoptera (EPT) taxa to invertebrate communities has been shown to decline sharply with increasing periphyton biomass, and there is a general shift away from "clean water" invertebrates with increased attached algae in streams.
### Table 23. Some examples of thresholds for biological responses to nutrients and other factors in streams.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Units</th>
<th>Response</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ambient TN:TP</td>
<td>&gt; 20</td>
<td>-</td>
<td>P limiting to benthic algae</td>
<td>USEPA 2000c</td>
</tr>
<tr>
<td></td>
<td>&lt; 10</td>
<td></td>
<td>N limiting to benthic algae</td>
<td>USEPA 2000c</td>
</tr>
<tr>
<td>TN and TP</td>
<td>&lt; 3.0 and &lt; 0.415 (mg/L)</td>
<td>Maximum benthic algae &lt; 200 mg/m²</td>
<td>Calculated from Dodds et al. 1997 In Dodds and Welch 2000</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.35 and &lt; 0.030 (mg/L)</td>
<td>Acceptable (mean &lt; 100 mg/m² and maximum &lt; 150 mg/m²) benthic biomass</td>
<td>Dodds et al. 1997</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.47 and &lt; 0.065 (mg/L)</td>
<td>Mean benthic algae &lt; 50 mg/m²</td>
<td>Dodds et al. 1997</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.25 and &lt; 0.021 (mg/L)</td>
<td>Mean benthic algae &lt; 50 mg/m²</td>
<td>Lohman et al. 1992</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.29 and &lt; 0.042 (mg/L)</td>
<td>Stream phytoplankton chlorophyll (a &lt; 8 \mu g/L)</td>
<td>Calculated In Dodds and Welch 2000 from other studies.</td>
<td></td>
</tr>
<tr>
<td>SRP</td>
<td>&lt; 0.047 (mg/L)</td>
<td>Prevent nuisance algal growth and preserve water quality suitable for salmonid fish in Irish rivers</td>
<td>McGarrigle 1993 In USEPA 2000c</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.01 (mg/L)</td>
<td>Uptake by periphyton saturated</td>
<td>Bothwell 1985, 1989 and Walton et al. 1995</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.01-0.03 (mg/L)</td>
<td>Maximum biomass of <em>Cladophora</em></td>
<td>Freeman 1986 and Watson 1989 In Welch et al. 1992</td>
<td></td>
</tr>
<tr>
<td>TDP</td>
<td>&lt; 0.010 (summer average) (mg/L)</td>
<td>Maximum periphyton biomass remained below 100 mg/m² in the Bow River, AB</td>
<td>USEPA 2000c</td>
<td></td>
</tr>
<tr>
<td>TP</td>
<td>0.022 (mg/L)</td>
<td>Flow-weighted median concentrations of 85 sites across the USA in relatively undeveloped basins</td>
<td>Clark et al. 2000</td>
<td></td>
</tr>
<tr>
<td>TIN and SRP</td>
<td>&gt; 0.61 and 0.060 (mg/L)</td>
<td>Deleterious effects on fish communities in low order Ohio streams</td>
<td>Miltner and Rankin 1998 In USEPA 2000c</td>
<td></td>
</tr>
<tr>
<td>IN and TP</td>
<td>&gt; 1.37 and &gt;0.17 (mg/L)</td>
<td>Significant effects on biotic integrity index for fish and invertebrates (Headwater streams, Ohio)</td>
<td>Miltner and Rankin 1998 In Dodds and Welch 2000</td>
<td></td>
</tr>
</tbody>
</table>
Table 23. Continued.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Threshold</th>
<th>Units</th>
<th>Response</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>TDN and TP</td>
<td>0.553 and 0.118 (mg/L)</td>
<td>High macrophyte biomass (200 g/m$^2$) in the Saskatchewan River, SK</td>
<td>Chambers and Prepas 1994</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.277 and 0.06 (mg/L)</td>
<td>Moderate macrophyte biomass (135 g/m$^2$) in the Saskatchewan River, SK</td>
<td>Chambers and Prepas 1994</td>
<td></td>
</tr>
<tr>
<td>DIN</td>
<td>0.08 (mg/L)</td>
<td><em>Cladophora</em> not growth limited</td>
<td>Freeman 1986 In Welch et al. 1992</td>
<td></td>
</tr>
<tr>
<td>TN</td>
<td>0.26 (mg/L)</td>
<td>Flow-weighted median concentrations of 85 sites across the USA in relatively undeveloped basins</td>
<td>Clark et al. 2000</td>
<td></td>
</tr>
<tr>
<td>Grazer densities</td>
<td>&lt; 3000 individuals/m$^2$</td>
<td>Proliferation of periphyton in New Zealand streams</td>
<td>Welch et al. 1992</td>
<td></td>
</tr>
<tr>
<td>Velocity</td>
<td>&gt; 0.3 (m/s)</td>
<td>Filamentous algae &lt; 100 mg/m$^2$</td>
<td>Biggs et al. 1998a In Biggs 2000a</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 0.1 (m/s)</td>
<td>Conducive to plant establishment</td>
<td>Chambers et al. 1991 In Chambers and Prepas 1994</td>
<td></td>
</tr>
<tr>
<td>Benthic chlorophyll $a$</td>
<td>&lt; 100 (mg/m$^2$)</td>
<td>Areal coverage of filamentous algae less than 20%</td>
<td>Welch et al. 1988</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 100 (mg/m$^2$)</td>
<td>Filamentous algae tend to dominate</td>
<td>Welch et al. 1988</td>
<td></td>
</tr>
<tr>
<td></td>
<td>5-26 (mg/m$^2$)</td>
<td>Reference conditions (median) for the Athabasca and Wapiti rivers, AB</td>
<td>Chambers and Guy 2004</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 100 (mg/m$^2$)</td>
<td>Salmon spawning impaired</td>
<td>Nordin 1985 In Chambers and Guy 2004</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&gt; 100 (mg/m$^2$)</td>
<td>&quot;excessive&quot; biomass</td>
<td>Welch et al. 1988</td>
<td></td>
</tr>
<tr>
<td>Macrophytes</td>
<td>100-500 (g/m$^2$)</td>
<td>&quot;tolerable upper limit&quot; of macrophyte biomass</td>
<td>Summarized In Chambers et al. 1999</td>
<td></td>
</tr>
<tr>
<td></td>
<td>&lt; 1-50 (% cover)</td>
<td>&quot;tolerable upper limit&quot; of macrophyte biomass</td>
<td>Summarized In Chambers et al. 1999</td>
<td></td>
</tr>
</tbody>
</table>
High nutrient levels have also been associated with detrimental effects to fish (reviewed in Biggs 2000a). For example, Sabater et al. (2000) reported that high periphyton biomass in the Oria River, Spain was related to a fish kill and fish diversity; wide diel DO fluctuations were believed to have caused a shift to more tolerant fish species (and likely caused the fish kill) at one location. Rankin et al. (1999) reported a negative correlation between phosphorus concentrations and indices of biological integrity for fish and invertebrates in Ohio streams.

The combined effect of nutrient enrichment on algae and higher trophic levels in streams is variable. In some cases, low-level enrichment of streams has been shown to increase benthic invertebrates, fish, and algal biomass (Perrin et al. 1987, Slaney and Ward 1993 both in USEPA 2000c). In others, enrichment has caused increased grazer biomass but has had no effect on periphyton biomass (Biggs and Lowe 1994 in USEPA 2000c). Increases in secondary production appear to occur regardless of whether enrichment also causes development of nuisance algal biomass (USEPA 2000c, Dodds and Welch 2000, Chambers et al. 2001). However, unlike lakes, the transition point at which the effects of nutrient enrichment shift from beneficial to detrimental has not been identified (Welch 1992 in USEPA 2000c).

5.1.3 Physico-Chemical Effects

As observed in lakes, stimulation of algal growth in streams increases turbidity and tends to reduce dissolved oxygen (USEPA 2000c). Phytoplankton blooms and dense mats of attached algae and macrophytes can cause significant diurnal DO swings and in extreme cases, can lead to fish kills. Additionally, death and senescence of primary producers can also deplete DO in streams. Effects of nutrient enrichment on DO depend on a variety of factors, including stream morphology, discharge, light, and temperature. Increased algal biomass leads to reduced water clarity, as observed in lakes (USEPA 2000c). In turn, increased turbidity can adversely affect macrophytes (through shading) but benefit attached algae (USEPA 2000c). Effects to pH (as a result of photosynthesis and respiration) may also occur, and in some instances, can lead to unfavourable conditions for invertebrates and fish.

5.2 FACTORS AFFECTING EUTROPHICATION/ALGAL BIOMASS

5.2.1 Nutrients

Nitrogen and phosphorus are the key nutrients controlling productivity in streams; there have been numerous studies in streams worldwide that have evaluated the relationship between nutrient concentrations and productivity (Bothwell 1989; Dodds 2006; USEPA 2000c). The following is a brief overview of the findings of selected studies on this issue.
5.2.1.1 Trophic Status

There have been fewer trophic categorization schemes developed for streams than for lakes. Several proposed schemes are summarized in Table 24. Carvalho et al. (2002) presented threshold criteria for European member states to “designate rivers subject to eutrophication” (Table 25). Dodds et al. (1998) developed a classification scheme based on a statistical analysis of nutrient concentrations and benthic chlorophyll \( a \) levels in hundreds of temperate streams from North America and New Zealand (Table 24). As the USEPA (2000c) pointed out, and which was later acknowledged by Dodds (2006), this scheme may reflect the effects of nutrient enrichment as the analysis was based on data collected after human settlement and did not exclude sites that may have experienced cultural eutrophication. Furthermore, the applicability of this categorization scheme to all environments is not known (Dodds and Welch 2000).

Table 24. Suggested boundaries for trophic classification of streams.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Oligotrophic</th>
<th>Mesotrophic</th>
<th>Eutrophic</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean benthic chlorophyll ( a )</td>
<td>(mg/m(^2))</td>
<td>&lt; 20</td>
<td>20-70</td>
<td>&gt; 70</td>
<td>Dodds et al. 1998</td>
</tr>
<tr>
<td>Maximum benthic chlorophyll ( a )</td>
<td>(mg/m(^2))</td>
<td>&lt; 60</td>
<td>60-200</td>
<td>&gt; 200</td>
<td>Dodds et al. 1998</td>
</tr>
<tr>
<td>Sestonic chlorophyll ( a )</td>
<td>(µg/L)</td>
<td>&lt; 10</td>
<td>10-30</td>
<td>&gt; 30</td>
<td>Van Niewwenhuysen and Jones 1996 In USEPA 2000c</td>
</tr>
<tr>
<td>TN</td>
<td>(mg/L)</td>
<td>&lt; 0.7</td>
<td>0.7-1.5</td>
<td>&gt; 1.5</td>
<td>Dodds et al. 1998</td>
</tr>
<tr>
<td>TP</td>
<td>(mg/L)</td>
<td>&lt; 0.025</td>
<td>0.025-0.075</td>
<td>&gt; 0.075</td>
<td>Dodds et al. 1998</td>
</tr>
</tbody>
</table>

Table 25. Threshold criteria used in European member states to designate rivers subject to eutrophication (Cardoso et al. 2001 In Carvalho et al. 2002).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>United Kingdom</th>
<th>Ireland</th>
<th>France</th>
<th>Joint Research Centre Ispra (Italy)</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP (mean)</td>
<td>(µg/L)</td>
<td>&gt; 100</td>
<td>&gt; 50</td>
<td>&lt; 100</td>
<td>75-200</td>
</tr>
<tr>
<td>Chlorophyll ( a ) (mean)</td>
<td>(µg/L)</td>
<td>&gt; 25</td>
<td>&gt; 60</td>
<td>&gt; 60</td>
<td>-</td>
</tr>
<tr>
<td>Chlorophyll ( a ) (maximum)</td>
<td>(µg/L)</td>
<td>&gt; 100</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
The USEPA (2000c) indicates that lake trophic classification schemes may also be applied to slow-moving rivers, as phytoplankton dominate in these systems. Development of predictive relationships between nutrients and periphyton are more problematic than for phytoplankton, and the USEPA (2000c) recommends the use of benthic chlorophyll $a$ as the basis for categorizing stream trophic status rather than nutrient concentrations in the river water. Furthermore, where nutrients are used as the basis for categorization, the USEPA (2000c) recommends the use of total nutrient fractions, rather than inorganic pools, as the basis. This is owing to the rapid depletion and recycling of the inorganic fractions of nutrients in stream ecosystems. However, use of total fractions of N and P as predictors of periphyton biomass is further compounded by the fact that these measures do not incorporate nutrients contained in the benthic biomass (whereas TN and TP do incorporate nutrients contained in phytoplankton biomass).

### 5.2.1.2 Nutrient Thresholds and Limiting Nutrients

Overall, biomass yields for a given concentration of nutrients are lower in streams than in lakes (Dodds and Welch 2000, reviewed in Dodds 2006), indicating that streams may have an inherently greater resistance to the effects of nutrient enrichment than lacustrine environments. Numerous studies have examined the relationship between nutrients and primary production in streams, with varying levels of success.

A number of studies have identified nutrient thresholds associated with growth saturation of periphyton at the cellular level or in mats (Table 26). In general, periphyton growth saturation at the cellular level occurs at very low ambient concentrations (generally 0.001-0.005 SRP mg/L), but saturation of mats of algae (i.e., areal biomass) is often reported to occur at somewhat higher concentrations (Table 26). Chambers et al. (2000b) reported that for oligotrophic cold-water rivers, nutrient saturation of cellular division of periphyton occurs at 1-5 µg/L of SRP and 2-3 µg/L DIN but that saturation in terms of areal biomass occurs at approximately 25-50 µg/L SRP and 15 µg/L DIN. The exact reason for this discrepancy is not known but may relate to the reduced ability of mats to accumulate ambient nutrients due to an increase in the laminar boundary close to the mat (Biggs and Stokseth 1996).
Table 26. Summary of cellular growth and algal mat nutrient saturation levels for periphyton.

<table>
<thead>
<tr>
<th>TP</th>
<th>TN</th>
<th>TDP</th>
<th>Ortho-P</th>
<th>SRP</th>
<th>DIN</th>
<th>Nitrates/ite</th>
<th>Type of Study</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>-</td>
<td>-</td>
<td>0.005</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Bothwell 1989</td>
<td></td>
<td>In Glozier et al. 2004</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.035</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Borchardt 1996</td>
<td></td>
<td>In Glozier et al. 2004</td>
</tr>
<tr>
<td>0.021</td>
<td>0.250</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Lohman et al. 1992</td>
<td></td>
<td>In Glozier et al. 2004</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.003</td>
<td>-</td>
<td>-</td>
<td>0.065</td>
<td>-</td>
<td>Chambers and Guy 2004</td>
<td></td>
<td>In Glozier et al. 2004</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.008</td>
<td>-</td>
<td>0.080</td>
<td>-</td>
<td>Cash et al. 2004</td>
<td></td>
<td>In Glozier et al. 2004</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.020</td>
<td>Walton 1990</td>
<td></td>
<td>In Biggs 2000a</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.007</td>
<td>Horner et al. 1990</td>
<td></td>
<td>In USEPA 2000c</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.002-0.005</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Chambers et al. 2000</td>
<td></td>
<td>In Scrimgeour and Chambers 2000</td>
</tr>
<tr>
<td>0.005</td>
<td>0.055</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Bothwell 1989, 1992</td>
<td></td>
<td>In Scrimgeour and Chambers 2000</td>
</tr>
<tr>
<td>&gt; 0.025-0.035</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Artificial streams (Alberta)</td>
<td>Chambers et al. 2000</td>
<td></td>
</tr>
<tr>
<td>&gt;0.040</td>
<td>&gt;0.009</td>
<td>&gt;0.05</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Athabasca and Wapiti-Smoky rivers, AB</td>
<td>Maximum biomass observed</td>
<td>Chambers et al. 2000</td>
</tr>
<tr>
<td>0.003-0.004</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Thompson River, BC</td>
<td>Algal (diatoms) growth rates &quot;nearly saturated&quot;</td>
<td>Bothwell 1985</td>
</tr>
<tr>
<td>0.0003-0.0006</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Experimental flumes, P-limited Thompson River, BC</td>
<td>Algal (diatoms) specific cellular growth rates saturated</td>
<td>Bothwell 1988</td>
</tr>
<tr>
<td>&gt; 0.03-0.05</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Experimental flumes, P-limited Thompson River, BC</td>
<td>Algal (diatom) peak areal biomass saturation</td>
<td>Bothwell 1989</td>
</tr>
<tr>
<td>0.001</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Experimental flumes, P-limited Thompson River, BC</td>
<td>Filamentous green and blue-green algae biomass</td>
<td>Bothwell 1992</td>
</tr>
<tr>
<td>0.0025 above background</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Experimental flumes, N-limited McKenzie River, Oregon</td>
<td>Filamentous green and blue-green algae biomass</td>
<td>Horner et al. 1988</td>
</tr>
</tbody>
</table>

Bothwell 1985, 1988, 1992
Scrimgeour and Chambers 2000
Welch et al. 1988
Research relating nutrient concentrations in natural streams to nuisance levels of biomass has been fairly extensively reported. Nutrient thresholds associated with "nuisance" levels of periphyton biomass have also been identified by a number of researchers, although the application of these thresholds to all systems is not known (Table 27). Sosiak (2002) reported that periphyton (mainly diatoms) biomass did not decrease in the Bow River downstream of the City of Calgary, despite P load reductions, when the ambient total dissolved phosphorus (TDP) was 10-33 µg/L, but did decline further downstream where TDP was <10 µg/L. Median concentrations of 6.4 µg/L TDP and 0.267 mg/L nitrate were identified as associated with maximum nuisance (150 mg/m²) levels of periphyton. Mainstone and Parr (2002) recently reviewed the issue of P in streams and suggested that the level of risk associated with P enrichment changes most rapidly from background conditions to approximately 200-300 µg/L TP.
Table 27. Summary of thresholds of nutrients and periphyton biomass to prevent nuisance conditions and water quality degradation in streams.

<table>
<thead>
<tr>
<th>TP</th>
<th>TN</th>
<th>SRP</th>
<th>DIN</th>
<th>Chlorophyll a</th>
<th>Impairment Risk</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>mg/L</td>
<td>mg/L</td>
<td>mg/L</td>
<td>mg/L</td>
<td>(mg/m²)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Nuisance growth</td>
<td>Welch et al. 1988, 1989</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>100-200</td>
<td>Tendency for filamentous species to dominate</td>
<td>Welch et al. 1988</td>
</tr>
<tr>
<td>0.03</td>
<td>0.350</td>
<td>-</td>
<td>-</td>
<td>&lt; 100</td>
<td>Prevent nuisance growth in most streams</td>
<td>Dodds et al. 1997</td>
</tr>
<tr>
<td>0.075</td>
<td>1.500</td>
<td>-</td>
<td>-</td>
<td>200</td>
<td>Eutrophy</td>
<td>Dodds et al. 1998</td>
</tr>
<tr>
<td>0.02</td>
<td>0.300</td>
<td>-</td>
<td>-</td>
<td>150</td>
<td>Nuisance growth</td>
<td>Clark Fork River Tri-State Council, MT</td>
</tr>
<tr>
<td>0.018</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>Nuisance periphyton (100 mg/m²) in the Bow River, AB</td>
<td>Sosiak 2002</td>
</tr>
<tr>
<td>&gt;0.02</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>Cladophora nuisance growth - 13 rivers in Ontario and Quebec</td>
<td>Chetelat et al. 1999</td>
</tr>
<tr>
<td>0.01-0.02</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Cladophora nuisance growth</td>
<td>Stevenson (Unpubl. Data)</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.060</td>
<td>0.430</td>
<td>-</td>
<td>Eutrophy</td>
<td>UK Environment Agency 1988</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.01</td>
<td>0.1</td>
<td>200</td>
<td>Nuisance growth</td>
<td>Biggs 2000a</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.003</td>
<td>0.025</td>
<td>100</td>
<td>Reduced invertebrate diversity</td>
<td>Nordin 1983</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.015</td>
<td>-</td>
<td>100</td>
<td>Nuisance growth</td>
<td>Quinn 1991</td>
</tr>
<tr>
<td>-</td>
<td>-</td>
<td>0.01</td>
<td>1.000</td>
<td>approx. 100</td>
<td>Eutrophy</td>
<td>Sosiak (pers. Comm.)</td>
</tr>
</tbody>
</table>

1 30-day biomass accrual time
2 TDP
In general, relationships between nutrients and periphyton biomass in streams are much weaker than relationships between phytoplankton and nutrients in lakes; establishment of clear thresholds and/or predictive equations requires further study (Dodds et al. 2002, Carr et al. 2005b). Periphyton biomass tends to exhibit significant temporal and spatial variabilities related to changes in environmental conditions, including discharge (i.e., flooding), temperature, velocity grazing, turbidity, and light. Furthermore, the periphyton growth cycle begins with a growth or 'accrual' phase followed by a 'sloughing phase' (Biggs 2000a), which may be accompanied by shifts in algal species dominance over this cycle. In turn, because species may respond differently to nutrient enrichment, the overall response of the entire community may vary in time depending on the stage of growth and the community composition at the point of nutrient enrichment.

Ultimately, periphyton biomass can vary considerably in time and space, making it difficult to measure or monitor and to construct predictive relationships between biomass and nutrients. Consequently, the results of studies examining the relationship between ambient nutrient concentrations and periphyton biomass have met with varying success. While it is difficult to compare regression models that differ in sample size and the magnitude of variation in both the dependent and independent variables, the overall predictive power of such relationships is generally lower than those constructed for phytoplankton (coefficients of determination are generally < 0.6). Dodds et al. (1997) generated a series of predictive equations relating concentrations of TN and TP to benthic algal biomass in streams using data from approximately 200 sites, although the correlation coefficients were quite low. Dodds et al. (2002) were able to explain approximately 40% of the variation in autotrophic biomass across a range of streams. Winter and Duthie (2000) reported similar but stronger relationships between TN and TP and epilithic diatoms for two streams in Ontario. Chambers et al. (2000b) reported that N and P were "moderately" correlated to periphyton biomass ($R^2 = 0.67$) for the Athabasca and Wapiti-Smoky rivers, AB. Despite relatively low predictive power of periphyton-nutrient equations, the USEPA (2000c) recommends the application of the equations produced by Dodds et al. (1997) where local information is lacking.

Numerous other studies have generated stronger predictive equations between nutrients and periphyton by incorporating additional variables in the correlations (Biggs 2000a, Vis et al. 1998). For example, Biggs (2000a) has proposed nutrient and benthic biomass criteria for New Zealand streams that incorporate flood events. Inclusion of flood event frequencies in the empirical model resulted in a greater predictive power than previous correlations, particularly for maximum benthic biomass (Biggs 2000b). The scheme proposed by Biggs (2000a) allows for higher nutrients in streams that are more hydrodynamically unstable. Similarly, Vis et al. (1998)
reported that using a combination of variables increased the predictability of periphyton biomass and diversity. The significance of factors other than nutrients in determining periphyton biomass is discussed further in Section 5.2.2.

Predictive relationships between nutrients and phytoplankton growth in rivers are stronger than periphyton relationships, although this subject has received less attention than periphyton in stream systems. Van Niewenhuyse and Jones (1996 In USEPA 2000c) generated a predictive equation relating phytoplankton biomass to nutrients, yielding a correlation coefficient of approximately 0.7. The USEPA (2000c) recommends the use of this equation where site-specific information is not available. Basu and Pick (1996) also generated a predictive equation relating TP to chlorophyll $a$ for Ontario and Quebec streams and Dodds (2006) later used the authors' raw data to generate a second equation for TN. Heiskary and Markus (2001) evaluated relationships between phytoplankton chlorophyll $a$ and nutrients in non-wadeable Minnesota rivers and found that TP was highly correlated to chlorophyll $a$ ($R^2 = 0.89$ and 0.91 for two years of study). TN also correlated to chlorophyll $a$ but the relationship was weaker ($R^2 = 0.25$ and 0.49); conversely TKN correlated even more strongly to chlorophyll $a$ than TP ($R^2 = 0.94$ and 0.96). The precise relationship between nutrients and phytoplankton also varies according to the nutrient concentrations. Yields of phytoplankton chlorophyll $a$ also tend to decrease per unit P, when TP exceeds 0.3 mg/L (Van Nieuwenhuyse and Jones 1996 In Dodds 2006).

Aquatic plants, on the other hand, tend to have adequate supplies of nutrients and are rarely nutrient limited. This is largely due to the abundant nutrient supplies in sediments and the ability of plants to use both aqueous and sediment nutrients for growth (USEPA 2000c). As a result, macrophyte biomass tends to be un-coupled from aqueous nutrient concentrations. Dodds and Welch (2000) recently indicated that they were unaware of any studies linking macrophyte biomass to nutrient concentrations in stream water and concluded that setting nutrient criteria for streams dominated by macrophytes would be difficult. Chambers (1993 In Sosiak 2002) reported that macrophyte biomass did not decline in the South Saskatchewan River following P load reductions from the City of Saskatoon. Chambers et al. (1999 In USEPA 2000c) indicated that the effects of nutrient enrichment can be cancelled out by competition with planktonic and epiphytic algae and reductions in light.

More recently, some research has linked nutrients in water to macrophytes. For example, macrophyte biomass declined in the Bow River following initial P load reductions and later even more markedly following nitrogen reductions (Sosiak 2002). Reductions in biomass following N load reductions were believed to be related to concomitant decreases of nitrogen in sediments. Biomass was reduced when median ambient concentrations of ammonia and nitrate/nitrite were reduced to 0.11-0.28 mg/L and 0.47-0.81 mg/L, respectively. Carr et al. (2003) found that mean
and maximum biomass of macrophytes from 28 rivers in southwestern Ontario were weakly but significantly correlated to N and P. The authors suggested that management programs focusing primarily on N would be successful at reducing macrophyte biomass.

Effects of nutrients may also vary according to the species of macrophyte. For example, Flynn et al. (2002) found a weak positive correlation between phosphorus in water and total percent macrophyte cover in the River Kennet, UK, but a negative correlation with the species *Ranunculus*.

Nitrogen is more commonly limiting for phytoplankton and periphyton growth in streams than in lakes and there are numerous examples indicating nitrogen limitation in stream environments (USEPA 2000c, Dodds et al. 2002, Dodds and Welch 2000, reviewed in Dodds 2006, Biggs 2000a, Biggs et al. 1998, Scrimgeour and Chambers 2000, Bothwell 1992), including Manitoba (e.g., Armstrong 2002). With respect to periphyton, nitrogen limitation is commonly reported for watersheds with rich geological sources of P (e.g., McKenzie River, Oregon, Bothwell 1992, Nechako River, BC, Perrin and Richardson 1997) or that are dominated by agriculture (reviewed in Scrimgeour and Chambers 2000). For example, Dodds et al. (1997), Scrimgeour and Chambers (2000), Chambers et al. (2000b), Taulbee et al. (2005), and Carr et al. (2005a) found that nitrogen was more closely correlated to benthic algal biomass than phosphorus. Perrin and Richardson (1997) observed a greater increase in periphyton biomass following N additions, relative to P additions, with the latter not differing significantly from controls.

Alternatively, there is evidence of P limitation in stream ecosystems (Bothwell 1985, Dodds et al. 1998, Welch et al. 1992, Bemstead et al. 2005) and many examples of co-limitation (e.g., Wapiti River, AB, Chambers et al. 2000b, Biggs et al. 1998). Numerous studies report a greater increase in algal biomass with addition of both N and P, relative to either alone (e.g., Perrin and Richardson 1997). Additionally, periphyton may exhibit 'luxury consumption' of P, where periphyton accumulate P in excessive amounts to take advantage of pulses of P in the environment. Therefore, periphyton can actually be N limited when aqueous N:P ratios indicate P limiting conditions (reviewed in Dodds and Welch 2000). Nonetheless, the limiting nutrient or nutrients will vary between systems and over time; Francoeur (2001), for example, demonstrated that periphyton may be N, P, or N and P limited and that N was as equally likely to be limiting as P. He also noted that multi-species algal communities are not likely to be limited by one nutrient. Sosiak (2002) suggested that aquatic plants may have been nitrogen limited in the Bow River, AB downstream of the City of Calgary and reviewed other studies that had reached the same conclusion. A similar conclusion was reached by Carr et al. (2003) who advocate N controls for macrophyte biomass reduction.
Nutrient limitation may also change in relation to point source discharges. Nitrogen typically becomes limiting due to eutrophication because point sources generally have low N:P ratios, denitrification may be augmented, and P is more readily deposited in sediments (USEPA 2000c). Additionally, point source discharges often relieve nutrient deficiencies for some distance downstream and the occurrence of nutrient saturating conditions and P or N limitation may vary spatially in a river, depending on nutrient sources. For example, the Athabasca River alternates between P or N limitation, N and P co-limitation, and nutrient saturation along its length in relation to point source discharges and tributary inflows (Chambers et al. 2000b).

As discussed for lakes, N:P ratios can also be applied to streams to assist in determining which nutrient is limiting (Table 23). However, periphyton TN:TP ratios may not parallel ambient ratios and measurement of TN:TP in benthic algae directly may provide more reliable information on which nutrient is limiting (USEPA 2000c). Similarly, because aquatic plants obtain most of their nutrients from sediments, aqueous nutrient concentrations may not adequately reflect nutrient conditions for macrophyte growth (Welch 1992 In USEPA 2000c).

Several studies have demonstrated the potential for developing and using chlorophyll a-nutrient relationships according to ecoregions (e.g., Dodds et al. 2002, Carr et al. 2005b), but additional study on this issue is required. Identification of baseline conditions in "pristine" streams within an ecoregion would be most appropriate, if differences between ecoregions do exist. This is particularly important where 'natural' enrichment of streams may occur, such as watersheds with P-rich rocks. Site- or region-(ecoregion) specific models relating nutrients to biomass tend to be stronger than predictive equations constructed with data from a wide geographical area (Dodds et al. 2002). Additionally, as Chambers and Guy (2004) point out, the 'breakpoint' from acceptable to unacceptable conditions due to nutrient enrichment varies among ecosystems and "it is not appropriate to advocate a single N or P concentration as protective of a designated water use nation-wide". Carr et al. (2005b) advocate using the ecoregion modeling approach to define relationships between nutrients and chlorophyll a, based on the results of a study of a 21-year data set for Alberta rivers.
5.2.2 Physical Factors

As indicated above, nutrient dynamics and effects to primary production are affected by various factors in streams, namely:

- Discharge, detention time, and scouring;
- Velocity;
- Depth;
- Shading;
- Water clarity;
- Water temperature;
- Grazing;
- Substrata; and
- Stream morphology.

In general, low river discharges favour uptake of nutrients and growth of attached algae and phytoplankton (EC 2004), largely due to effects of related variables such as water depth and light availability. Low flow conditions tend to favour the development of large phytoplankton blooms in nutrient-rich rivers (USEPA 2000c). However, relationships between river discharge and nutrients are variable. On the one hand, dilution of point sources of nutrients is greater under high flows. Conversely, effects of non-point sources are generally greater during periods of high discharge as they are associated with high precipitation and runoff. In addition, Flynn et al. (2002) found that macrophyte percent cover and biomass was best predicted by river discharge (negative correlation).

Hydrology, specifically the occurrence of flood events, is also highly significant in determining the biomass and composition of periphyton communities (Biggs 2000a, Biggs et al. 1998). Flooding causes sloughing of periphyton and frequent flooding events tend to limit the maximum accrual of periphyton biomass; the longer the period of time between the flood disturbances, the greater the accrual of periphyton biomass. Therefore, environments with high flooding frequencies tend to have lower periphyton biomass (Biggs and Close 1989, Biggs 2000a, Biggs et al. 1998). Biggs and Close (1989) found that "percent time in flood" was the most important variable explaining periphyton biomass in New Zealand streams. The significance of flood events on periphyton biomass has supported the development of site-specific models relating nutrients to algal biomass (Dodds et al. 2002). Additionally, because periphyton communities undergo succession between flood events, the duration between these disturbances will also
affect the community composition. Environments with low flood frequencies tend to be more
dominated by climax communities of filamentous cyanobacteria and green algae, whereas riffle
sites experiencing frequent flood disturbances tend to be dominated by diatoms (Biggs et al.
1998).

Velocities may affect the ability of attached algae to accumulate nutrients, with uptake increasing
with velocity until a critical level when drag is increased and attached algae are sloughed from
their substrates. For example, Welch et al. (1988) found that chlorophyll $a$ increased linearly
with velocity (maximum velocity was approximately 22 cm/s) across sites in Washington
streams (note that velocities were not high enough to cause sloughing). Additionally, velocities
affect the presence and type of aquatic plants and periphyton in streams (USEPA 2000c, Biggs
2000a).

The precise relationship between velocities and periphyton biomass/growth is complex and
appears to vary with the stage of accrual and community composition, and potentially, nutrients.
Dense mats may require higher velocities to obtain sufficient nutrients when ambient nutrients
are low but when nutrient supply is great, velocity limitations are removed and high velocities
may only impose detrimental effects (Biggs and Stokseth 1996). Communities dominated by
stalked diatoms and filamentous algae may exhibit negative correlations between biomass and
velocity whereas growth rates of prostrate diatoms may increase with velocity (Biggs and
Stokseth 1996). One explanation for this could be that filamentous algae experience greater drag
at high velocities, which in turn limits biomass. Velocities also affect phytoplankton growth; as
velocity decreases and therefore, river travel time increases, more time is available for
phytoplankton to grow in a given segment of a stream.

Water clarity and/or light availability may be limiting in some systems, particularly turbid prairie
streams which carry a high concentration of suspended solids. Additionally, light availability in
streams is affected by canopy cover. In a recent review of periphyton in streams, Biggs (2000a)
summarized the factors affecting periphyton biomass and indicated that nutrients and light were
considered the primary limiting factors, with temperature also playing a significant role. Streams
with low nutrient concentrations, that are light-limited and scour-dominated, are characterized by
periphyton biomass less than 50 mg/m$^2$. In some systems light limitation may attenuate the
effects of nutrient enrichment on algal biomass (USEPA 2000c). Greenwood and Rosemond
(2005) reported that periphyton biomass and composition in two headwater mountain streams,
North Carolina, were largely unaffected by long-term nutrient enrichment, likely due to light
limitation (streams have intact canopies) and/or grazing. Seven of eight rivers studied in the US
National Water Quality Inventory study did not have nuisance algal growth in spite of high
nutrient concentrations; this observation was attributed to high turbidity and light limiting
conditions (reviewed in Litke 1999). Turbidity has been found to limit periphyton growth at levels between 7 and 23 NTUs (Quinn et al. 1992 in USEPA 2000c). TSS levels of 22-30 mg/L were associated with reduced periphyton biomass (Horner et al. 1990 in USEPA 2000c).

Phytoplankton biomass in deep, slow-moving rivers is limited by light attenuation, which increases with algal biomass (USEPA 2000c). The theoretical maximum phytoplankton biomass than can be achieved in rivers, before light attenuating effects of the algae themselves results in light limiting conditions, is 250 mg/m³ (USEPA 2000c). Non-algal turbidity may further limit phytoplankton growth, particularly in deep, slow-moving rivers. Furthermore, turbidity or TSS tend to vary according to river discharge, and as a result, light limitation may exhibit considerable seasonal variability in relation to storm events.

The significance of light limitation may vary significantly according to the group of primary producers. Biggs (2000a) indicated that a large reduction in light intensity (> 60%) is needed, at least for New Zealand streams, to have a significant effect on periphyton growth. Conversely, macrophytes are commonly limited by light and typically light determines the maximum depth at which aquatic plants grow (USEPA 2000c). In addition to effects on light availability, high levels of TSS may reduce periphyton biomass through physical effects, namely increased drag. TSS has been negatively correlated to periphyton biomass in some streams (e.g., Oria River, Spain, Sabater et al. 2000).

Water temperature may be limiting in some environments either year-round (e.g., Arctic streams) or seasonally (e.g., north temperate streams). In general, higher temperatures increase algal (periphyton and phytoplankton) growth and the effects of nutrient enrichment are typically greater at higher temperatures (USEPA 2000c). Additionally, temperature affects the algal species composition, as species display varying thermal preferences and tolerance ranges. These preferences give rise to seasonal succession of algal taxa, typically beginning with diatom dominance in spring, with dominance shifting to greens and cyanobacteria over the summer. As discussed for lakes, cyanobacteria tend to prefer higher water temperatures and diatoms lower ones. According to Storr and Sweeney (1991 In USEPA 2000c), *Cladophora* prefers water temperatures of approximately 18 °C and does not grow above 25 °C. Although temperature is a critical governing factor in lotic systems, at low nutrient concentrations diatoms tend to dominate over the whole growing season and at high nutrient concentrations, green and blue-green algae dominate for the whole period (USEPA 2000c).

Substrate is an important variable determining the presence and distribution of both aquatic plants and attached algae (USEPA 2000c). Sediment type also affects the type of aquatic plants and periphyton in lotic systems (USEPA 2000c). Biggs (2000a) reviewed factors affecting
stream periphyton and indicated that in general, the larger, more stable the substratum, the
greater the periphyton biomass. Biomass increases in the order: sand < gravel < cobble <
boulder < bed rock, in part due to differences in substratum availability. Additionally, the
community structure is determined by substratum with filamentous taxa dominating on large
substrates and unicellular diatoms on fine substrates.

5.2.3 Biological Factors

Grazing can limit algal growth in streams, even in enriched systems, and the occurrence of a
trophic cascade or top-down controls has been demonstrated in lotic ecosystems as it has in
lacustrine environments (USEPA 2000c). However, the use of this pathway (i.e.,
biomanipulation) for controlling algal biomass is not as far advanced as it is for lakes.
Interestingly, the occurrence of top-down control of phytoplankton in rivers is associated with
conditions similar to those coined the "clear water state" in lakes. That is, severe biomass
reduction of phytoplankton by filter-feeding grazers is typically associated with reduced turbidity
and proliferation of macrophytes but differs from the clear water state in lakes in that dissolved
nutrients tend to increase (USEPA 2000c).

Caraco et al. (2006) recently demonstrated that over a 15-year period, phytoplankton were
controlled primarily by grazers in the Hudson River, New York and biomass was not closely
related to either nutrients or flow. Grazing by the invasive zebra mussel explained 90% of the
variation in chlorophyll a. However, blooms of cyanobacteria periodically occurred despite the
top-down controls. Welch et al. (1992) estimated that low periphyton is maintained where grazer
densities exceed 3,000/m².

Also similar to lakes, macrophytes serve an important role in streams, primarily acting as buffers
for eutrophication. Macrophytes can inhibit phytoplankton growth, stabilize sediments, reduce
sediment resuspension, reduce flow velocities, and reduce sediment transport (USEPA 2000c).
As the USEPA (2000c) indicated "to obtain the desired biological integrity of an aquatic
community, macrophytes should be present and healthy". Aquatic plants are also good
indicators of nutrient conditions in streams as their presence/absence and abundance can signal
nutrient deficiency or excess.
5.3 NUTRIENT CRITERIA

Various jurisdictions in Canada, including national and provincial agencies, as well as nations world-wide have suggested or adopted nutrient criteria for streams for the prevention of nuisance plant or algal growth, as well as actual criteria for algal biomass. In many instances, the criteria are generic and have not been derived for site-specific waterbodies or regions, beyond the provincial boundary scales. A summary of current provincial and CCME guidelines for streams, as well as proposed site- or region-specific criteria for selected sites in Canada, are provided in Table 28.
Table 28. Summary of provincial, national, region- or site-specific nutrient guidelines for streams in Canada.

<table>
<thead>
<tr>
<th>Region</th>
<th>Usage</th>
<th>TP (mg/L)</th>
<th>TN (mg/L)</th>
<th>Chlorophyll a Phytoplankton (mg chlorophyll a/m²)</th>
<th>Periphyton</th>
<th>Notes</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>Aquatic Life</td>
<td>None</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Notes: To prevent nuisance growth and reproduction of aquatic rooted,</td>
<td>BCMWLAP 1998</td>
</tr>
<tr>
<td></td>
<td>Recreation</td>
<td>None proposed</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>attached and floating plants, fungi, or bacteria or to otherwise</td>
<td></td>
</tr>
<tr>
<td>Alberta</td>
<td>Aquatic Life</td>
<td>0.050</td>
<td>1.0</td>
<td>-</td>
<td>-</td>
<td>render the water unsuitable for other beneficial uses.</td>
<td>AENV 1999</td>
</tr>
<tr>
<td>Montane ecoregion, Alberta</td>
<td>NA</td>
<td>0.002</td>
<td>0.100 (as</td>
<td>-</td>
<td>-</td>
<td>Interim guidelines</td>
<td>Chambers and Guy 2004</td>
</tr>
<tr>
<td>Lower foothills ecoregion,</td>
<td>NA</td>
<td>0.003</td>
<td>0.105 (as</td>
<td>-</td>
<td>45</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alberta</td>
<td>NA</td>
<td>0.004</td>
<td>0.074 (as</td>
<td>-</td>
<td>12</td>
<td></td>
<td></td>
</tr>
<tr>
<td>dry mixedwood ecoregion,</td>
<td>NA</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alberta</td>
<td>Bow River near</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Sosiak 2002</td>
</tr>
<tr>
<td>Calgary, AB</td>
<td>NA</td>
<td>0.018</td>
<td>-</td>
<td>-</td>
<td>150 (maximum)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Saskatchewan</td>
<td>All (see note)</td>
<td>0.025</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>To prevent nuisance growth and reproduction of aquatic rooted,</td>
<td>SERM 1997</td>
</tr>
<tr>
<td>Manitoba</td>
<td>(tributary at the</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>attached and floating plants, fungi, or bacteria or to otherwise</td>
<td>Williamson 2002</td>
</tr>
<tr>
<td></td>
<td>point of entry to</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>render the water unsuitable for other beneficial uses.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>reservoirs, lakes,</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>or ponds)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.050</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ontario</td>
<td></td>
<td>0.030</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Interim; considered to be “general guidelines”</td>
<td>Ontario MOEE 1994</td>
</tr>
</tbody>
</table>
Table 28. Continued.

<table>
<thead>
<tr>
<th>Region</th>
<th>Usage</th>
<th>TP (mg/L)</th>
<th>TN (mg/L)</th>
<th>Chlorophyll $a$ Phytoplankton (mg chlorophyll $a$/m²)</th>
<th>Chlorophyll $a$ Periphyton (mg chlorophyll $a$/m²)</th>
<th>Notes</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quebec</td>
<td>Aquatic Life and Recreation</td>
<td>0.03</td>
<td>-</td>
<td></td>
<td></td>
<td></td>
<td>Streams Ministere de L’Environnement du Quebec 2004</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.02</td>
<td>-</td>
<td></td>
<td></td>
<td></td>
<td>In streams entering lakes</td>
</tr>
<tr>
<td>National (CCME)</td>
<td>Aquatic Life</td>
<td>Not to exceed trigger ranges; and not to increase by more than 50% above “baseline” level</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>Trigger Ranges: &lt; 0.004 mg/L: ultra-oligotrophic; 0.004-0.01 mg/L: oligotrophic; 0.01-0.02 mg/L: mesotrophic; 0.02-0.035 mg/L: meso-eutrophic; 0.035-0.100 mg/L: eutrophic; &gt; 0.100 mg/L: hyper-eutrophic</td>
</tr>
</tbody>
</table>
5.3.1 CCME Phosphorus Guidance Framework

The CCME (2004) recently issued a phosphorus guidance framework for management of Canadian freshwater ecosystems as described in detail in Section 4.6.1. The framework incorporates the OECD (1982) trophic categorization scheme for TP with a slight adjustment (Table 29); the OECD (1982) mesotrophic category has been split into two categories due to the large variation in community composition and biomass that reportedly occurs over the range of 10-35 µg/L in Canadian freshwater systems. Under this framework, ecosystem objectives are identified (enhance, protect, or restore), the reference condition is established (see Section 5.3.2 for a discussion of common approaches) and the current TP concentration is compared to the associated trigger range defined on the basis of reference conditions. Additionally, the framework also includes an additional ‘clause’ indicating that increases in TP greater than 50% above the baseline are construed as increased risk.

The trophic categorization and trigger ranges are intended to be applied to both lakes and rivers (CCME 2004). However, as Environment Canada (2004) indicates, streams can typically sustain higher concentrations of phosphorus than lakes “without observable changes in community composition and biomass” due to more rapid flushing of phosphorus from these ecosystems. Environment Canada (2004) further acknowledged that the OECD trophic categorization scheme is more stringent than others that have been developed for streams, such as that developed by Dodds et al. (1998) and applied by the USEPA (2000c). Therefore, Environment Canada (2004) indicates that these alternative schemes could be applied, where warranted, to specific stream ecosystems (e.g., large rivers, sites with high baseline concentrations).
Table 29. USEPA nutrient criteria based on aggregate ecoregions: streams and rivers. NA = criteria not available (USEPA 2002). Ecoregion VI is located south of Manitoba.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Aggregate Ecoregion</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>I</td>
</tr>
<tr>
<td>TP</td>
<td>(mg/L)</td>
<td>0.047</td>
</tr>
<tr>
<td>TN</td>
<td>(mg/L)</td>
<td>0.31</td>
</tr>
<tr>
<td>Chl a</td>
<td>(µg/L)</td>
<td>1.80</td>
</tr>
<tr>
<td>Turbidity</td>
<td>(FTU/NTU)</td>
<td>4.25</td>
</tr>
</tbody>
</table>

1 This value appears inordinately high and may either be a statistical anomaly or reflects a unique condition. In any case, further regional investigation is indicated to determine the sources, i.e., “measurement error, notational error, statistical anomaly, natural enriched conditions, or cultural impacts”.

I = Willamette and central valleys;
II = western forested mountains;
III = xeric west;
IV = great plains grass and shrubland
V = south central cultivated great plains;
VI = corn belt and northern great plains;
VII = mostly glaciated dairy region;
VIII = nutrient-poor largely glaciated upper Midwest and northeast;
IX = southeastern temperate forested plains and hills;
X = Texas-Louisiana coastal and Mississippi alluvial plains;
XI = central and eastern forest uplands;
XII = southern coastal plain;
XIII = southern Florida coastal plain; and
XIV = eastern coastal plain.
5.3.2 Defining Baseline Nutrient and Algal Conditions: Reference Conditions

Site-specific criteria (i.e., for a waterbody, ecoregion, or segment of a stream) have been proposed or adopted by various agencies and derivation of site- or region-specific criteria for nutrients in streams is generally advocated (e.g., Chambers et al. 2000b, Dodds et al. 2002, Vis et al. 1998, Chambers and Guy 2004). There are a variety of methods that can be used to derive site-specific nutrient criteria for freshwater streams and the methods are similar to those suggested for lakes (see Section 4.6.2 for a detailed discussion). These methods generally fall into the following approaches:

- **Use of the reference condition approach:** ‘pristine’ or reasonably unaltered conditions for a given stream or segment of a stream are defined as a statistical condition derived for appropriate reference streams. This is often done through use of reference streams in the same ecoregion and by quantitatively defining percentiles of nutrient concentrations.
- **Use of historical data:** criteria for a given stream are based on historical conditions (e.g., pre-development).
- **Water quality models:** models may be developed to estimate conditions for a past period of time and/or to evaluate site-specific considerations relevant to establishing nutrient criteria.
- **Scientific/ecological studies:** criteria may be derived and/or refined based on site-specific studies of nutrient – algal growth relationships.

There are numerous water quality models available to simulate nutrients and trophic effects for streams. These include models such as the 2-dimensional QUAL2K, the 3-dimensional WASP, and HSPF, among others. Additionally, the USEPA Basins model offers users the ability to integrate land use effects through a watershed-scale approach. Until fairly recently, most water quality models for streams lacked a periphyton component. However, WASP and QUAL2K have recently incorporated a periphyton component. More simplistic models (e.g., empirical equations) can also be employed where predictive relationships between nutrients and algal biomass have been derived.

Use of reference conditions for derivations of site-specific nutrient criteria is another commonly applied approach. For example, Dodds et al. (1997) derived nutrient criteria for the Clark Fork River, Montana, using reference conditions. More recently, Chambers and Guy (2004) proposed nutrient criteria for several ecoregions in Alberta, based on defined region-specific reference conditions. Reference river reaches were defined as either upstream of all point sources or > 100 km downstream of a point source.
5.3.3 USEPA Nutrient Guidance Framework

As a component of the national strategy for the development of nutrient criteria, the USEPA has issued a guidance document for the derivation of site-specific nutrient criteria for streams (USEPA 2000c). This technical guidance manual was published to assist States and Tribes with the development of regional nutrient criteria (USEPA 2000c). Typically, the USEPA produces numerical criteria which are then adopted by state agencies as water quality standards across the nation (i.e., a single criterion is developed to be applicable to all waterbodies across the country). However, because there is a great deal of variability in nutrient concentrations and responses of aquatic ecosystems to nutrient enrichment, a single numerical criterion could not be generated for the entire nation. Rather, the guidance document describes methods and approaches through which site-specific nutrient criteria can be developed and provides suggested nutrient criteria as a 'starting point'.

The USEPA developed 'nutrient criteria guidance' for nitrogen, phosphorus, (i.e., the "causal indicators") and 'other nutrient parameters' including chlorophyll a and turbidity (i.e., the "response indicators") under Section 304(a) of the Clean Water Act. Criteria have been derived for given waterbody types (i.e., streams, lakes/reservoirs, and estuaries) and aggregate ecoregion. The intent of these criteria is to provide states with a starting point for development of state or region-specific nutrient standards defined to support designated uses of waters. States would examine the candidate criteria and further develop or modify the criteria to become more site-specific.

USEPA criteria for streams in 13 of the 14 aggregate ecoregions are presented in Table 29. These criteria are intended as a starting point for the development of state standards and were derived from actual conditions measured in streams within each of the aggregate ecoregions. As is immediately evident, reference conditions vary considerably across the ecoregions. TP concentrations range from 0.01 to 0.076 mg/L, TN ranges from 0.12 to 2.18 mg/L, chlorophyll a ranges from 0.40 to 3.75 µg/L and turbidities range from 1.30 to 17.50 formazin turbidity units (FTU)/nephelometric turbidity units (NTU). The highest TP9 and TN concentrations occurred in aggregate ecoregion VI, the corn-belt and northern Great Plains. This includes parts of North and South Dakota, Minnesota, Iowa, Illinois, Michigan, Ohio, Nebraska, Indiana, and a small portion of Wisconsin, and is characterized by nutrient-rich soils (USEPA 2000d).

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9 A very high TP concentration (0.128 mg/L) was derived for ecoregion X, but the USEPA believes this is either an anomaly or a unique condition.
Table 30. Reference conditions for streams in aggregate ecoregion VI, Level III, ecoregions 46 and 48 (USEPA 2000d).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Ecoregion 46</th>
<th>Ecoregion 48</th>
</tr>
</thead>
<tbody>
<tr>
<td>TP</td>
<td>(mg/L)</td>
<td>0.10238</td>
<td>0.0875</td>
</tr>
<tr>
<td>TN</td>
<td>(mg/L)</td>
<td>1.4</td>
<td>1.158</td>
</tr>
<tr>
<td>Chlorophyll a</td>
<td>(µg/L)</td>
<td>14</td>
<td>-</td>
</tr>
<tr>
<td>Turbidity</td>
<td>(FTU/NTU)</td>
<td>4.33</td>
<td>6.14</td>
</tr>
</tbody>
</table>

Reference conditions for streams in the two ecoregions bordering the province of Manitoba are presented in Table 30. Reference conditions for ecoregion 46, the northern glaciated plains, indicate a higher TP and chlorophyll $a$ and a lower TN and turbidity, than the aggregate ecoregion as a whole (USEPA 2000d). Similarly, TN and turbidity were lower and TP higher in ecoregion 48, the Lake Agassiz plain, than the aggregate ecoregion as a whole.

The technical guidance manual for streams describes a process for developing nutrient criteria that entails the following nine steps (USEPA 2000c):

- Identify water quality needs and goals;
- Classify rivers and streams first by type and then by trophic status;
- Select variables for monitoring nutrients;
- Design a sampling program for monitoring nutrients and algal biomass in rivers and streams;
- Collect data and build database;
- Analyse data;
- Develop nutrient criteria based on reference conditions and data analyses;
- Implement nutrient control strategies; and
- Monitor effectiveness of nutrient control strategies and reassess the validity of nutrient criteria.

It is also noted that many of the above steps will have already been completed and the process may focus primarily on the latter steps (i.e., derivation of nutrient criteria).
Developing Nutrient Criteria

The USEPA (2000c) recommends three methods for developing nutrient criteria:

- Identification of reference conditions;
- Development and application of predictive relationships; and
- Use and application of thresholds in the scientific literature.

Reference conditions can be derived through characterization of references reaches for each stream class, identification of the 75th percentile of a frequency distribution of reference streams for a class of streams, or derivation of the 5th to 25th percentile of the frequency distribution of the general population of a class of streams (USEPA 2000c). The USEPA (2000c) specifies that a minimum of three minimally impaired reference systems should be used for each stream class where identifying reference conditions using the first method.

The predictive relationships approach might entail use of trophic state classifications, models, and/or biocriteria. Empirical models constructed using nutrients as well as other physical and biological variables that are known to alter algal biomass tend to be more accurate than models based on nutrients alone. For example, the periphyton models of Biggs (2000b) in which both nutrients and hydrology are considered are better predictors of biomass than other models. Alternatively or additionally, published nutrient thresholds, such as those presented in Table 14, could be evaluated as potential nutrient criteria. However, the applicability and suitability of a published threshold for a given stream reach, stream, or region must be verified. Ideally, a combination of all of these approaches would be used as a 'weight of evidence' approach in developing criteria. Criteria must also consider effects to downstream environments, including lakes (USEPA 2000c).

5.3.4 Other Jurisdictions

Numerous other jurisdictions have developed nutrient and algal biomass criteria for streams but detailed discussion of this literature is beyond the scope of this review. However, New Zealand has recently derived nutrient and biomass criteria (tables 31 and 32) based on the current scientific information and a brief discussion of these criteria is particularly useful. Biggs (2000a) proposed nutrient and benthic biomass criteria for New Zealand streams that incorporate flood events (Table 32). This scheme allows for higher nutrients in streams that are more hydrodynamically unstable as it acknowledges the relationship between period of time between flood events (and therefore periphyton sloughing) and biomass accrual.
Table 31. Provisional biomass and cover guidelines for periphyton in gravel/cobble bed streams (Biggs 2000a).

<table>
<thead>
<tr>
<th>Usage</th>
<th>Parameter</th>
<th>Diatoms / cyanobacteria</th>
<th>Filamentous algae</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aesthetic Recreation (in season)</td>
<td>Maximum cover of visible stream bed</td>
<td>60% &gt; 0.3 cm thick</td>
<td>30% &gt; 2 cm long</td>
</tr>
<tr>
<td></td>
<td>Maximum AFDM (g/m²)</td>
<td>N/A</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td>Maximum chlorophyll a (mg/m²)</td>
<td>N/A</td>
<td>120</td>
</tr>
<tr>
<td>Benthic biodiversity</td>
<td>Mean monthly chlorophyll a (mg/m²)</td>
<td>15</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>Maximum chlorophyll a (mg/m²)</td>
<td>50</td>
<td>50</td>
</tr>
<tr>
<td>Trout habitat and angling</td>
<td>Maximum cover of whole stream bed</td>
<td>N/A</td>
<td>30% &gt; 2 cm long</td>
</tr>
<tr>
<td></td>
<td>Maximum AFDM (g/m²)</td>
<td>35</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td>Maximum chlorophyll a (mg/m²)</td>
<td>200</td>
<td>120</td>
</tr>
</tbody>
</table>

Table 32. Predicted nutrient thresholds associated with limiting periphyton biomass in New Zealand streams. Nutrient concentrations are mean monthly concentrations over a year (Biggs 2000a).

<table>
<thead>
<tr>
<th>Days of Accural</th>
<th>Chl a - 50 mg/m²</th>
<th>AFDM = 35g/m²</th>
<th>Chl a - filamentous green algae-dominated = 120mg/m²</th>
<th>Chl a - diatom-dominated = 200mg/m²</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>SIN (mg/L)</td>
<td>SRP (mg/L)</td>
<td>SIN (mg/L)</td>
<td>SRP (mg/L)</td>
</tr>
<tr>
<td>20</td>
<td>&lt; 0.02</td>
<td>&lt; 0.001</td>
<td>&lt; 0.295</td>
<td>&lt; 0.026</td>
</tr>
<tr>
<td>30</td>
<td>&lt; 0.01</td>
<td>&lt; 0.001</td>
<td>&lt; 0.075</td>
<td>&lt; 0.006</td>
</tr>
<tr>
<td>40</td>
<td>&lt; 0.01</td>
<td>&lt; 0.001</td>
<td>&lt; 0.034</td>
<td>&lt; 0.0028</td>
</tr>
<tr>
<td>50</td>
<td>&lt; 0.01</td>
<td>&lt; 0.001</td>
<td>&lt; 0.019</td>
<td>&lt; 0.0017</td>
</tr>
<tr>
<td>75</td>
<td>&lt; 0.01</td>
<td>&lt; 0.001</td>
<td>&lt; 0.010</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>100</td>
<td>&lt; 0.01</td>
<td>&lt; 0.001</td>
<td>&lt; 0.010</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>


5.4 NUTRIENT/EUTROPHICATION MANAGEMENT

There are numerous issues and actions to consider for the management of nutrient overenrichment of streams including (USEPA 1998):

- Evaluate effects of land use change on waterbodies;
- Altering land use practices in the watershed;
- Enhance nitrogen-phosphorus cycling on different land uses to reduce mobilization;
- Riparian zone management (e.g., enhance shading of streams);
- Channel restoration;
- Biological controls (e.g., introduce efficient periphyton grazers);
- Hydrology and hydraulics (i.e., identify natural flow regimes and use this information to better replicate natural conditions in the area while generating power or preserving intended reservoir levels);
- Remove obsolete impoundments;
- Restore riparian and floodplain wetlands;
- Point source nutrient reductions or elimination; and
- Storm water management.

Many efforts have been made to reduce nutrient releases to freshwater streams, in particular, there has been a concerted effort to reduce P discharges from municipal WWTFs. These reductions have had variable levels of success in reducing or eliminating eutrophication of streams. Stream recovery processes have been slow and variable. Chambers et al. (1989 In Chambers et al. 1997) indicated that prairie rivers are particularly slow to respond to P reductions, which may be explained in part by the significance of aquatic plants in these ecosystems. Aquatic plants can obtain nutrients from sediments, thereby showing "a cumulative response to nutrient loading" and a reliance on the need for reductions in sediment nutrient concentrations (i.e., lag times). Alexander and Smith (2006) in a recent analysis of long-term temporal trends in US streams demonstrated that while nutrient concentrations have declined at many sites over the last several decades, fewer sites have shifted trophic condition.
The USEPA (2000c) indicates that there are comparatively fewer examples for streams than for lakes where recovery has been thoroughly studied. The Bow River, AB, is one exception. The Bow River is a fast-flowing, gravel-bed river that had problems with excessive algae (periphyton biomass averaged approximately 300-400 mg chlorophyll $a/m^2$) and plant growth (biomass $>2,000$ mg/m$^2$) until nutrient loading reduction was implemented. Phosphorus load reductions from municipal WWTFs (80%) caused a notable but slow decline in TDP concentrations and periphyton biomass in the river. The delay in recovery may have reflected slow reductions in sediment TP. Similarly, Jarvie et al. (2005) suggested that where river water SRP concentrations were reduced to $< 0.05$ mg/L, SRP release from sediments could become more important. Nitrogen loading was later reduced from the municipal WWTFs for the purposes of reducing ammonia. Macrophyte biomass declined rapidly after the loading reductions to levels of $< 200$ mg/m$^2$ although the mechanism for this change is not known. It was postulated that the nitrogen reduction may have increased N limitation at the roots, as nitrogen was not considered limiting (DIN:TDP ratios were consistently over 20).

Defining which nutrient is most limiting is a critical step in nutrient management (USEPA 2000c). As indicated in Section 5.2.1.2, nitrogen is frequently limiting in streams which may warrant nitrogen controls. However, the most effective management strategy may depend on the importance of point vs. non-point sources to streams. As the USEPA (2000c) points out, where nitrogen limitation is created due to discharge of point sources with a low N:P ratio, the best management strategy may be the control of P. Conversely, where non-point sources dominate, reductions of N inputs may be the more appropriate method. The USEPA (2000c) recommends that the "most prudent approach to controlling nutrient enrichment, regardless of the limiting nutrient, is to set criteria for maxima of N and P, and try to limit inputs of both". Similar advocations have been postulated by other researchers (e.g., Dodds et al. 2002).

Additionally, the relative importance of phytoplankton, periphyton, and macrophytes in a stream or stream reach requires consideration in nutrient management. Streams dominated by phytoplankton may experience blooms of nitrogen-fixing cyanobacteria and control of N:P ratios may be a significant management method. Conversely, streams dominated by periphyton may benefit from N and P loading reductions because periphyton are not characterized by heterocystous cyanobacteria (Dodds and Welch 2000). Management with respect to macrophytes is more complex as aquatic plant growth is linked to sediment nutrients, which in turn may or may not respond to loading reductions. Dodds and Welch (2000) indicated that due to the inability or limited ability to link nutrient concentrations to macrophyte biomass, setting nutrient criteria for streams dominated by macrophytes will be difficult. Chambers et al. (1999) similarly speculated that the outcome of management of macrophytes through nutrient controls is
difficult to predict. Furthermore, there is some indication that macrophyte biomass is linked to P in some environments, N and P in others, and primarily N in yet other systems.

In some cases, the various primary producer groups may be limited by different nutrients, complicating the issue of nutrient control management. Ultimately, these factors should be examined on a case-by-case basis and the management objectives will need to be clearly defined in order to focus management efforts. This is particularly important as one strategy may be effective in reducing nuisance growth for one group of producers but not another (e.g., P load reductions in the Bow River were insufficient to reduce macrophyte biomass, but were generally adequate for periphyton, Sosiak 2002).
6.0 MARINE/ESTUARIES

The basic cause of nutrient over enrichment in estuaries and coastal environments is excessive nutrient levels in fresh waters entering the marine environment and direct inputs to coastal waters. The problem is widespread: USEPA (2001) described many, if not most, of the United States coastal waters are moderately if not severely polluted with excess nutrients, especially nitrogen and phosphorus. Likewise, the European Environmental Agency (2001) reported that adverse effects due to nutrient enrichment are widespread in Europe's coastal waters. Problems in northern arctic and sub arctic environments have been less well documented, possibly because major issues have not arisen: USEPA (2001) does not consider the estuarine environments of Alaska and the European Environmental Agency (2001) stated that in Arctic water with very sparsely populated drainage areas, eutrophication is not an issue of concern.

In Manitoba, the coastal waters of Hudson Bay are the final receiving environment for the province's fresh waters. The Nelson River flows from the outlet of Lake Winnipeg through several hundred kilometres of northern Manitoba, passing through several lakes (Playgreen, Cross, Sipiwas, Split and Stephens lakes and smaller reservoirs) and receiving inflow from several major river systems (in particular the diverted Churchill River via the Burntwood River) before reaching Hudson Bay at the Nelson River estuary.

The following sections provide a brief description of the Nelson River estuary and adjacent waters of Hudson Bay followed by an overview of issues and conclusions from studies of eutrophication in north temperate estuarine and coastal marine environments.

6.1 HUDSON BAY AND THE NELSON ESTUARY

The oceanography of Hudson Bay has historically received little attention, presumably due to the absence of resources of commercial interest. ArcticNet is currently conducting a research program in the Bay, but these results are not yet available in published reports. Stewart and Lockhart (2005) provided an overview of the Hudson Bay marine ecosystem, which is the source of the following information.

Hudson Bay receives Arctic marine water from the Foxe Basin and freshwater runoff from a catchment basin that is larger than those of the Mackenzie and St. Lawrence rivers combined. During summer, the surface layer of the bay is diluted by freshwater runoff and warmed by solar radiation, resulting in a strong vertical density gradient that prevents mixing and the transport of nutrients from deeper waters. This stratification is thought to limit biological productivity. In winter, vertical stratification might weaken, permitting some mixing with deeper waters.
The water in Hudson Bay is relatively clear, with Secchi disk measurements of 18.2 m offshore and 11 to 12 m inshore. Nutrient chemistry, biomass and productivity of Hudson Bay are not well documented. Based on limited data, production appears to be low and comparable to other seasonally ice-free Arctic waters. The authors note that there is too little information to accurately determine production, in particular in relation to ice algae and the occurrence of a spring bloom. Production appears to be greater in coastal waters, particularly in embayments and estuaries where there is periodic upwelling of deeper marine nutrient rich water. Incomplete vertical mixing and resultant low regeneration of nutrients, particularly nitrogen, appear to limit primary production in Hudson Bay. Deep-water mixing and freshwater runoff are important sources of nitrate and total nitrogen with atmospheric deposition accounting for about 10%. Nitrate and phosphate levels in surface waters are very low in summer. While river runoff carries large quantities of carbon and nutrients into the marine ecosystem, the river waters are less concentrated in nutrients than Hudson Bay coastal waters. Offshore in Hudson Bay, maximum chlorophyll $a$ occurs below the pycnocline where nutrients are higher.

The Nelson River is the largest river entering Hudson Bay from the west. Its estuary has been the subject of several surveys (see studies in Schneider-Vieira et al. 1993) and studies are currently being conducted by Manitoba Hydro (in relation to potential future hydroelectric development) and ArcticNet. In a review of the estuaries of Hudson Bay, Schneider-Vieira et al. (1993) provided a brief description of the Nelson River estuary, which provides the basis for the following description. The estuary is formed in the broad, funnel-shaped mouth of the Nelson River and is well-mixed except for a deep narrow central channel. On either side there are extensive flats of sand and clay, which are exposed at low tide. The large fresh water input of the Nelson River forms only 3-4% of the water moving inshore and offshore as a result of the tides. Concentrations of nitrogen and phosphorus are relatively low in the estuary. Concentrations of dissolved phosphorus are approximately three times higher in marine than fresh waters, suggesting a marine origin, while levels of nitrogen, though more variable, are also fifty percent higher in marine than in fresh water.

Phytoplankton biomass was highest in the nearshore and lowest further offshore. Most phytoplankton collected in nearshore regions were freshwater species, indicating that they were carried into the estuary on the flow of the Nelson River. During the survey of the estuary, chlorophyll $a$ values of 0.4 to 5.1 mg m$^{-3}$ were recorded.

As with Hudson Bay as a whole, information published to date on the Nelson River estuary is inadequate to describe the seasonal cycle of production, in particular the potential role of ice algae and an early spring bloom of algae.
6.2 EFFECTS OF EUTROPHICATION IN ESTUARINE AND COASTAL ENVIRONMENTS

The topic of eutrophication in coastal and marine environments has been the subject of intense research (e.g., USEPA 2001; Howarth and Marino 2006); research generally focuses on estuaries impacted by human usage and few studies related to eutrophication have been conducted in arctic systems comparable to Hudson Bay and the Nelson Estuary. There is currently too little information on nutrient conditions in the Nelson Estuary and adjacent waters of Hudson Bay to determine their trophic status; information from the 1970s and 1980s suggested low productivity and nutrient limitation. Howarth (2003) (In Howarth and Marino 2006) also noted that human activity has probably increased nitrogen flux to Hudson Bay little, if at all. The following provides a very brief overview of the current understanding of the role of nitrogen and phosphorus in eutrophication in estuarine and coastal environments, with emphasis on northern environments.

Nutrient enrichment in coastal areas has been associated with increased rates of primary productivity, changes in algal and vascular plant biomass, reduced populations of fish and shellfish, reductions in transparency, loses of biodiversity and oxygen depletion in bottom waters (USEPA 2001). The first signs of nutrient enrichment (e.g., an increase in the supply of nutrients, changes in algal species, and reduction in water transparency) are proceeded by the other, more severe symptoms.

The relative role of nitrogen and phosphorus as the causal agents of anthropogenic eutrophication has been the subject of considerable debate. Guildford and Hecky (2000) compared measured values of TN, TP and chlorophyll a across a broad range of lake and ocean sites. The authors concluded that there was a common relationship between the algae of fresh and marine systems in that the N or P limitation of growth was dependent on the TN and TP concentrations and the TN:TP ratio in a given environment, whether freshwater or marine.

Smith (2006), in a comparison of 92 coastal zone ecosystems, found that average concentrations of chlorophyll a were strongly dependent on the mean concentrations of TN and TP in the water column. The identity of the primary growth limiting nutrient appeared to be generally predictable from water column TN:TP ratios. Smith did note however, that these analyses are based on mean values of TN and TP; many coastal systems exhibit a strong seasonality in nutrient limitation. However, the transition between N and P limitation appears to be much more abrupt than in freshwater systems (in fresh waters there is a range of concentrations that are described as "co-limited"). The difference between freshwater and marine systems was attributed to the algal communities. In freshwater systems, N deficiency frequently selects for phytoplankton dominance by heterocystous cyanobacteria, which can fix atmospheric N, thereby
increasing N levels in the water column. In marine ecosystems, dominance by cyanobacteria in the water column is more rare, planktonic nitrogen fixation is less commonly observed and fixation rates are much lower than in freshwater systems with comparable fertility. Finally many shallow coastal ecosystems are likely to have strong potential for the loss of fixed nitrogen through denitrification at the sediments.

Some brackish marine environments, such as the Baltic Sea, have large cyanobacterial blooms during the summer months; recent studies analyzing bottom sediments indicate that nitrogen fixing cyanobacterial blooms are nearly as old as the present brackish water phase of the Baltic Sea (Bianchi et al. 2000 In Laanemets et al. 2006). Nutrient enrichment has exacerbated these blooms. In general, a low N:P ratio of dissolved inorganic nutrients is a major pre-requisite for bloom formation, but other factors, such as temperature, wind, light conditions, and the mixed depth, have to be suitable. During the period of May to September, Nausch et al. (2004) found that algae in the central Baltic Sea shifted from a condition of nitrogen-limitation (following the spring bloom) to phosphorus limitation concurrent with the development of a nitrogen-fixing cyanobacterial bloom.

Howarth and Marino (2006) reviewed the evolving understanding of nutrient limitation in estuarine environments and noted that by the 1990s the general consensus was that nitrogen was the culprit behind most, if not all, eutrophication problems in coastal estuaries with salinities greater than 10 – 12. The authors noted that the N fixation response generally does not occur in more saline waters, even when they are strongly N limited. The authors also noted that other factors increase the prevalence of N limitation in estuarine and coastal marine environments. Estuaries receive water, not only from freshwater rivers, but from offshore marine sources. Denitrification that occurs on coastal shelves lowers the N:P ratio in marine waters. Finally, P is more available in saline than fresh waters, because it is less bound to the sediments and to suspended particulate matter. However, the authors noted that the situation is complex and that there is evidence that the potential for P limitation increases with increased nutrient loading. The mechanisms limiting cyanobacterial nitrogen fixation are complex, and appear to be related to a combination of low growth (possibly linked to high sulphate concentration) and grazing by planktonic and benthic organisms. Notable exceptions to the general N limitation occur in brackish waters (e.g., the Baltic Sea) and shallow systems with abundant vegetation (e.g., coastal seagrasses) where cyanobacteria can form abundant, epiphytic growth.

The effect of nutrient limitation on ice algae and bacteria in the Baltic Sea was studied in winter 2002 by Kuosa and Kaartokallio (2006) through the use of in situ experimental additions of nutrients. Snow cover had a large influence on production of the algal community, which consisted of cold-adapted diatoms, dinoflagellates and other taxa. Growth of these algae
appeared to be primarily light-limited; nutrient additions had variable effects on algal growth which the authors related to the physiological condition of the algae.

6.3 NUTRIENT CRITERIA

The USEPA (2001) outlines the process of developing site-specific criteria for estuarine and coastal environments. It was considered impossible to derive a single criterion for all estuaries or even for classes of estuaries, as these environments are highly unique. Steps in the development of site-specific criteria are: (1) determine historical information describing trends so that pre-impact nutrient concentrations can be estimated; (2) identify suitable reference areas; (3) develop an understanding of nutrient concentrations through the use of hydrologic and loading models; (4) convene a panel of experts to interpret information in steps 1-3 to develop nutrient criteria; and (5) consider whether these criteria will protect adjacent coastal waters.

Smith (2003), in a review of eutrophication in freshwater and coastal marine ecosystems, concluded that although estuarine and coastal marine systems can be hydrologically and spatially complex, phytoplankton biomass in these systems tends to respond to changes in the water column concentrations of N and P, indicating the utility of controls on nutrient inputs. Smith (2003) cited numerous examples of models that have been developed relating nutrient concentrations and other parameters to indicators of eutrophication such as chlorophyll a concentration. In addition, studies have shown that improvements in coastal zone water quality may follow reductions in external nutrient loading.

Howarth and Marino (2006) noted that the relationship between nutrients and phytoplankton in estuarine environments is very complex due to the interaction of nutrient limitation, light limitation, water residence times and other factors. The authors noted that, although N appeared to be the primary nutrient associated with eutrophication in many systems, other estuaries are P limited or there is a seasonal switch from N to P limitation. Finally, an overabundance of P in incoming freshwaters can lead to other undesirable effects, such as a reduction in silica and subsequent reduction in diatoms, which may be an important food group for estuarine fauna. Therefore, they recommended that both N and P be targeted in nutrient management programs.

Although, as noted above, it is difficult to establish universal criteria for the protection of estuarine environments, some criteria have been developed. The Swedish eutrophication classification system as presented by Smith (2003) is provided in Table 33.
Table 33. The Swedish eutrophication classification system (Smith 2003).

<table>
<thead>
<tr>
<th>Class</th>
<th>Nutrient pollution level designation</th>
<th>Summer TN (µg/L)</th>
<th>Summer TP (µg/L)</th>
<th>August Chlorophyll a (µg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Very low</td>
<td>&lt; 252</td>
<td>&lt; 15</td>
<td>&lt;= 1.5</td>
</tr>
<tr>
<td></td>
<td>252 -</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Low</td>
<td>308</td>
<td>15 - 19</td>
<td>1.5 - 2.2</td>
</tr>
<tr>
<td>3</td>
<td>Moderate</td>
<td>364</td>
<td>19 - 24</td>
<td>2.2 - 3.2</td>
</tr>
<tr>
<td></td>
<td>364 -</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>High</td>
<td>448</td>
<td>24 - 31</td>
<td>3.2 - 5.0</td>
</tr>
<tr>
<td>5</td>
<td>Very high</td>
<td>&gt; 448</td>
<td>&gt; 31</td>
<td>&gt; 5.0</td>
</tr>
</tbody>
</table>
7.0 TOWARDS DEVELOPMENT OF NUTRIENT CRITERIA

The preceding sections of this report have provided an overview of the current state of scientific knowledge of Lake Winnipeg as it pertains to nutrient enrichment, followed by an overview of the scientific literature pertaining to effects of nutrient enrichment and its management in lacustrine, stream and, very briefly, coastal marine environments. The intent of this review was to provide information that would assist Manitoba Water Stewardship in the development of nutrient criteria for Lake Winnipeg, its watershed, and the downstream environment. Comparison of the information presented in Section 3.0 Lake Winnipeg, with the analyses that have been conducted for lakes where nutrient enrichment is a concern as described in Section 4.0, clearly illustrates that the current understanding of nutrient enrichment in Lake Winnipeg is in an early stage. Much information has been gathered, but considerable analysis of existing data, coupled with the collection of additional field data to address key information gaps is required before a scientifically defensible nutrient management strategy for the lake can be implemented. Similarly, analysis of data already in hand for the upstream watershed in Manitoba, coupled with collection of additional site-specific field data will be required prior to the identification of the appropriate management strategy. Information presented in Section 4 will assist in determining the kinds of analyses that need to be conducted for Lake Winnipeg, but the current understanding of Lake Winnipeg is too limited to identify which of the many lakes described in Section 4 provide a good model for Lake Winnipeg. Information in Section 5 will provide guidance in the types of analyses required in support of the development of nutrient criteria for the waters upstream and downstream of Lake Winnipeg.

The following section provides:

- a synopsis of key considerations regarding the temporal and spatial variability on Lake Winnipeg (Section 7.1);

- an overview of steps that could be implemented to synthesize and analyse data currently in-hand (Section 7.2);

- a description of information gaps that need to be addressed during the process of developing a nutrient management strategy for Lake Winnipeg (Section 7.3); and

- next steps in the development of nutrient criteria for Lake Winnipeg and its watershed within Manitoba.
7.1 A QUESTION OF SCALE: TEMPORAL AND SPATIAL CONSIDERATIONS

Lake Winnipeg is a highly complex environment, variable in time and space. The range of conditions that may occur in the lake must be considered when defining the existing condition of the lake, as well describing the past, ‘pristine’ state that provides a benchmark for assessing anthropogenic impacts and predicting the future condition.

7.1.1 Temporal Variation

At the largest temporal scale, Lake Winnipeg is a remnant of glacial Lake Agassiz and the shorelines and basin are still evolving as a result of isostatic uplift and the normal progression of a glacial lake (Baird and Stantec 2000). Analysis of algal remains in sediment cores indicates that the lake for the past millennium has existed in different states, marked by changes in the phytoplankton community, with periods of dominance by cyanobacteria alternating with periods dominated by diatoms and other groups (Kling 1998). These oscillations have generally been attributed to warm dry periods alternating with cold wet periods (Kling 1998). Sediment core data also reveal periodic changes in nutrient or other conditions in the lake (Mayer et al. 2006).

At the decadal scale, the lake experiences profound changes in the volume of inflow (Baird and Stantec 2000), which affects lake levels, outflows and nutrient loading from the watershed. Other changes at the decadal scale (e.g., air temperature, wind) may also occur but have not been examined. It is expected that these changes would also alter the physical, chemical, and biological characteristics of the lake, although this has not been explicitly assessed.

Seasonally, there are pronounced changes in the physical, chemical, and biological characteristics. Some are obvious, such as the distinction between ice-cover and open-water seasons but others are less apparent, for example, available nutrient data suggest that levels change over the course of the open-water season, tending to increase in fall compared to the summer months. Algal biomass also follows clearly defined seasonal patterns though the exact timing may vary between years. At the time scale of days to weeks, events may occur that are significant for the lake. For example, although the lake is generally not stratified, stratification may intermittently occur and when it does the resulting hypoxia exerts effects on the benthos that last over a year (Hann 2005). Algal blooms are also transient, developing and dissipating rapidly depending on conditions of temperature, sunlight and wind, when other factors, such as nutrient levels, are appropriate. Finally, within the day, conditions in the lake will change. The most pronounced changes are likely associated with photosynthesis which is closely related to the diurnal cycle of light.

Super-imposed on these “natural” variations, there are long-term changes that may or may not be related to anthropogenic effects. The arrival in Lake Winnipeg of new species of phytoplankton
(Kling 2006), zooplankton (Salki 2005), and fish (Franzin et al. 2003) have been documented; the impact of these new species on the ecosystem is unknown. The hydrologic regime of the lake has been markedly altered by water regulation, beginning in the watershed in the late 1800s and extending to Lake Winnipeg itself in 1976 (Baird and Stantec 2000). Additionally, flows in major tributaries such as the Saskatchewan and Red rivers have been decreasing and increasing, respectively. Water temperature appears to have increased slightly over the past 100 years (McCullough 2004), and temperature may continue to change.

Consideration of the above highlights several important issues:

- Sampling programs should consider the range of variability exhibited by the parameter under investigation. For example, nutrient and algal levels during the growing season in a lake are best characterized based on frequent (biweekly or more often) sampling. Likewise, the frequency of occurrence of stratification and potential oxygen depletion at depth can only be ascertained through frequent sampling, as this is a transient phenomenon. As most parameters of interest in eutrophication studies exhibit strong seasonality, sampling should include the entire open-water season, and, ideally, some representative part of the winter.

- Interpretation of data and comparison to other periods must consider whether data collection was comparable in terms of the variability discussed above and, if not, conclusions must be adjusted accordingly. When comparisons are being made over several years, the influence of variables such as inflow should be considered. For example, the most recent studies of Lake Winnipeg have occurred during a period of relatively high inflows and lake levels (within the exception of the drought of 2003). Much of the previous two decades (since Lake Winnipeg Regulation) were relatively dry, while the period prior to LWR was very wet. The 1969 limnological survey occurred during a period of record high water levels on Lake Winnipeg.

- It is very difficult to define a “pristine” state for Lake Winnipeg since even the natural condition was quite variable (e.g., phytoplankton community, Kling 1998).

- Separating the effects of natural temporal variation from anthropogenic effects is an extremely difficult task. As discussed in Section 7.3, tools such as models and experimental approaches may help.
7.1.2 Spatial Variation

Divided into two distinct basins, with markedly different climate, inflows, morphological characteristics, chemical characteristics and biota, the north and south basins of Lake Winnipeg should be considered as two separate lakes. The major rivers entering these basins drain watersheds that arise in different regions, to the extent that the occurrence of drought in one watershed may be associated with normal flows in the other. Both natural and anthropogenic influences in the watersheds are also markedly different, making generalization difficult. The basins themselves also do not represent homogenous environments, in part because the lake lies along the divide between Precambrian and sedimentary geologies, but also because the inflowing rivers are markedly different in their characteristics and associated biota. This spatial heterogeneity is clearly demonstrated in studies of water quality and biota (e.g., Franzin and Watkinson no date, Salki 2005).

Spatial variation, like temporal variation, must be considered in both the collection and interpretation of data, in particular where long term data sets are being compared.

7.2 LAKE WINNIPEG – SYNTHESIS AND ANALYSIS OF EXISTING DATA

Based on information in Section 3.0, the most striking observation is the apparent resilience shown by Lake Winnipeg to the environmental perturbations that it has experienced over the last century and particularly in the last 50 years. Based on available data, the majority of species identified in the lake in the 1920s are still present, no dramatic increase in nutrient concentrations or chlorophyll \( a \) biomass is evident for the past 40 years (based on available data), and the lake maintains a large fishery. In part, this resilience may be because the lake appears to always have been somewhat nutrient rich, with a low retention time so water is rapidly renewed. There is also a lack of widespread permanent stratification, which reduces the potential for widespread, frequent anoxia at depth.

An important caveat to this observation is that changes have likely occurred, but have not been clearly demonstrated in the available scientific data. More dramatic changes may have become manifest more recently. For example, satellite imagery suggests an increasing frequency of cyanobacterial blooms from the mid 1990s to 2005 compared to the mid 1980s to mid 1990s; this could represent a long-term trend or reflect differences in the hydrological (or other) conditions during these two periods.

The key question at the moment is whether the changes seen in the last decade are within the range experienced by the lake over the last millennia or whether the combination of environmental stressors, potentially including nutrient loading, hydrologic alterations in the
watershed (both upstream and at the lake), changes in shorelines, introduction of exotic species and climate change (either cyclical or long term) have brought it to a critical threshold.

An immediate requirement is the synthesis of existing recent and historic nutrient (and supporting) data into a comprehensive database associated with clear information regarding sample collection methodology, timing and location to permit characterization of existing conditions in the lake and comparison to past conditions, to the extent feasible.

The synthesis of existing data pertaining to the algal community into a database, and analysis of the species composition and biomass in recently collected samples would provide a valuable tool to gain a better representation of the entire algal community. Current measurement methods are biased to species with chlorophyll $a$ as their primary photosynthetic pigment (i.e., diatoms are likely underrepresented) and to species that form surface blooms (i.e., certain species of cyanobacteria).

There is currently an abundance of data regarding the biota of the lake that have been collected for other purposes (e.g., assessment of climate change) that could provide useful information. For example, the zooplankton data provide a long-term record of conditions in the lake based on comparable sampling methods. Results need to be considered in light of the varying physical conditions under which they have been collected and then analyzed to determine whether there is correlation with environmental variables of interest. However, a holistic approach needs to be taken, so that unexpected linkages could be identified. The existing fish sampling program could also be adjusted so that it could address issues potentially pertinent to eutrophication (see Section 7.3).

A holistic approach in the analysis of existing data is advocated. Changes in one component of the ecosystem as a result of a particular stressor would be expected to be accompanied by related changes in other parts of the ecosystem.

### 7.3 INFORMATION GAPS

The following information gaps pertaining to the development of nutrient criteria for Lake Winnipeg have been identified based on the review of Lake Winnipeg literature summarized in Section 3.0 and the overview of nutrients in freshwater aquatic ecosystems presented in Sections 4.0 and 5.0. It is cautioned that this gap analysis may include issues that are either currently being addressed or have been proposed to be addressed in the future. The following provides a brief summary of primary information gaps identified through this exercise.
1. Water Balance

Development of a water balance for Lake Winnipeg is critical for the evaluation of eutrophication and lake trophic status for several reasons:

- Water balances serve as the foundation for nutrient balances (in several ways, including consideration of issues such as retention times);
- A full accounting for inflows, outflows, and changes to lake storage is needed to develop an accounting of all sources and sinks of nutrients;
- The role of groundwater is unknown in terms of its role hydrologically as well as a potential source of nutrients. This may be most critical for littoral areas;
- A water balance would provide the backbone for development of water quality models, and hence, is critical to development of predictive tools for future or hindcasting of lake conditions; and
- A water balance is needed to evaluate how temporal changes in hydrology, as they may relate to Lake Winnipeg Regulation or due to changes in the overall climatological and hydrological cycles, affect eutrophication of Lake Winnipeg.\(^{10}\)

2. Nutrient Balance

Similarly, it is believed that a thorough accounting of internal and external sources and sinks of nutrients should be derived and a nutrient balance be constructed. This is typically the first step in a lake eutrophication study and the development of nutrient criteria. The particular value beyond the obvious (i.e., the quantification of sources of nutrients) is in the ability to compare all of the relative sources and sinks, including internal cycling. This is especially important from a management perspective as this information is critical for identifying potential mitigation and management options. It is further suggested that the following be considered:

- Estimation of the entire nitrogen cycle, including quantification of nitrification as well as denitrification. Recent efforts have generated estimates of N-fixation but not losses due to denitrification, and the significance of nitrogen fixation to the overall nitrogen budget in Lake Winnipeg is unknown. As described in Section 4.0, the

\(^{10}\) The LWIC (2005) states: “that scientist attribute the…eutrophication of the North Basin of Lake Winnipeg to…regulation of the lake as a hydroelectric power generation reservoir leading to the hold-back of water during the productive spring and summer season”.

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overall significance of N fixation can vary substantially between lakes. In some instances large masses of fixed N can be negated by denitrification;

- Derivation of estimates of internal loading, including fluxes of nitrogen and phosphorus from sediments. As described in Section 4.0, internal loading can be a highly significant source of nutrients in lakes, particularly shallow lakes, and should be evaluated in order to develop a nutrient management strategy. If internal loading is a significant factor for Lake Winnipeg, lake recovery may be delayed even following large external nutrient load reductions (see Section 4.5 for discussion). Without prior knowledge of internal loading in the lake, it would be difficult to track the effectiveness of nutrient management strategies (e.g., internal loading may continue to supply adequate nutrients to surface waters long after external sources are reduced, and in some instances internal loading may actually increase following external load reductions);

- Internal loading estimation should be conducted for a nutrient budget at an annual scale (i.e., estimate the annual flux of N and P from sediments to the lake water) as well as on a finer time scale (i.e., seasonally) to evaluate how seasonal changes in internal cycling may affect nutrient availability to primary producers. The available nutrient data for Lake Winnipeg presented in Section 3.0 indicate that TP increases over the open-water season which may reflect the occurrence of internal loading. Seasonality of internal loading can be very significant in terms of contributing to conditions conducive to cyanobacterial blooms. However, without a formal nutrient balance (and accompanying water balance) the potential explanation for this observation is not known;

- In order to estimate internal loading, additional information on vertical differences in nutrient concentrations and related variables must be collected. That is, depth profiles for \textit{in situ} variables should be collected and assessed to ascertain the occurrence of stratification and/or DO depletion near the sediment-water interface. Furthermore, samples should be collected for analysis of nutrients across depth to facilitate calculation of internal loads and to provide some empirical study of the occurrence of internal loading (i.e., are concentrations of P higher at depth, do they increase over the open-water season); and

- Consideration of the role of erosion and resuspension as a source (or sink) of nutrients in Lake Winnipeg, most notably the littoral zone. Resuspension can be a significant source of nutrients in lakes, although in some cases it has been found to reduce nutrients in the water column and/or the bioavailability of nutrients to primary producers.
3. Factors Limiting Phytoplankton Growth

As described in Section 3.0, there has been some study of the role of nutrients and light in limiting phytoplankton growth in Lake Winnipeg. However, it would be beneficial to evaluate nutrient limitation in phytoplankton over the open-water season in conjunction with measurements of light limitation, concentrations of nutrients, phytoplankton species composition and biomass, and nitrogen fixation. This would collectively provide a more comprehensive assessment of the complexity of factors that affect phytoplankton growth and the occurrence of N-fixing cyanobacteria.

Due to the occurrence of erosion/resuspension in Lake Winnipeg, the heterogeneity of turbidity and light, the complex interactions between resuspension, nutrients, and phytoplankton growth, and the suggestion that eutrophication of Lake Winnipeg is, in part, attributed to the “damming of the Saskatchewan River upstream of the north Basin, causing its sediment load to be deposited before reaching Lake Winnipeg, resulting in more transparent water and increased light for algal growth” (LWIC 2005), a targeted study focused on this issue would be valuable. This could be accomplished through studies along light gradients in the lake and/or through a mesocosm approach.

4. Shoreline Processes and the Littoral Zone

The majority of ecosystem-related research and monitoring on Lake Winnipeg has focussed on offshore areas, as most work was conducted using larger vessels as research platforms. Evaluation of nutrients and algae in the nearshore environment, including describing the presence and abundance of macrophytes, would be beneficial to evaluating the overall condition of Lake Winnipeg. Similarly, consideration of conditions (and temporal changes) in Netley Marsh would assist in defining the current status of this ecosystem.

5. Trophic Cascades and Food Web Effects

As described in Section 4.0, top-down effects and shifts in food web structures can be good indicators of eutrophication and have been found to be highly significant in controlling phytoplankton growth in many lakes. For example, shifts in zooplankton communities may indicate trophic cascades (e.g., decrease in large cladocerans often occurs with eutrophication leading to reduced grazing pressure and subsequent increases in phytoplankton). Furthermore, the structure of the fish community may be particularly significant in top-down control of primary productivity (and shifts between alternative stable states), as has been clearly demonstrated through numerous mesocosm and whole-lake studies. The large number of biomanipulation studies provide a good indication of the significance of top-down
controls in this regard (see Section 4.7 for discussion). The appearance and increasing dominance of rainbow smelt in Lake Winnipeg requires further evaluation in this regard. In particular there is a need to identify the trophic position of this species in the Lake Winnipeg food web and to ascertain what effect its presence may have on top-down controls (i.e., does rainbow smelt affect zooplankton and grazing of phytoplankton).

7.4 NUTRIENT CRITERIA - NEXT STEPS

Addressing the information gaps described in Section 7.3 are important in the development of nutrient targets and management options. However, the identification of nutrient criteria also requires the identification of goals, which are in turn, linked to site-specific considerations such as natural ‘background’ conditions for a lake. Site-specificity is important because lakes naturally vary in trophic status. Furthermore, as the south and north basins of Lake Winnipeg differ considerably in many respects (e.g., water residence time, depth, morphology), consideration of the basins separately may be warranted. As outlined in Section 4.6.2, nutrient criteria are often defined on the basis of a “reference condition”. However, there are several options for identifying a reference condition for a given waterbody including:

- development of reference conditions using ‘pristine’ (ideally) or ‘less affected’ comparable waterbodies (often derived on an ecoregional basis);
- use of historical nutrient data for the given lake (i.e., if historical data exist);
- development and use of water quality models to hind-cast nutrient concentrations;
- use of paleolimnological models to reconstruct past nutrient conditions; and/or
- use of site-specific studies of nutrient-algal relationships (i.e., identification of critical thresholds).

Use of a paleolimnological model is an attractive option for reconstructing the nutrient history of Lake Winnipeg given that the lake is quite unique and definition of reference conditions using other lakes does not appear feasible. Development of a water quality model would also be valuable in this regard but may not be feasible given the size and complexity of the system in question. It is also suggested that regardless of what method is used to identify past nutrient conditions, other factors that affect nutrient-phytoplankton relationships in the lake should be considered when developing criteria. This is particularly important should it be determined that temporal changes for such factors have occurred over time (e.g., if turbidity or temperature has changed).
Additionally, consideration should also be granted to the significance of proposed criteria on the downstream environment; as indicated by the USEPA (2000a), criteria should ensure protection of the immediate (several kilometres) downstream environment. That stream and marine ecosystems are reported to be more frequently nitrogen-limited than lakes (See Sections 5.0 and 6.0), should be considered in the development of regional nutrient criteria. The relative importance of nitrogen and phosphorus in streams may vary between systems and should be considered in terms of the most significant primary producers and issues within each waterbody (i.e., nuisance periphyton or macrophyte growth may require different nutrient management strategies). As described in Section 5.0, the significance of nitrogen may vary between streams dominated by phytoplankton, periphyton, and macrophytes.
8.0 LITERATURE CITED


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