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**Evaluating the Lake Management Approach, Applied
Biomanipulation Techniques and Progress in Restoring
Ecological Function of Littoral Macrophytes in
Grenadier Pond, Canada**

by

Christine Laura Tu

**A thesis submitted in conformity with the requirements for the degree of Master of Science
Graduate Department of Zoology
University of Toronto**

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M. Sc. Thesis (2000)

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Abstract

Grenadier Pond, Canada, is a shallow, eutrophied, urban pond system. Pond managers employed biomanipulation techniques in an attempt to alleviate symptoms of eutrophication. Alternate stable state theory may provide an additional or alternate approach to nutrient reduction/abatement strategies. Biomanipulation techniques apply alternate stable state theory through: macrophyte-phytoplankton competition, piscivore/grazer predation pressures and mitigation of physical disturbances. Long-term success of biomanipulation in shallow systems has been linked to intermediate nutrient levels and well-established and relatively abundant macrophyte communities. An evaluation of the aquatic macrophyte regeneration project in Grenadier Pond did not yield the expected ecological functional relationships between sediment characteristics (total phosphorus, organic content, water content) and rooted-macrophyte density. Wave-wind action may be a main mechanism of nutrient redistribution in the littoral zone in Grenadier Pond. Newly planted macrophytes seem to provide some macrobenthic invertebrate habitat. Sediment chloride may be influencing macrophyte diversity in Grenadier Pond.

This thesis is dedicated to the memory of my father,

Thomas Chuan Tu, Ph.D.

To the future of my son,

Thomas Andrew Gyongyossy

And to the love that is my husband,

Edward Andrew Gyongyossy, MD.

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Chapter 1 : The Management of Shallow Urban Ponds: Grenadier Pond, A Case Study

1.0 Introduction

Hindsight is supposed to be 20/20, or something close to it. Yet this “post-mortem” approach is only as good as the tools with which you measure an outcome. The types of tools available are often specific to the subject matter being evaluated. The subject matter of this project is the ecological management and rehabilitation of Grenadier Pond, a shallow, urban pond impacted by cultural eutrophication. The “tools” are assessment of decisions made and work done in attempts to “rehabilitate” the pond ecosystem.

My interests are whether management decisions, precursors to management actions, have been sufficiently comprehensive to be appropriate for Grenadier Pond given their stated objectives. My second interest is whether the management objectives fulfill an ecological objective: functional recovery. I evaluate management decisions and overall approach through: 1) a review of the limnological knowledge of the Pond (historical and present) to evaluate the degree to which its biological conditions are adequately represented in the management plan and 2) compare the plan with other similar case studies. I am particularly interested in the management approach of biomanipulation, specifically littoral macrophyte plantings. Determination of the extent to which macrophyte function has returned to Grenadier Pond is accomplished through: 1) direct field measurements and statistical analyses and 2) an extensive literature review.

One might ask: “Why is it important to evaluate lake management decisions?” Ideally, lake management is comprehensive, drawing on the collective wisdom and experience of scientists, engineers, as well as other members of the environmental community. However, realistically, options arising from such a consortium are often expensive. If initial funding is received, additional monies almost always require more elaborate rationale, with no guarantee of a successful bid. Rehabilitation efforts that must contend with erratic fits and starts stand less of a chance in performing as intended (Moss, 1999; Carpenter and Kitchell, 1992). So, when it comes down to vying for more project funding, assessment and refinement of previous work/decisions can be pivotal to a convincing argument. Such a step provides a “second” chance to re-evaluate the benefits or appropriateness of a particular remedial action against the costs within the context of a specific site. The results of such an endeavour can be the leverage lake managers need to obtain necessary funding or at the very least arrest an ill-advised project. Grenadier Pond is a prime candidate for this assessment/refinement step, as present resources are too limited and the costs too high for the full implementation of long-term management to proceed without further evidence of progress. Ultimately, one would hope that the whole project can become an important case study in aquatic system management.

Assessment serves another purpose. The opportunity to study all the aspects and impacts of environmental change is a luxury often only afforded limnological academics. Lake managers benefit when academics apply this knowledge to refining predictive models or furthering our understanding of ecological theory. According to Hobbes and Norton (1996), rehabilitation/restoration ecology “has largely progressed on an ad hoc, site-and-situation specific basis with little development of general theory or principle”. While development of a body of knowledge is not easily or quickly achieved, a methodical, properly controlled assessment can enlighten charges of cause and effect, a necessary prerequisite for theory development.

In keeping with the above ideas, the goals of this research project are summarized below and presented in three chapters:

1. To evaluate the extent to which the Grenadier Pond management plan has considered and capitalized on the unique characteristics of the Pond’s ecosystem, including citizen involvement (Chapter 1);
2. To compare the overall management approach developed for Grenadier Pond with other studies of degraded or altered aquatic ecosystems (Chapter 1);
3. To acquire further understanding of biomanipulation and how littoral zone dynamics may interrupt or interact with the process of cultural eutrophication in a shallow urban pond (Chapter 2);
4. To evaluate a prescribed and common lake rehabilitation strategy (i.e. biomanipulation); the re-establishment of littoral macrophytes in Grenadier Pond (Chapter 3).

1.1 Grenadier Pond: Natural History and Recent Limnological Changes

1.1.1 General Description: Grenadier Pond, situated within the western boundaries of High Park, Toronto (43° 38’ latitude, 79° 28’ longitude), is a naturally occurring, shallow pond that has become highly eutrophic over the last century. Dense, seasonal algal blooms of blue-green algae have become a regular feature of the pond, causing consistently low summer water clarity with average Secchi depths varying from 1.2 m in 1985 (Zimmerman *et al.*, 1986) to 0.55 m in 1993/94 (Gartner Lee Ltd (GLL), 1995) to 1.0 m in 1996/97 (Olding, 1998). The average water depth is 2.9 m, with a maximum depth of 6.5 m located in the central basin (GLL, 1995). The surface area of the pond is 18.6 ha (Zimmerman *et al.*, 1986). The pond’s largely urbanized watershed is approximately 245 ha, although historically the catchment area was closer to 477 ha (GLL, 1995).

Grenadier Pond and its watershed lie on the sand and silty-sand deposits of glacial Lake Iroquois (GLL, 1995). The generally high permeability of the underlying catchment suggests historically high infiltration capacity of groundwater that is not likely to have been affected significantly by urbanization (GLL, 1995). In terms of surface water contributions, Wendigo Creek is the main feeder stream to the pond and is mostly buried under city streets, transporting storm water directly to the north end of the pond (see Fig. 1.1). Four other storm water outfalls are located along the western shore, while the eastern shore drains about 32.8 ha of parkland (GLL, 1995). Drainage from Catfish Pond to the west also enters Grenadier Pond just above the only pond outflow, located at the southwest corner. The pond outflow is now controlled by a weir and piped to the Humber River, where it eventually flows into Lake Ontario.

Historically, Grenadier Pond mixed directly with Lake Ontario when seasonally high lake water levels washed over a sand bar separating the two systems at the south end of Grenadier Pond (Wainio *et al.*, 1976). The construction of the railway and then Lakeshore Boulevard during the late 1800's required extensive landfilling over the general location of the sand bar, resulting in a permanent barrier to seasonal water exchange (Wainio *et al.*, 1976). The pond was further distanced from Lake Ontario influence when, in 1955 and 1956, 4.4 ha of the southern part of the pond were filled to accommodate the construction of the Queen Street West extension (Zimmerman *et al.*, 1986).

1.1.2 Macrophytes: Historical mapping (c. 1870s) of the pond suggests a near continuous ring of aquatic vegetation and wetlands characterized the pre-urban habitat of Grenadier Pond (Archives of the City of Toronto, Department of Parks and Recreation). The pond area lost to landfilling in the 1950's likely contained perimeter vegetation/marshland that has been progressively lost since (GLL, 1995; Zimmerman *et al.*, 1986). Zimmerman *et al.*, (1986) noted vegetation changes on aerial photos dating from 1947 to 1972 (Fig. 1.2). More recent aerial photos and field reconnaissance show only two remnant wetland communities, one in the south-west corner and the other at the very north end of the pond (GLL, 1995). Aerial photos have also documented the growth of a sediment plume directly south of the north-end wetland that may explain the perched condition of the wetland above the pond.

In addition to changes in wetland area, approximately 45% of the shoreline (southern and eastern shores) (Zimmerman *et al.*, 1986) was cleared of vegetation in the 1950's and replaced with a hard concrete edge (Wainio *et al.*, 1976). No significant littoral macrophytes naturally regenerated in these areas. At present, the diversity and abundance of littoral macrophytes (emergent, submerged and floating) in Grenadier Pond is low, with established littoral plant colonies occurring only along the western and upper-eastern shoreline (Fig. 1.2). The past removal of macrophytes, aggressive cultural eutrophication and associated changes in both sediment and water quality are likely contributors to the present paucity of aquatic plants. Table 1.1 summarizes the changes in littoral and wetland plant species from 1973 to 1997 in Grenadier Pond.

Figure 1.1

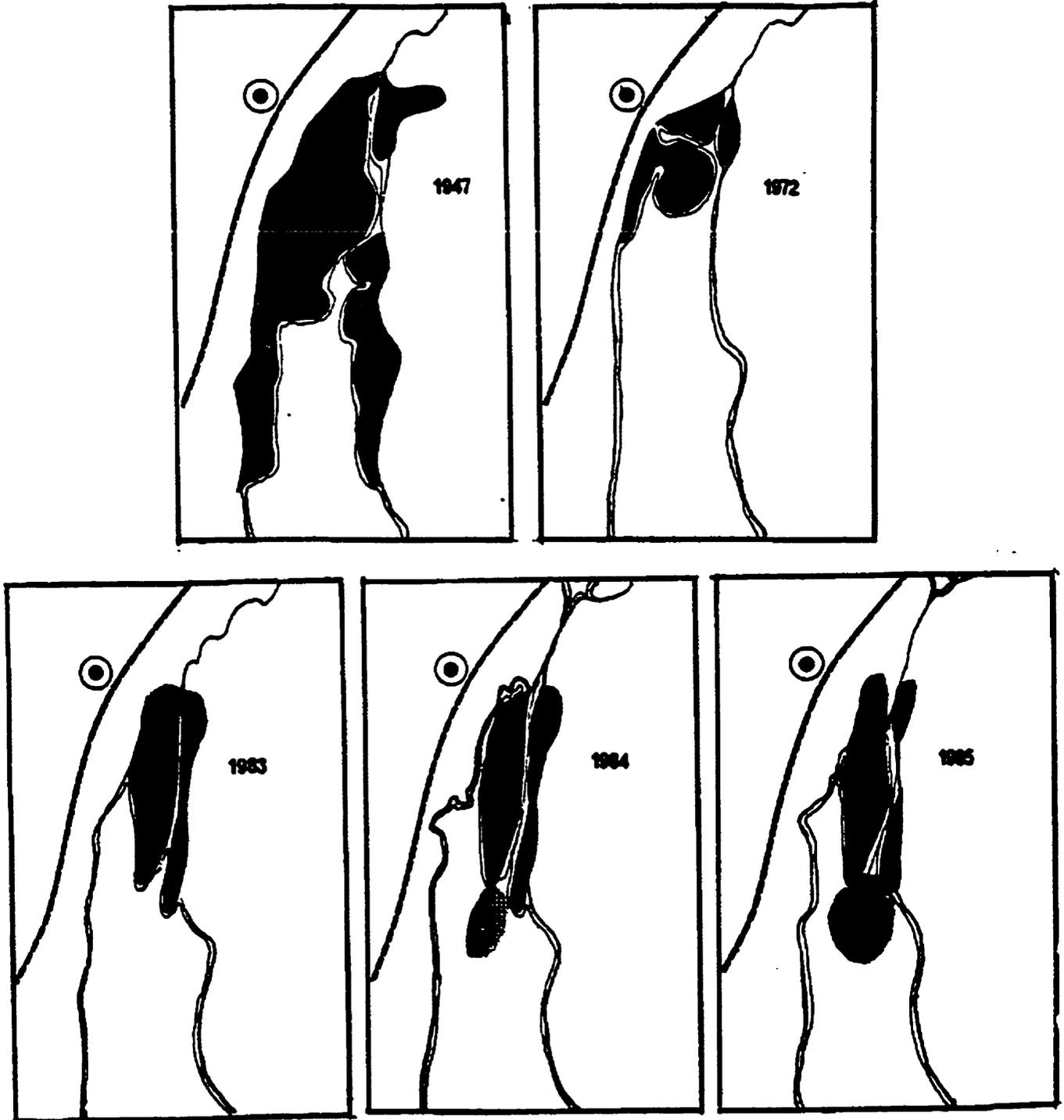


Figure 1.2 Changes in Marsh area in Grenadier pond's north end from 1947 to 1985.
 The light hatched areas shown on the 1984-85 maps represent a subsurface sediment wedge.
 Figure reprinted with permission from Zimmerman et al., 1986

Table 1.1 Past and Present Aquatic Macrophyte Inventories on Grenadier Pond
Excluding Regeneration and Non-Vegetated Sites

| Category | Species | Common Name | Past Studies | | | | | | | This Study: | | |
|------------------------|--------------------------------|--------------------------------|--------------------|-------------|-----------------------|---------------|------------|------------|------------|-------------|-----------|-----------|
| | | | Wainio et al, 1976 | Varga, 1989 | Warren and Neil, 1989 | Kamstra, 1993 | TRCA, 1995 | TRCA, 1997 | TRCA, 1998 | EV1, 1998 | EV2, 1998 | EV3, 1998 |
| Deep Emergent Marsh | <i>Typha latifolia</i> | Broad-leaved Cattail or common | | x | NP | x | x | x | x | x | NF | x |
| | <i>Scirpus validus</i> | Soft-stem Bulrush | x | NF | NP | NF | x | x | x | NF | NF | NF |
| | <i>Acorus calamus</i> | Sweet Flag | x | x | NP | x | NF | x | x | NF | x | x |
| | <i>Iris pseudacorus</i> | Yellow Iris or Yellow Flag | x | x | NP | x | NF | NF | NF | NF | NF | NF |
| | <i>Iris versicolor</i> | Blue Iris or Blue Flag | | x | NP | NF | x | x | x | NF | NF | NF |
| Shallow Emergent Marsh | <i>Lythrum salicaria</i> | Purple Loosestrife | | x | NP | x | x | x | x | NP | NP | NP |
| | <i>Lycopus europaeus</i> | European Waterhorehound | | x | NP | x | NF | NF | NF | NP | NP | NP |
| | <i>Lycopus americanus</i> | Cut-leaved Waterhorehound | | x | NP | | NF | NF | NF | NP | NP | NP |
| | <i>Polygonum lapathifolium</i> | Nodding or Pale Smartweed | | NF | NP | x | NF | NF | NF | NP | NP | NP |
| Floating | <i>Nymphaea odorata</i> | Fragrant waterlily or white | | x | NP | x | x | x | x | x | x | x |
| | <i>Nuphar variegatum</i> | Yellow waterlily | | x | NP | x | x | x | x | x | NF | NF |
| | <i>Sagittaria polyrrhiza</i> | Greater Duckweed | | x | NP | | NF | NF | NF | NF | NF | NF |
| | <i>Lemna minor</i> | Duckweed | | x | NP | | NF | NF | x | x | x | x |
| Submerged | <i>Ceratophyllum demersum</i> | Cornel | | x | x | x | x | x | x | x | x | x |
| | <i>Potamogeton pectinatus</i> | Sago Pondweed | | x | x | x | x | x | x | x | x | NF |
| | <i>Potamogeton crispus</i> | Crispy Pondweed | | x | x | | NF | NF | NF | NF | NF | NF |
| | <i>Potamogeton illinoensis</i> | | | NF | x | x | NF | NF | NF | NF | NF | NF |
| | <i>Elodea nuttallii</i> | Nuttall's Waterweed | | x | NF | x | NF | NF | NF | NF | NF | NF |

NF - not found in study

NP - not part of study (i.e. study restricted to certain species, e.g. submergents or to certain lake area, e.g. plant-openwater interface)

x - identified in study

(Chapter 3 discusses changes to the macrophyte community as a result of plantings and macrophyte-sediment relationships in the littoral zone in 1998.)

Other possible factors leading to plant loss and/or minimized expansion are: the removal/burial of the seedbed, increased salinity, littoral slope, wave action, competition with aggressive native and non-native species and disturbance/herbivory by large waterfowl populations.

1.1.3 Fisheries: Information on the Grenadier Pond fishery dates back as early as 1836, but the majority of such data exists from more recent decades (60's, 70's, 80's and 90's). Table 1.2 lists the fish species, any evidence of spawning and year of study for fish known to be in or have been in the pond. How many of the 21 fish species listed in Table 1.2 still reside in the pond today is unknown. The Toronto Region Conservation Authority (TRCA, formerly MTRCA) conducted the most recent sampling effort in June 1997 when a total of 10 species were caught (Jon Clayton, per. comm.). Four littoral locations around the pond were sampled by boat-electroshocking. The highest biomass was associated with the benthivorous brown bullhead (*Ameiurus nebulosus*) (19,044g), with the second highest biomass attributed to the benthivorous (as adults) pumpkinseed (*Lepomis gibbosus*) (11,202g). Pumpkinseed biomass has been characterized as "overabundant" (when small) in the fish survey reported by GLL (1993). Three large adult carp (*Cyprinus carpio*), non-native to North America, were also caught in the 1997 TRCA survey and are expected to exist in larger numbers in the pelagic. Significant carp spawning activity in both the north and south ends of the pond was observed by the author in 1998. Also found in the 1997 TRCA survey were 14 adult largemouth bass (*Micropterus salmoides*), a top predator in Grenadier pond; no bass fry were caught and no nests were observed during the survey. Sunfish (i.e. pumpkinseeds and bluegills) predate on unguarded bass nests and are generally under less fishing pressure than bass (GLL, 1995). Limited sites for spawning, over-fishing and competition from sunfish are likely factors in reducing the bass population despite stocking efforts.

Another top predator with historic presence in the pond is the northern pike (*Esox lucius*). Stocking efforts in the past decade may have maintained a remnant pike population, but recent surveys (1993 and 1997) did not catch nor observe any pike. Suitable pike spawning habitat would have been severely compromised by the past removal of shoreline vegetation (1995 and 1956, see Section 1.1.2) and the installment of a weir at the pond outfall in the second half of the 1900's (M. Boyce, per. comm). The weir effectively arrested seasonal flooding of remaining soft-stemmed riparian vegetation, a necessary condition for pike spawning (Scott and Crossman, 1973). Together with habitat alteration, over fishing and decreased water have likely lead to a decline, if not extirpation, of pike. Adult pike are a sought-after species for sport-fishing and they require clear, vegetated water to hunt in their typical "ambush" style (Grimm, 1989).

Table 1.2 History of Fish in Grenadier Pond *

| Fish Species Found | | Evidence of Spawning | Year of Study |
|------------------------|--------------------------------|--|--|
| Common Name | Scientific Name | | |
| Longnose Gar | <i>Lepisosteus osseus</i> | no evidence - 1976 | 1976 - Wainio |
| Bowfin | <i>Ameioba</i> | no evidence - 1976 | 1976 - Wainio |
| Gizzard Shad | <i>Alosa sp.</i> | Yeast breeding successfully - Wainio 1976 | 1976 - Wainio |
| Northern Pike | <i>Esox lucius</i> | no evidence - 1976 no evidence - 1987 | 1936 - Wainio 1980 - Wainio 1980 - Tough 1976 - Wainio 1987 - TRCA*** |
| White Sucker | <i>Catostomus commersoni</i> | no evidence - 1976 | 1976 - Wainio 1985/86 - DMD 1987 - TRCA |
| Gold Fish | <i>Carassius auratus</i> | no evidence - 1976 | 1976 - Wainio 1985/86 DMD 1987 - TRCA |
| Carp | <i>Cyprinus carpio</i> | observed mating behaviour - 1989 observed mating behaviour - 1976 observed mating behaviour - 1988 | 1989 - Tough 1971 - Wainio 1976 - Wainio 1985/86 - DMD 1987 - TRCA 1988 - CT |
| Golden Shiner | <i>Notemigonus crysoleucas</i> | abundant - 1976 | 1976 - Wainio 1986 - DMD 1987 - TRCA |
| Common Shiner | <i>Luxilus cornutus</i> | rare - 1976 | 1976 - Wainio |
| Brown Bullhead Catfish | <i>Ameiurus nebulosus</i> | spawning observed - 1989 by caught - 1976 | 1989 - Wainio 1989 - Tough 1976 - Wainio 1986 - DMD 1983 - GLL 1987 - TRCA |
| Channel Catfish | <i>Ictalurus punctatus</i> | no evidence - 1976 | 1976 - Wainio |
| Banded Killifish | <i>Fundulus diaphanus</i> | | 1989 - Tough 1971 - Wainio |
| Brook Silverside | <i>Coregonus hoyi</i> | | 1989 - Tough |
| Rock Bass | <i>Ambloplites rupestris</i> | | 1989 - Tough 1985/86 - DMD 1987 - DMD |
| Pumpkinseed | <i>Lepomis gibbosus</i> | abundant - 1976 all YOY - 1989 abundant - 1987 | 1989 - Tough 1971 - Wainio 1976 - Wainio 1984 - DMD 1985/86 - DMD 1983 - GLL 1987 - TRCA |
| Bluegill | <i>Lepomis macrochirus</i> | observed spawning - 1989 abundant - 1976 | 1989 - Tough 1971 - Wainio 1976 - Wainio 1985/86/87 - DMD 1987 - TRCA |
| Smallmouth Bass | <i>Micropterus dolomieu</i> | no evidence - 1976 observed nesting sites - 1987 | 1971 - Wainio 1976 - Wainio 1987 - DMD |
| Largemouth Bass | <i>Micropterus salmoides</i> | young fry caught - 1989 young fry caught - 1976 "successfully spawning" - 1987 | 1947 - Wainio 1989 - Tough 1971 - Wainio 1976 - Wainio 1985/86/87 - DMD 1987 - TRCA ** Stocked by Parks and Rec. 1986 and 1987 |
| Black Crappie | <i>Pomoxis nigromaculatus</i> | observed spawning - 1989 abundant - 1976 | 1980 - Wainio 1989 - Tough 1971 - Wainio 1976 - Wainio 1985/86/87 - DMD 1987 - TRCA |
| Yellow Perch | <i>Perca flavescens</i> | "successfully spawning" - 1976 | 1989 - Tough 1971 - Wainio 1976 - Wainio 1985/86/87 - DMD 1987 - TRCA |
| White Perch | <i>Morone americana</i> | | 1976 - Wainio 1986 - DMD 1983 - GLL |

* Table adapted and modified from GLL, 1986.

*** 1987 - TRCA data from Jon Clayton, per. Comm.

In general, fish diversity has declined over the past few decades with an over-representation of zooplanktivorous (i.e. pumpkinseed) and benthivorous (i.e. carp and catfish) fish species, which is a predictable shift as a system eutrophies.

1.1.4 Total Phosphorus Concentrations in Water Column: Grenadier Pond is considered highly eutrophic (Zimmerman *et al.*, 1986) and most of its problems, directly or indirectly, relate to excessive in-pond nutrient concentrations (GLL, 1995). Total phosphorus (TP) concentrations in the pond have been measured by various groups since 1986 to present and are summarized in Table 1.3. The range in TP in these records is from a minimum of 0.04 mg L⁻¹ in 1992 to maximum of 0.40 mg L⁻¹ in 1994, with an overall average of 0.15 mg L⁻¹ (\pm 0.12). The range in TP concentration from 11 other Greater Toronto Area lakes with a maximum depth >5 m is 0.01 to 0.06 mg L⁻¹ (Olding, 1998). The provincial water quality objectives recommend an in-lake TP concentration of 0.02 mg L⁻¹ or less to avoid nuisance algal growth (Ministry of Environment and Energy (MOEE), 1994).

Although a main source of present day external phosphorus to Grenadier pond is storm water (approximately 22% of annual water inputs), prior to 1990 the pond was vulnerable to direct discharge of untreated sewage during heavy rain events through combined sewer overflow (CSO) connections with storm water drains (GLL, 1995). In-pond TP levels have not demonstrated a clear pattern of decline since the CSO disconnection, likely due to internal loading from phosphorus-rich sediments. Chapter 3 discusses littoral sediment characteristics in further detail based on data collected in 1998. Another significant source of external phosphorus, identified in 1993, is bird feces that wash directly into the pond and are produced largely by an over-abundant population of resident Canada Geese (*Branta Canadensis*) (GLL, 1995).

1.1.5 Salinity: According to Zimmerman *et al.*, (1986), Grenadier Pond chemistry "is heavily influenced by sodium and chloride ions presumably derived from road salting." The report further states that in Grenadier Pond, chloride ions account for 29.3% of conductivity as compared to only 10.3% in other inland lakes. More recent water column chloride data are not available, however Chapter 3 presents and discusses chloride data for littoral sediments and possible effects on plant growth.

Table 1.3. Total Phosphorus Concentrations (mg L^{-1}) in Grenadier Pond, 1986-1997*

| Year Sampled | 1986 | 1992 | 1993 | 1994 | 1997 |
|-------------------------------|--------------|--|-------------|--|-------------|
| <i>Zimmerman et al., 1986</i> | 0.086, 0.087 | | | | |
| <i>City of Toronto</i> | | 0.13, 0.23, 0.14, 0.38, 0.04, 0.36, 0.32, 0.04, 0.35 | | | |
| <i>GLL, 1995</i> | | | 0.18, 0.12 | 0.09 ¹ , 0.22 ¹ , 0.09 ¹ , 0.16 ¹ , 0.4, 0.06, 0.09 | |
| <i>MOEE</i> | | | | 0.06, 0.06, 0.06, 0.06 | |
| <i>Olding, 1998</i> | | | | | 0.10 |

*Table adapted and modified from GLL, 1995, with permission.

¹ = Winter sampling (February); all others are summer sampling (July, Aug or Sept).

1.2 Grenadier Pond: Management Process from 1990 – 1998

1.2.1 Management Approach: In response to concerns of park users over the decline in High Park's natural environment, in particular the wetlands of Grenadier Pond, the City of Toronto's Parks and Recreation Department has undertaken rehabilitation planning and/or management actions since 1990. The overall eutrophied state of Grenadier Pond was considered unacceptable by the majority of park users and inconsistent with the City of Toronto's Parks and Recreation Department's dedication to park stewardship (City of Toronto, 1992). The City of Toronto (1992) presented an ecosystem approach to pond rehabilitation and received strong support from other municipal and provincial agencies, the University of Toronto, special interest groups and the local community. The following year, terms of reference for the rehabilitation of the aquatic ecosystem of Grenadier Pond were sent out to numerous local consulting firms (City of Toronto, 1993). The project was awarded that same year and a baseline study of the pond was initiated in the fall of 1993.

1.2.2 Issue Identification: The main ecological issues in Grenadier Pond are cultural eutrophication and the undesirable suite of changes this process has visited on Grenadier Pond and the associated wetlands. Adding to this present ecological stress are the residual effects of past alterations to pond habitat and dynamics.

Prior to professional consultation, six sub-issues were identified (City of Toronto, 1992; City of Toronto, 1993):

1. Aesthetic deterioration due to massive algal blooms (i.e. poor water clarity and malodour);
2. Impairment of wetland communities;
3. The current paucity of littoral vegetation/natural shoreline;
4. Inappropriate stabilization of the pond water level;
5. The high sediment load contained in storm water; and
6. Decreased recreational opportunities (i.e. bird-watching, fishing, boating and nature interpretation).

After the first wave of ecological issues was tabled, three important political/social issues were identified following public consultation (City of Toronto, 1992):

- A need to provide to park users and other interested groups, a greater understanding of existing conditions in Grenadier Pond;
- A need to inform park users and other interested groups of the rationale for selecting specific intervention steps, the degree and timing of management required; and
- Full disclosure to park users and other interested groups of the impact of proposed intervention and management techniques on the Pond's present ecology.

Some of these initial "complaints" about the pond were what park users could "see" or "smell" (i.e. 1, 2, 6), while a Technical Resource Group identified other issues (i.e. 2-5) as symptoms of larger ecological imbalances that should be given priority in a rehabilitation context (City of Toronto, 1992). Additional ecological issues that arose through consultation were (GLL, 1995):

- A) High nutrient loadings from external and internal sources;
- B) Lack of oxygen in the bottom waters;
- C) Abundance of waterfowl;
- D) Abundance of sunfish and decline in predator fish;
- E) Habitat alteration;
- F) Lack of wildlife diversity; and
- G) Human access and education.

1.2.3 Rehabilitation Goals: Through professional consultation and continual feedback from advisory level committees, six main ecological goals, four public education goals and one overall pond goal were set for Grenadier Pond and adopted in principle by the City of Toronto, Parks and Recreation Department. The various management goals will be collectively referred to as Rehabilitation Goals and are listed below:

Ecological Goals (GLL, 1995):

1. Target summer Secchi depth of 2.0m or greater;
2. Target in-pond TP concentration of 0.05 mg L⁻¹:
 - 2a) Decrease storm water loadings by 25%;
 - 2b) Decrease waterfowl loadings by 75%;
 - 2c) Increase groundwater flows by 25%;
 - 2d) Decrease internal nutrient loading by 75%;
3. Renaturalize substantial portion of shoreline and increase littoral zone to 20% of lake bed area;
4. Return or enhancement of target animal species;
 - 4a) Fish: Northern Pike and Largemouth Bass;
 - 4b) Mammals: Muskrat;
 - 4c) Amphibians: Leopard Frogs, Green Frogs and Gray Tree Frog;
 - 4d) Water birds: Virginia Rail, Sora, Rail Marsh Wren, Swamp Sparrow and Moor Hen;
5. Extend north wetland southward and increase total wetland area to 1.2 ha; and
6. Restore north wetland connection to Wendigo Creek.

Political/Social Goals (GLL, 1995)

1. Provide education outreach to anglers promoting catch and release of all fish;
2. Designate sanctuary zones during spawning activity of target fish (enforcement/patrol recommended);
3. Provide education outreach to all park users discouraging the feeding of all birds, particularly geese; and
4. Provide forum for ongoing public consultation and dissemination of information regarding proposed and current management actions.

Overall Goal (City of Toronto, 1992);

1. Eventually achieve a self-organizing, self-sustaining aquatic ecosystem.

1.2.4 Management Techniques: Table 1.4 provides a summary of management techniques or investigations undertaken by the City of Toronto and Partners to achieve rehabilitation goals in Grenadier -

Table 1.4. Chronology of Rehabilitation Techniques Applied to Grenadier Pond 13

| <i>Technique Category</i> | <i>Action Taken</i> | <i>Goal Addressed</i> | <i>Year Implemented</i> | <i>Agency/Partner/Sponsor</i> |
|--|--|-----------------------|-------------------------|---|
| Biomanipulation or other biologically based technique | piscivore stocking | ecological | 1994, 1995, 1997 | PR, CNS |
| | native plant propagation | ecological | 1994 | PR PR, GLCF2000, CNS, Friends of Altona Forest |
| | SE corner shoreline naturalization | ecological | 1995 | PR, GLCF2000, DU |
| | goose deterrents (barrier planting) | ecological | 1995 | PR |
| | installation of brush bundles | ecological | 1995 | MNR |
| | aquatic plant propagation | ecological | 1996 | TRCA, local school |
| | shoreline softening/regrading (SW corner, East cove, East Shore) | ecological | 1996 | PR, GLCF2000, DU |
| | installation of wood duck nesting boxes | ecological | 1996 | TRCA, DU |
| | shoreline and macrophyte planting (SE corner, East cove, East shore) | ecological | 1997 | PR, GLCF2000, DU |
| Engineered or Traditional | north wetland restoration | ecological | 1997/1998 | TRCA, PR |
| | Hillside Gardens irrigation modification | ecological | 1995 | PR |
| | Watershed downspout disconnection program | ecological | 1995/1996 | City Works Service |
| | Pond outfall weir reconstruction | ecological | 1996 | PR, City Works, GLCF2000, DU |
| | Storm water retention pond expansion | ecological | 1996/1997 | City Works Service, PR |
| Educative | Storm water quality monitoring | ecological | 1996/1997 | City Works Service |
| | educative signage | public education | 1994 - 1997 | PR |
| | walking tours | public education | 1994 - present | citizens, PR in-kind support |
| | pond rehab. Display in local school | public education | 1994 | High Park Alternative School |
| Investigative | yellow fish road program | public education | 1995 | HPCAC |
| | permanent habitat restoration signage | public education | 1997 | PR |
| | baseline pond study | ecological | 1993 | PR, GLL |
| | sediment characterization study | ecological | 1996 | PR, EC, TRCA |
| | groundwater influence study | ecological | 1996 | PR, GLL |
| | pilot sediment treatment (discussions only) | ecological | 1997 | PR, EC, TRCA |

PR = Parks and Recreation, City of Toronto; CNS = Canadian National Sportsman Show; DU = Ducks Unlimited; GLCF2000 = Great Lakes Cleanup Fund; TRCA = Toronto Region Conservation Authority; EC = Environment Canada; HPCAC = High Park Citizens Advisory Committee; GLL = Gartner Lee Limited;

(Summarized in City of Toronto, 1997). The various techniques are organized by type (e.g. biological, engineered, educative) and listed in order of occurrence. The general rehabilitation goal(s) addressed by each management action is also indicated.

Note, monitoring efforts independent of restoration goals have been carried out by a number of government agencies: TRCA (macrophyte) and MOEE (in-pond water quality) and City Works Services (storm water quality and downspout disconnection).

1.3 Discussion of Different Management Approaches

Grenadier Pond managers have taken an ecosystem approach to pond rehabilitation. This means considering all the connections, physical, chemical, biological, within the natural environment and the temporal scale in which they function. It is an attempt to understand the greater picture and correct imbalances or remove deleterious influences within a continuum context. At the crux of this paradigm is the notion that piecemeal restoration is not likely to be sustainable and emphasis should be placed on restoring the overall system of which a remnant was once a part (Moss, 1999). This could ultimately point towards re-establishing connections between Grenadier Pond and Lake Ontario in addition to re-establishing fluctuating water levels, naturalizing the shoreline and "re-connecting" the north wetland, all of which follow this holistic reasoning. However, re-creating the past does not always come without an ecological price. Re-connection between Lake Ontario and Grenadier Pond may bring about more problems than currently exist, as Lake Ontario has certainly sustained impacts to water quality and the fishery during the 20th century that would have unknown effects on Grenadier Pond (GLL, 1995). Managers, hopefully working in concert with scientists, must decide carefully what is ecologically advisable and/or viable. For instance, it is not possible to de-urbanize Grenadier's watershed, but projects like downspout disconnection should be evaluated carefully, as they affect groundwater hydrology. Perhaps the greatest social/political challenge to this approach is figuring out how to fit humans, and all their modern activities into this paradigm, and fit they must. In recognition of this need, consultation with the public on goal development has been a priority management action for Grenadier Pond, as it was in other studies employing an ecosystem approach (Kairesalo *et al.*, 1999; MTRCA, 1997).

A variety of rehabilitation/restoration techniques may be considered within an ecosystem approach, as many different kinds of problems need to be addressed within an entire aquatic ecosystem. However, the overall Grenadier management team, including advisory groups, has in general not accepted suggested actions like sediment removal, capping or chemical application. These kinds of techniques were seen as too aggressive or unproven and not compatible with an ecosystem approach. High cost, typically associated with these techniques, may also have been a significant factor. Even a new storm water settling pond (although eventually approved) met with considerable contention, primarily over loss of "play area" for

area residents' children. Conversely, the biomanipulation projects have been well received in spite of not having any long-term monitoring program to document progress or success. This project and peripheral TRCA efforts are the only exceptions. Acceptance may be linked to biomanipulation's more passive approach and its cost effectiveness (see Chapter 2).

A second approach to "rehabilitation", with some similarities to ecosystem management, is integrated management (IM). IM studies are usually characterized by collaboration between lake managers, scientists and the public in the determination of rehabilitation goals with the combined use of engineered, chemical and biological techniques (Carpenter and Lathrop, 1999; Donabaum *et al.*, 1999; Walker and Havens, 1995; Heiskarky and Walker, 1995). These are also elements of the ecosystem approach, however, the studies employing integrated management often include traditional engineering techniques (e.g. dredging, chemical flocculation of the sediments, settling ponds, in-flow diversion). These studies often focus on attaining only one "ecological" goal, i.e. reduction in either chlorophyll-*a* concentration or algal assemblages (Heiskarky and Walker, 1995; Walker and Havens, 1995, respectively), although this "goal" is often expressed as one of "improved water quality" which usually means improved water clarity. A holistic picture of the system does not appear to be pivotal to this approach. Donabaum *et al.*, (1999) indicated that integrated management could be very expensive, but that some results can occur within a timeframe agreeable to typical lake management schedules.

A third approach is often titled resource-based management. For freshwater ecosystems, this usually refers to: fisheries, drinking water supply or recreation. Human-centred activity determines the resource and rehabilitation efforts focus strictly on the "needs" for that resource. For example, a joint management committee overseeing fish-stocking control programs is considered comprehensive if it includes anglers, commercial fisherman and water managers (Hosper and Jagtman, 1990). Stream restoration efforts undertaken by some Conservation Authorities fall under the direction of fish management plans that primarily consider fish community composition (past and present), fish health and stream catchment characteristics as they pertain to fish habitat (NVCA, 1998).

1.4 Comparison of Grenadier Pond Issues to Studies in the Literature

The following section identifies the main ecological and political/social issues for lake rehabilitation found in the recent literature. These are compared to those identified for Grenadier Pond and reasons for differences are discussed.

1.4.1 Ecological Issues

1.4.1.1 Limited Understanding: When dealing with eutrophication, a commonly cited ecological issue was the limited understanding of how to affect internal nutrient loading, particularly in shallow lakes (Welch and Cooke, 1995; Walker and Havens, 1995). Initially, Grenadier Pond managers did not list internal loading separate from external loading as a major issue. The omission likely occurred because issues were tabled at the beginning of the management process, before pond conditions were assessed, and therefore the significance of internal loading was not clearly stipulated until later. The issue of gaining a better understanding of existing conditions, first identified by Grenadier Pond users, is echoed in the literature and expanded upon by Moss (1999) to include greater knowledge of ecological function on a holistic plane. Phillips *et al.*, (1999) discuss the relevance of understanding function to lake management using the complexity of eutrophic influences: they have proved to be more complex than previous research suggested. This translates into lake managers no longer having the assurance that a given technique, directed simply at water quality improvement, will also result in predictable ecological change.

1.4.2 Political/Social Issues

1.4.2.1 Funding: Perhaps surprisingly, funding issues for the rehabilitation of Grenadier Pond were not highlighted in the management process. This may be because the groups being consulted on issue identification (i.e. technical advisors, interest groups, park users) would ultimately not have a role in deciding the level of funding afforded to the effort. Grenadier Pond rehabilitation is largely government funded at the municipal and federal (GLCF2000) levels and therefore subject only to internal review. Another possibility is that Grenadier Pond managers decided that the high costs of ecosystem management were "necessary to resolve conflicting interests of conservation, angling, swimming and other recreational activities" (Donabaum *et al.*, 1999). As it stands, funding has been reasonably forthcoming to Grenadier Pond, in part because the work is essentially applied limnology with a potential to try new or innovative technologies (M. Boyce, Per. Comm.). This follows the recognized, and rather regrettable, trend in freshwater research that applied projects attract the bulk of government funding over fundamental research (Moss, 1999). Another funding issue is tied to controlling non-point pollution (e.g. phosphorus), which often requires improvements to land use that are too expensive for lake managers to support (Carpenter and Lathrop, 1999). To alleviate this economic burden, the Parks and Recreation Department partnered with agencies responsible for storm water control/quality (i.e. Public Works and MOE), thus dividing the expense of external nutrient reduction in the watershed. However, such opportunities may not present themselves every situation. In addition, cost-sharing – focused externally - does not address the considerably higher expense of reducing internal nutrient loading, which, ultimately, may be the key to reversing eutrophication in the system (Welch and Cooke, 1995).

1.4.2.2 Lake User Conflict: Ecologically sustainable lake rehabilitation is strongly dependent on the commitment of lake users to support appropriate management techniques (Kairesalo *et al.*, 1999). Grenadier Pond managers addressed this issue from the beginning of the management process with significant and ongoing consultation with lake users and interest groups. Several other lake studies identified the need to obtain this support but tended to focus on anglers over any other user group (Carpenter and Lathrop, 1999; Shapiro, 1990; Grimm, 1989). As Harper (1992) identified, concentration on anglers can lead to limnological considerations taking a "back seat" to fish management issues only to find that fish stocking or removal are not universally acceptable measures, even if it might result in a return of ecological lake health. This was the case for Lake Alte Donau, a shallow urban lake in Vienna (Donabaum *et al.*, 1999). The Grenadier Pond project has gone ahead with fish stocking with the financial support of the Canadian National Sportsman's Show. But local, individual fishers have been a largely passive audience via educative signage around the pond and a few self-motivated attendees at public meetings.

1.4.2.3 Motivational Discrepancy: According to Carpenter and Lathrop (1999), lake managers have to reconcile the temporal differences between political and ecological scales. The reversal of a process that has taken decades to establish will take time, considerably longer than the average elected term of office. Longer-term thinking, by comparatively short-lived governments, is a significant paradigm shift that has not yet entered our society, but may well be the most important step towards effective and successful ecological management (Philips *et al.*, 1999). In the mean time, for however long political commitment lasts, it is important that lake managers be able to identify successful management actions (Walker and Havens, 1995). This important issue was not identified for Grenadier Pond, but has strong relevance to future management, as the Toronto municipal government recently merged with neighbouring cities without specifying commitment to current projects.

1.5 Comparison of Grenadier Pond Goals to Studies in the Literature

From the literature and various policy documents, I have identified five main ecological goals and compared them with Grenadier Pond. As for political/social goals, other studies, if they mention them at all, tended to discuss these items as adjunct activities to the ecological recovery plan. Their inclusion was deemed important, even crucial for some projects, but not identified as a "rehabilitation or management goal". Therefore, the following section focuses on ecological management goals.

1.5.1 Goal #1: Clear Water and/or Reduction of Nuisance Algal Blooms: Depending on the lake, clear water and reduction of algal blooms may be technically synonymous. In some cases, problem turbidity may be definitively more sediment related. In the case of the latter, the goal of clear water may require attention to sediment resuspension. While a single, undifferentiated goal of clear water or reduction in

nuisance algae was common in the literature (Perrow *et al.*, 1997; Heiskarky and Walker Jr, 1995; Carpenter and Kitchell, 1992) and adopted by Grenadier Pond managers, other similarly stated goals included attributes often associated with clear water, such as macrophyte dominance and fish densities (Timms and Moss, 1984; Grimm, 1989). Unlike those for Grenadier Pond, other studies did not cite a specific Secchi depth as a target for water clarity.

1.5.2 Goal #2: Target Values for External and/or Internal Nutrient Load Reduction: Strong agreement exists among scientists and managers that a reduction in external and/or internal nutrient loading must be part of a lake recovery scheme (see Chapter 2). However, managers would sooner support a standardized or readily determinable number on which to anchor their nutrient reduction strategies, as was determined for Grenadier Pond. This "number" is scarce within the literature as lake systems differ greatly; assigning a single target phosphorus value across ecoregions or even larger areas is not appropriate. Alternatively, water quality goals could specify an average, or minimum or maximum concentration for target parameters for a specific period of time (Heiskarky and Walker, 1995) particularly when tied to current or historical levels (e.g. 25% reduction or within 20% of historical levels). While computer models are often used to aid in the diagnosis of an individual lake's needs, precision and accuracy are generally proven or disproven only upon application (and sometimes not even then) and rarely are the results directly transferable to other rehabilitation projects (Lehman, 1986). Consequently, a general goal of nutrient load reduction is most often cited in the literature (Donabaum *et al.*, 1999) with specifics left to case studies that may or may not lead to success in other lakes. An exception to this trend is the Netherlands government that has set a reduction rate for external phosphorus loading at $1\text{g.P.m}^{-2}\text{.year}^{-1}$ to promote lake recovery (Grimm, 1989). This goal was originally intended to apply to point source phosphorus loads, such as sanitary sewer outflows, and may not be achievable in cases of non-point sources, which are typically harder to control. The Ontario government has also set water quality objectives for in-lake phosphorus concentrations at $20\mu\text{g L}^{-1}$ to avoid nuisance concentrations of algae, and $10\mu\text{g L}^{-1}$ to protect against aesthetic deterioration (MOEE, 1994), but these are only guidelines for water quality managers.

1.5.3 Goal #3: Aquatic Habitat Restoration: Of particular and growing interest is the restoration of wetlands, macrophytes and fallen trees (Carpenter and Lathrop, 1999). These bio-structural goals are fundamentally important to sustaining the aquatic communities within a restoration framework (e.g. littoral-using fish, benthic invertebrates, water fowl). Further, macrophyte communities, and to a lesser extent wetlands, may be a critical factor in causing/maintaining food-web shifts that favour a clear-water state over turbid waters (see Chapter 2 for details). Grenadier Pond has incorporated habitat restoration goals for all of the above reasons.

1.5.4 Goal #4: Improve Fishery: In many eutrophied lakes, the loss of top predator fish is common (e.g. pike and bass). For both ecological and recreational reasons, a return of such fish is generally supported,

although a goal of increasing the value of the fishery is perhaps secondary or minor (Kairesalo *et al.*, 1999). Grenadier Pond managers support a return of not only top predator fish, but also other animals associated with an intact aquatic ecosystem.

1.5.5 Goal #5: System Self-Sustainability: Moss (1999) identifies perhaps the most important management goal as the restoration of a system "to conditions where minimal or no on-going management is needed." Having a system once again functional without perceived impairment or artificial support mechanisms within an urban or otherwise altered environment is tantamount to human society successfully re-entering the larger ecological equation. This goal, explicitly stated, does not regularly appear in the literature, but is strongly espoused by all who have participated in developing a vision for Grenadier Pond. At present, there is a split in the scientific community as to whether positive results from commonly used restoration techniques can be maintained without continuous management (see Chapter 2). Grenadier Pond managers have taken this dichotomy seriously by rejecting management techniques (e.g. hypolimnetic aeration) that do not compliment the goal of system self-sustainability.

1.6 Assessment of Techniques Used in Grenadier Pond

The array of techniques applied to Grenadier Pond is generally in line with other shallow lake rehabilitation projects (Donabaum *et al.*, 1999; Kairesalo *et al.*, 1999; Philips, 1999; Carpenter and Lathrop, 1999). Reinstitution of water-level fluctuation and investigations to increase groundwater inflow are, however, unique to this project. In Chapter 2, the merits of traditional (i.e. external and/or internal nutrient load reductions) and biological (i.e. biomanipulation) techniques are discussed in detail, concluding that a combination of these techniques is the recommended option. Philips *et al.*, (1999) suggest the order of implementation should be: 1) reduction of external point and non-point sources of phosphorus; 2) reduction of internal phosphorus loading; 3) biomanipulation to create clear water conditions that encourage macrophyte growth; and 4) focus on developing diverse plant and animal communities to stabilize managed fish communities.

Strictly speaking, significant external nutrient reduction efforts did not occur as the first management action in Grenadier Pond (i.e. storm water quality improvements began in the third and fourth years of management, see Table 1.4). Biomanipulation, in the form of piscivore (northern pike) stocking, was the first action to take place in 1995. The potential benefits of this action were probably not realized by the pond, as minimal fish biomass/lake area was not considered (see Chapter 2), nor was the provision of spawning or foraging habitat for the new residents. Other shallow lake studies in the literature have demonstrated that piscivore stocking is most successful at shifting food web structure when teamed with extreme planktivore removal by as much as 75% (see Chapter 2). The mass removal of such fish in Grenadier Pond (e.g. pumpkinseed and bluegill) would not be compatible with the recreational fishing that

is a popular past time on the pond, and would likely take considerable public education to convince pond users that a loss of fish would be a "good thing". Also, the cost of such a large-scale operation was, in the early management phase, prohibitive. It may be still.

Piscivore stocking continued for the next two years, with more aggressive habitat improvements to coincide with pike and bass life-cycle requirements. The most recent fish survey (TRCA, 1997) did find adult largemouth bass, but no pike were caught in the nearshore areas (see Table 1.2). Spawning surveys for these stocked fish species have not been undertaken.

The obvious remediation technique that has not been put into practice is internal nutrient load reduction from the sediments. Since existing conditions were documented, the importance of this nutrient source has not been overlooked. Almost all the investigative measures are geared towards assessing nutrient control from the sediments or diluting the effect of sediment-phosphorus release.

Grenadier Pond falls into the class of shallow lakes whose summer thermal stratification is vulnerable to erosion during periods of high wind. According to Welch and Cooke (1995), this situation effectively allows hypolimnetic phosphorus, released from deep, anaerobic sediments, to reach the epilimnetic zone and facilitate the replication of fast-growing blue-green algae. Their paper also suggests that long-term management of shallow lakes should utilize techniques that address phosphorus-sediment stores, with extent of treatment depending on the magnitude of internal nutrient loading (Welch and Cooke, 1995). The contribution of internal loading in Grenadier Pond has been estimated at 124.4 kg year⁻¹ or 32.1% of total loadings (GLL, 1995), but more recent speculation by consulting biologists and Grenadier's Technical Resource Group members speculate that internal loading is perhaps the main source of phosphorus in the system. Sediment dredging and chemical flocculation of sediments are techniques that demonstrate a high degree of success when used in concert and/or sequence with other rehabilitation measures (Donabaum *et al.*, 1999; Phillips *et al.*, 1999). Grenadier Pond managers considered both of these engineered approaches, and several others, but as discussed in the previous section, costs were too high and/or compatibility with an ecosystem approach too low. New technology for sediment removal, that boasts lower impact on overall pond ecology, has been investigated but no commitment made. To date, littoral macrophyte plantings are the only management action that may impact sediment nutrient content. This hypothesis is explored in detail in Chapter 3.

1.7 Summary

The majority of issues identified and ecological goals developed for Grenadier Pond are typical of those associated with other urban pond rehabilitation studies. Specific ecological considerations, unique to the Grenadier Pond system (e.g. hydrological history with respect to Lake Ontario, alteration history, current

watershed contributions/conditions), demand that a tailored rehabilitation plan be devised and allow for some originality of management techniques and investigations.

The majority of management actions funded to date have a strong visual or intuitive appeal to lake users and managers (e.g. fish stocking, shoreline planting, settling pond, water level fluctuation). The theory of how these techniques are supposed to bring about improvements is reasonably clear. These techniques have been used on other lakes, some create an immediate aesthetic impact and all are amenable to "billboard/pictorial" summaries. They therefore also complement the goals of education and of communication with the general public. It is, perhaps, for these reasons that full funding has been successfully obtained for only these rehabilitation actions, two of them being biomanipulation techniques.

Grenadier Pond managers, like many others, have the difficult job of attracting sponsorship to less "glamorous" and less well understood management techniques, specifically internal nutrient load reduction. This is a challenge that must be met, given that the general consensus of the scientific community is that addressing the internal nutrient load should be one of the first steps taken towards ecological recovery from eutrophication. Until such funding becomes available the schedule for implementing rehabilitation efforts seems destined to follow the subjective interests of funding agencies and/or corporate sponsorship instead of a recovery agenda more firmly rooted in ecological theory. This is not to say that the projects to date will not have a positive impact on Grenadier Pond, but only monitoring, preferably long-term, is likely to provide this information.

Although the rationale for monitoring was clearly stated in the rehabilitation management plan, funding again has been an issue for Grenadier Pond. Other studies that have benefited from comprehensive monitoring are linked mostly to academic institutions. Carpenter and Kitchell (1992) strongly argue that whole-lake projects, like that for Grenadier Pond, are very appropriate for the scientific study of biomanipulation. The fact that Grenadier Pond managers have recently capitalized on such an opportunity, by helping to fund an evaluation of the shoreline/littoral naturalization projects, favourably demonstrates the less-common commitment to integrate scientific understanding with lake management.

1.8 References

Carpenter, S. R., and R.C. Lathrop, 1999. Lake restoration: capabilities and needs. In *Hydrobiologia*, 395/396 : 19-28

Carpenter, S.R. and J.F. Kitchell, 1992. Trophic Cascade and biomanipulation: Interface of research and management - A reply to the comment by DeMelo et al., In *Limnology and Oceanography*, 37 (1) : 208-213

City of Toronto, 1997. *Grenadier Pond - Shoreline Softening, 1996-1997 Year-End Progress and Financial Report*. Prepared by the Department of Parks and Recreation for Great Lakes 2000 Cleanup Fund. Unpublished.

City of Toronto, 1993. *High Park Proposals for the Rehabilitation of the Aquatic Ecosystem of Grenadier Pond, Wendigo Creek* . Unpublished.

City of Toronto, 1992. *High Park Proposals for Restoration and Management and Framework for Implementation. A Report to Neighbourhoods Committee*. Prepared by the Department of Parks and Recreation. Unpublished.

Donabaum, K., Schagerl, M., and M.T. Dokulil, 1999. Integrated management to restore macrophyte domination. In *Hydrobiologia*, 395/396 : 87-97

Gartner Lee Limited, 1995. *Proposals for the Rehabilitation of Grenadier Pond, Wendigo Creek and Associated Wetlands*, prepared for the City of Toronto Parks and Recreation Department. Unpublished consultant's report.

Grimm, M.P., 1989. Northern Pike (*Esox lucius* L.) and aquatic vegetation, tools in the management of fisheries and water quality in shallow waters. In *Hydrobiol. Bull.* 23: 59-65

Heiskarky, S.A. and W.W. Walker, Jr., 1995. Establishing a chlorophyll *a* goal for run-of-the-river reservoir. In *Lake and Reservoir Management*, 11(1) : 67-76, November

Kairesalo, T, Laine, S., Luokkanen, E., Malinen, T., and J. Keto, 1999. Direct and indirect mechanisms behind successful biomanipulation. In *Hydrobiologia*, 395/396 : 99-106

Lammens, E. H., 1999. The central role of fish in lake restoration and management. In *Hydrobiologia*, 395/396: 191-198

Lehman, J.T., 1986. The goal of understanding in limnology. In *Limnology and Oceanography*, 31(5):1160-1166

Melzer, A., 1999. Aquatic macrophytes as tools for lake management. In *Hydrobiologia*, 395:396 : 181-190

Ministry of the Environment and Energy, 1994. *Water Management, Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of the Environment and Energy*. Queen's Printer for Ontario. ISBN 0-7778-3494-4.

Moss, B., 1999. Ecological challenges for lake management. In *Hydrobiologia*, 395/396 : 3-11

Metropolitan Toronto Region Conservation Authority (MTRCA), 1997. *Legacy: A Strategy for a Healthy Humber; The Report of the Humber Watershed Task Force*. Published by the Metro Toronto Region Conservation Authority

Nottawasaga Valley Conservation Authority, 1998. *A Framework Approach to Stream Restoration Monitoring*. Document issued at Nottawasaga Valley Conservation Authority workshop on stream restoration. March. Unpublished.

Olding, D.D., 1998. *Evaluation of Water Quality and Algal Communities of Triangle Pond and Grenadier Pond, Summer 1996, 1997. Report #9807.1*, prepared for Toronto Region Conservation Authority. Unpublished.

Perrow, M., Meijer, M.L., Dawidowicz, P. and H. Coops, 1997. Bio-manipulation in shallow lakes: state of the art. In *Hydrobiologia* 342/343: 355-365

Phillips, G., Bramwell, A., Pitt, J., Stansfield, J., and M. Perrow, 1999. Practical application of 25 years' research into the management of shallow lakes. In *Hydrobiologia*, 395/396 : 61-76

Scott, W.B. and E.J. Crossman, 1973. *Freshwater Fishes of Canada*. Canadian Government Publishing Centre, Ottawa, Canada.

Timms, R.M. and B. Moss, 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. In *Limnology and Oceanography*. 29(3): 472-486

Wainio, A., Barne, J., Roswell, J. and K. McIntosh, 1976. *An Ecological Study of Grenadier Pond and the Surrounding Areas of High Park - Toronto*. Unpublished (sponsored by General Foods)

Walker, W.W., Jr. and K.E. Havens, 1995. Relating algal bloom frequencies to phosphorus concentrations in Lake Okeechobee. In *Lake and Reservoir Management*, 11(1) : 77-83, November

Welch, E.B and G.D. Cooke, 1995. Internal phosphorus loading in shallow lakes: Importance and control. In *Lake and Reservoir Management*, 11(3) : 273-381, December

Chapter 2 : An Overview of Alternate Stable State and Biomanipulation Theories, Application and the Special Role of Aquatic Macrophytes: Literature Review

2.0 Introduction

Cultural eutrophication, as opposed to the more gradual, natural eutrophication, remains one of the world's principal problems of water quality (Boers *et al.*, 1991; Phillips *et al.*, 1999). Late-stage eutrophication is defined by excessive nutrient loading. Sometimes this is associated with a deterioration or loss of macrophytes, particularly a decrease in the area occupied and biomass of submerged species (Timms and Moss, 1984; Hosper and Jagtman, 1990; Ozimek *et al.*, 1990), high algal biomass dominated by cyanobacteria (de Bernardi and Giussani, 1990; Boers *et al.*, 1991), increased turbidity (Meijer and Hosper, 1997) and an increase in sediment loading from the surrounding watershed (Timms and Moss, 1984). Further consequences to the food-web may include a decline or disappearance of top predator fish (e.g. northern pike from northern temperate lakes), dominance by cyprinids, a shift to small-bodied zooplankton (Meijer *et al.*, 1989; Boers *et al.*, 1991; Carpenter and Lathrop, 1999) and, in some cases, winter fish kills (Moss, 1984) brought about by progressive bottom-water anoxia. In contrast to the more widely observed effects listed above, an increase in macrophytes to the "nuisance level" can also occur (Galanti *et al.*, 1990).

The management of eutrophication can depend on the political and social "setting" as much as the biological considerations and limitations of a system. For example, in urban-centred, shallow ponds the focus is on eliminating or reducing unsightly algal blooms that negatively affect the recreational/conservation value of a water body (Hosper and Jagtman, 1990). Urban park/pond users place high value on clear-water for swimming and visual aesthetics (Moss, 1990; Gartner Lee Limited, 1995) with a growing interest in ecological integrity/ecosystem health. There is a general perception by the public that biomanipulation techniques are more symptomatic than preventative but are more synchronized with goals of ecological integrity than traditional engineered techniques. Concerns are often expressed that further "damage" will result from heavy-handed or potentially irreversible measures.

Eutrophication has conventionally been thought of as a continuum – driven by ever-increasing nutrients. Recently, this continuous perspective has been challenged by researchers who have advocated an alternative stable state perspective suggesting that aquatic systems exist in either a turbid- or clear-water system state relatively independent of nutrient concentrations. Such alternative system states might be less

responsive to the traditional approach to eutrophication management: nutrient abatement and reduction. Eutrophied systems in a turbid stable state might be better managed through biologically based restoration techniques such as food-web alteration (i.e. biomanipulation) that do not necessarily assume low nutrient levels. The introduction of biomanipulation gives lake managers and researchers an additional, or perhaps alternative method, to nutrient abatement or engineered options for the improvement of water quality (Kairesalo *et al.*, 1999; Gophen, 1990). This chapter examines alternate stable state theory, with a focus on macrophyte function, and its relevance to biomanipulation as a management strategy for eutrophied, shallow ponds on largely urbanized watersheds.

2.1 Alternate Stable States

The potential for existence of alternate stable states has major implications for lake management (Perrow *et al.*, 1997; Meijer and Hosper, 1997; Jorgensen and di Bernardi, 1998). This theory suggests that the turbid, eutrophied state is not necessarily the only eutrophied state. This section discusses stable state theory, as it is currently understood in shallow water systems where the littoral zone plays a large role in overall system dynamics.

The two possible states identified are a clear- and a turbid-water state. The turbid-water state is descriptively similar to conventional eutrophication, however there is no suggestion that the mechanisms driving either state are the same. The turbid-water state is characterized by moderate-high nutrient levels, no or minimal submerged macrophyte beds, loss of piscivores dependent on littoral macrophyte habitat (e.g. pike), an over abundance of planktivores (e.g. bream) and/or benthivores, disappearance of large bodied zooplankton, productive, abundant phytoplankton, particularly of the blue-green variety (Hosper and Jagtman, 1990; Beklioglu and Moss, 1996; Perrow *et al.*, 1997; Jorgensen and de Bernardi, 1998). In contrast, the clear-water state is characterized by a range of low to high nutrient levels, abundant macrophyte beds (particularly submergent, but also floating and emergent), piscivores that effectively prey on planktivores and abundance of large bodied zooplankton, such as *Daphnia* (Hosper and Jagtman, 1990; Beklioglu and Moss, 1996; Jorgensen and de Bernardi, 1998; Carpenter and Lathrop, 1999). A variety of algal species (often greens and diatoms) are also a part of the clear-water state, with representation varying between lake systems. Research indicates that both states have significant internal buffers (i.e. negative feedbacks) that resist change and stabilize the system in one or the other state (Hosper and Jagtman, 1990; Perrow *et al.*, 1997). However, a major question is whether each of the states exists only under definably different nutrient regimes or whether they can exist alternately despite nutrient concentrations (Moss, 1990).

Turbid states are generally associated with high nutrient concentrations (i.e. eutrophy), but so are some clear-water, macrophyte-dominated situations, especially in very shallow ponds. Whether managers subscribe to an alternative stable state paradigm or the conventional phosphorus-eutrophication paradigm, nutrient abatement programs are still the most common approach to dealing with "eutrophied systems" (Meijer *et al.*, 1989; Gophen, 1990; Carvalho, 1994). However, Boers *et al.* (1991) point out that nutrient abatement alone is insufficient to shift the turbid-water system when internal loading, i.e. the recycling of bioavailable nutrients from the sediments to the water column, is high. Furthermore, the nutrient levels of shallow urban lakes may be so high that a reduction is not a practical goal without risking significant biological/ecological losses through heavy-handed engineered interventions (e.g. draining/dredging or chemical application). The interest in evaluating stable state theory is the possibility that system state could be "flipped" into a clear-water phase at relatively high nutrient levels.

The argument for two discreet regimes suggests that macrophyte dominated, clear-water systems may exist uniquely in shallow lakes with low water column available phosphorus as rooted plants can better exploit the nutrient reserves found in sediment; similarly, phytoplankton dominance may exist uniquely when nutrient content is very high, flushing rates are low and cyanophytes are already the dominant algal community (Timms and Moss, 1984; Beklioglu and Moss, 1996). However, Meijer and Houser (1997) offer some empirical data suggesting that there is a range of nutrient concentrations within which either stable state can exist. Perrow *et al.* (1997) suggest that the shared zone falls within the intermediate nutrient level or even high nutrients if the system is nitrogen limited. Jorgensen and de Bernardi (1998) contend that the internal buffer capacity of either state is stressed only as nutrient levels rise (clear-water) or decrease (turbid water) to threshold values, maintaining either the clear or turbid state up to a point. Later I evaluate case studies to help define what might be the upper nutrient threshold for a clear-water state.

2.1.1 System State Stability: Identifying conditions that stabilize the turbid- or clear-water states is a reasonable starting point to answering the question of whether either state can exist independent of the nutrient regime. Turbid-water systems rely on four main stabilizing mechanisms: 1) phytoplankton competition with macrophytes 2) changes in phytoplankton grazability, 3) low grazing pressure by zooplankton and 4) physical disturbance. The clear-water state is also maintained through mechanisms that directly counter those of the turbid-water state. These opposing forces will be described jointly, followed by a discussion of stabilizers unique to the clear-water state.

2.1.1.1 Phytoplankton Competition with Macrophytes: An established and dominant phytoplankton community can help to perpetuate its reign by out competing its vascular counterparts for nutrients (P, N, C), restricting macrophyte light levels and earlier seasonal growth (Moss, 1990). In the case of cyanophytes, many species have floatation mechanisms that allow them to compete for surface light. They may also be able to avoid grazing by migrating to lower water levels in the evening, the prime time for

grazing by the large-bodied zooplankton *Daphnia* (DeMelo *et al.*, 1992), although extensive grazing on cyanophytes by *Daphnia* is not universally accepted. In contrast, established macrophyte beds may successfully compete for phosphorus through storage as biomass or through an ability to exploit sedimentary nutrients (Timms and Moss, 1984, Carvalho, 1990; Meijer and Houser, 1997). There is also some evidence for a denitrifying microenvironment occurring within the beds of rooted aquatic plants (Beklioglu and Moss, 1996b) or lower pH's favouring desirable carbon species. Of course, these advantages over phytoplankton may only be relevant if nutrients are limited. The physical presence of dense canopy structures can also effectively shade-out seasonally-late algal species (Ozimek *et al.*, 1990; Carvalho, 1990; Meijer and Houser, 1997). There is some evidence that, when macrophytes are dominant and nutrient levels low to moderate, species composition may shift to favour taller plants that can better deal with the shading of epiphytes and filamentous algae (Moss, 1990; Houser and Jagtman, 1990).

2.1.1.2 Changes in Phytoplankton Grazability: The net effect of eutrophication (low N:P ratio, high pH) on the algal community is dominance of cyanophytes, an algal groups generally considered "inedible" (as defined by shape, size, toxicity) as compared to other species (de Bernardi and Guissani, 1990). De Bernardi and Guissani (1990) determined that certain species of cyanophytes (i.e. *Anabaena flos-aquae* and *Aphanizomenon*) are of poor nutrient value and therefore considered a poor food source for *Daphnia*. Other cyanophyte species also have the tendency to gather in large colonies forming a physical deterrent to the gape-size limitations of *Daphnia* and other grazers. The toxicity of some blue-greens to aquatic biota is well documented (Moss, 1990; Carmichael and Gorham, 1977). A pervasive argument, born from these observations, is that grazing pressure by large-bodied zooplankton is reduced due to the inedibility of dominant algae species and thus helps to stabilize their persistence (Gophen, 1990; McQueen, 1990; de Bernardi and Guissani, 1990; Timms and Moss, 1984).

However, regulation of algae may not be driven by their availability to grazers. A paper by Augusti *et al.* (1992), postulates the existence of a "natural upper-limit" to zooplankton grazing suggesting that grazing pressure reaches some maximum that cannot increase as eutrophication maximizes. The paper looked at phytoplankton abundance and size in Florida lakes, the importance of self-regulation of the algal community and the level of evidence for nutrient limitation in limnetic phytoplankton. It concluded: 1) the ceiling on grazing is most apparent when the algal community is dominated by cyanophytes and 2) the rate of increase in zooplankton abundance decreased as biomass of phytoplankton increased in direct relation to increasing eutrophication. Changes in the phytoplankton species composition (i.e. dominance of inedible species) may be the initial mechanism by which rates of zooplankton production slow down, but persistence of cyanophytes may be explained by a rapid increase in available nutrients to these opportunistic algal species. In other words, above a certain nutrient level or threshold, the process of zooplankton grazing (or settling of algae) is irrelevant. As nutrient levels continue to rise, phytoplankton growth continues to increase rapidly now controlled mainly by light (Jorgensen and de Bernardi, 1998).

2.1.1.3 Low Zooplankton Grazing Pressure: The eventual loss of large-bodied grazers is a possible outcome if algal-inedibility is a critical limitation to large-bodied zooplankton. Low zooplankton populations (accompanied by low grazing pressure) can also occur due to heavy predation pressure from zooplanktivores. As zooplanktivorous fish are largely size-selective predators, the large-bodied zooplankton are the targeted prey species and the first to decline (Gohen, 1990) when fish populations increase. The often-observed state of low zooplankton populations in eutrophied systems housing an overabundance of planktivores may serve equally with the internal dynamics of phytoplankton to stabilize the turbid-water state. Timms and Moss (1984) suggest that a continuous factor is more likely to regulate system state than is the dominance of phytoplankton, which can vary spatially or temporally. They favour predation on zooplankton by planktivores as a likely candidate. It follows, therefore, when predation on planktivores and grazing pressures are high, the clear-water state is favoured. Moss (1984) and Beklioglu and Moss (1996b) demonstrated that clear-water, plant-dominated systems might not necessarily switch to turbid ones following fertilization, providing large zooplankton grazers also persist.

A more recent study, conducted by de Bernardi and Guissani (1992), contradicts suppositions on cyanophyte inedibility and renews interest in the relationship between blue-greens and zooplankton. Their study involved enclosure experiments that either excluded or included *Daphnia obtusa* and *Bosmina longispina*, observing that blue-green algae populations grew to four times their initial densities in the absence of zooplankton as compared to the near total disappearance of blue-greens (*Oscillatoria rubescens* and *Lyngbya limnetica*) in the presence of the grazers. This strongly suggests that grazers can eat blue-green algae and thrive, at least under the experimental conditions provided. However, the study did not indicate the nutrient regime of the in/exclosures which is important to estimating the relevance of interspecific phytoplankton competition or perhaps other factors (N: P or light) relating to eutrophication that could underscore the zooplankton-phytoplankton dynamic. There was also no experimental account for bacteria-mediated consumption of net plankton. Jeppesen *et al.* (1997) hypothesize that large *Daphnia* sp. can graze on seston when the phytoplankton population is not suitable and thus avoid starvation. This may not be a long-term solution for *Daphnia* because seston may not afford all the nutrient benefits of algae and other phytoplankton, but it may allow a sparse population of *Daphnia* to persist until resources once again can sustain larger populations.

2.1.1.4 Physical Disturbance/Anchoring of Sediments: Up to this point, the turbid-water state has been assumed to result in over-abundant phytoplankton clouding the water column, particularly blue-green algae. However, an important reason shallow waters become turbid is wind-wave action that causes the resuspension of fine, flocculent, nutrient-rich sediments (Hosper and Jagtman, 1990). This effect can be exacerbated by the general lack of rooted macrophytes in hypereutrophic lakes. However, in cases of increasing macrophyte density and abundance, there is an accompanying increase in sedimentation rate by

decreasing the physical influence of water turbulence on loose, nutrient-rich sediments, effectively reducing the available phosphorus for algal growth and favouring a clear-water state (Meijer and Houser, 1997; Beklioglu and Moss, 1996b; Boers *et al.*, 1991; Moss, 1990; Balls *et al.*, 1989). While this is an ongoing, self-perpetuating process, algal colonies that form significantly before seasonal macrophyte growth (summer) may not be greatly affected by such a mechanism.

A secondary consideration for turbid-water stability revolves around strong positive feedback between sediment-induced and algal-induced turbidity from benthivorous fish, such as carp, catfish, bream or roach, fish that are generally abundant in warm, eutrophied waters. These benthivorous fish have feeding patterns that could force homeostasis on the turbid water state through continual stirring-up of loose littoral sediments rich in nutrients which either muddy the water directly or favour blue-green algal dominance (Houser and Jagtman, 1990; Kairesalo *et al.*, 1999), for reasons discussed in previous sections (e.g. competition for light and nutrients). Other benthivores, such as adult sunfish and bass do not disturb bottom sediments to the same degree as the fish species indicated above. Sunfish and bass "pick" the sediments for invertebrates as opposed to "stirring" the sediments. Therefore decisions to remove benthivorous fish should take specific feeding "styles" into account as well as whether the species is native or non-native to the system. Waterfowl can also cause turbidity as they up-rooted young shoots for food and "paddle" in shallow waters.

Three conclusions seem justified: 1) Competition for light eventually becomes the limiting factor for algal populations under highly eutrophic conditions; 2) Under some nutrient levels, zooplankton may be able to control algal populations through grazing; 3) Wind and wave action may be primary actions working to maintain the turbid state in shallow waters in the absence of limnetic phytoplankton, augmented or not by the effects of benthivorous fish.

2.1.2 Clear-Water Stability and Macrophytes: Current theory attributes clear-water states to two main stabilizing mechanisms: macrophyte function and predation/grazing pressures. It has been suggested that the main ecological "function" of macrophytes is stabilization of the clear-water state through i) competition with phytoplankton, ii) stabilization of flocculent sediments, iii) control of nutrient dynamics, iv) allelopathy, and v) providing refuge habitat for large-bodied zooplankton (Timms and Moss, 1984; Moss, 1990; Houser and Jagtman, 1990; Beklioglu and Moss, 1996b). Functions (i) and (ii) have already been addressed in conjunction with turbid-water stabilization.

2.1.2.1 Governing Nutrient Dynamics: Nutrient dynamics within a vegetated water body behave differently than in a non-vegetated or algal dominated lake. Submerged macrophyte beds appear to behave as sinks for nutrients during the crucial growing season for algae (spring/late summer), potentially becoming a significant competitor (Ozimek *et al.*, 1990; Boers *et al.*, 1990). In a paper by Crowder and Painter (1991),

organic C and P in sediments were positively correlated with plant cover at 42 sites within a water column depth of 0.5m in the Bay of Quinte. The study further suggested that C and P are added to the sediment by the plants rather than being the pre-existing condition favouring macrophyte establishment in the first place. The net effect of macrophyte influence on nutrient cycling is that there are fewer nutrients available for algal growth.

2.1.2.2 Allelopathy: Allelopathy has occasionally been identified as an algal control mechanism. Certain macrophytes, particularly *Characean* species, secrete toxins that appear to target phytoplankton (Wium-Andersen *et al.*, 1982; Beklioglu and Moss, 1996b; Ozimek *et al.*, 1990, Balls *et al.*, 1989). In the case of Lake Bleiswijkse Zoom, The Netherlands, the sustained control of algal growth from the period of July to November is credited to the rapid colonization by submerged macrophytes due to nutrient competition for nitrogen and possibly to allelopathy by *Characean* species (Meijer *et al.*, 1989).

2.1.2.3 Refugia: A final hypothesis of macrophyte-mediated clear-water is the physical complexity of plant beds that afford refuge to both predators and prey. The importance of this effect depends on the magnitude of zooplankton grazing pressure for algal suppression and water clarity promotion. Dense macrophyte beds can harbour large-bodied zooplankton from planktivory, while at the same time providing lurking habitat for piscivore species like pike that prey on planktivores, reducing predation pressure on large bodied zooplankton and increasing grazing on algae (Moss, 1990; Hosper and Jagtman, 1990; Carvallo, 1994). Several authors concur with the premise of a macrophyte predation refuge, but stress that the actual mechanisms for maintaining effective grazers are complicated (Harper, 1992; Meijer *et al.*, 1989; Meijer and Hosper, 1997).

An appropriately vegetated littoral area is well documented as the site for pike spawning as well as offering protection from intraspecies predation for young-of-the-year pike (Grimm, 1989; Hosper and Jagtman, 1990; Beklioglu and Moss, 1996). Shapiro (1990) and Hosper and Jagtman (1990) suggest littoral macrophyte refugia offer the only option for large-bodied zooplankton survival after biomanipulation due to physical hindrance of predators and provision of low light conditions less suited to visually hunting planktivores. A paper by Phillips *et al.* (1999), reported that plant bed density was negatively correlated with planktivorous fish abundance and positively correlated with cladoceran grazer abundance, strongly supporting the idea that dense plant beds provide a refuge from fish predation. In Hudson's Bay, England, established stands of water lilies (*Nuphar lutea L.* and *Nymphaea alba L.*) were postulated as the key to maintaining clear-water and minimal cyanophyte growth, despite high nutrients and the presence of zooplanktivorous fish (Timms and Moss, 1984). The lilies provided a refuge for grazing zooplankton and/or encouraged behaviour patterns in zooplankton that confined them to the lilies by day and/or stimulated movement by twilight. Water lilies are also documented as the favoured "hiding spot" for large *Daphnia spp.* in Little Mere, England, another water body with good water clarity, planktivores and

relatively high nutrient concentrations (Beklioglu and Moss, 1996b). In shallow lakes, macrophytes may thus be considered a refuge for pelagic cladocerans, comparable to vertical migration in deep lakes (Jeppesen *et al.*, 1997).

As a final note, Shaprio (1990) cautions that no matter the type of refuge, its loss appears to promote algal abundance levels at equal or greater than pre-biomanipulation conditions. The predation refuge hypothesis has been shown to backfire if the plant beds also harbour zooplanktivores. In the case of Hoveton Great Broad, *Polyphemus pediculus* (an invertebrate predator) lives within plant stands and preys on cladocerans (Timms and Moss, 1984). Therefore, care must also be taken to properly monitor extant pond populations.

The main conclusion for clear-water stabilization is the importance of well-established, abundant macrophyte communities. These are integral to stabilizing the clear water state by directly limiting nutrient availability to phytoplankton (bottom-up), indirectly supporting continuous piscivore predation/zooplankton grazing pressures (top-down control) and reducing the impact of physical disturbance through sediment stabilization.

2.1.3 Destabilization of System State: Timms and Moss (1984) concluded that loss of either state might occur through extreme changes in trophic levels. Past lake research indicates that the turbid-water state can be de-stabilized through a rapid removal of planktivorous fish (Hosper and Jagtman, 1990; Shapiro, 1990). However, more recent debate challenges the longevity of such a shift if planktivores are allowed to return. As for the clear-water state, discussions so far suggest a vulnerability to "high" nutrient loading and the rapid loss of established macrophyte beds. These are often observed together as symptoms or causes of eutrophication, but it seems reasonable that destabilizing an established plant bed requires more than just high nutrient loading. Rapid macrophyte loss can occur for a variety of reasons, including mechanical harvesting, herbicides, propeller damage, poor light availability (due to competition with phytoplankton and wind/wave action re-suspending sediments), hydrological fluctuations, grazing by mammals or waterfowl and uprooting by benthivorous fish (Blindow, 1992). However, from a nutrient loading perspective, rapid macrophyte loss is tied to rapid epiphyte growth (Hosper and Jagtman, 1990; Carvalho, 1994).

Epiphytes are algae that utilize macrophyte stems and leaves as a growth substrate. As identified in the previous section 2.1.1.1, under normal growing conditions vascular plants have developed strategies for handling epiphytic shading. However, high and rapid nutrient loading introduces a shift in the ecological balance in favour of fast-growing epiphytes that quickly take advantage of the surge in available nutrients and surface area and render rooted macrophyte "shading defenses" ineffective over only a few growing seasons (Ozimek *et al.*, 1990). Poorly rooted or floating macrophyte species that, in part, exploit nutrients in the water, are also likely to benefit from increased nutrient supply, but increasing epiphytic growth will

reduce the light intensity available to these macrophytes so that a greater portion of fixed energy is diverted to maintenance and less to growth (Phillips *et al.*, 1978). As macrophytes decline, epiphytes lose a spatially valuable growing substrate and free-floating phytoplankton (e.g. cyanophytes) eventually displace or dominate the community. The rise of cyanophyte dominance occurs only subsequent to the epiphyte-macrophyte competition (Phillips, *et al.*, 1978; Timms and Moss, 1984; Ozimek *et al.*, 1990) and may persist even if nutrients reach a steady state. Free-floating phytoplankton, including blue-greens, exploit a niche that is opened up by the decline of macrophytes, they do not appear to drive macrophyte decline. Phillips *et al.* (1978) suggests that a reduction of allelopathic secretions by macrophytes, together with less competition for nutrients by epiphytes, allow for the growth of free-floating phytoplankton.

2.2 Biomanipulation

Attempts at reversing cultural eutrophication have been underway for several decades, with much of the basic research conducted in Western European countries and North America. In the early 1980's, a technique that had been studied and practiced for several decades in countries like The Netherlands and Denmark was given the North American title of "biomanipulation" (Shapiro, 1990). Biomanipulation, also known as top-down control or trophic cascade, generally refers to the reduction of planktivorous and benthivorous fish biomass meant to accelerate the recovery process of a eutrophied lake (Meijer *et al.*, 1989; Gophen, 1990; Boers *et al.*, 1991). As with alternate stable-state existence, biomanipulation strongly depends on the themes of predation, grazing pressure by and nutrient competition between members of the aquatic ecosystem, ultimately leading to improved water quality (DeMelo *et al.*, 1992).

In more recent years, biomanipulation has come to include broader operational mechanisms and ecological goals. The classic idea of top-down-control still demands attention, but other organisms in the food-web have been identified as players, including aquatic plant communities and the habitat complexity they provide (Hosper and Jagtman, 1990; Ozimek *et al.*, 1990; Beklioglu and Moss, 1996b). In addition, presumed species interactions are no longer fixed to the unidirectional top-down cascade, lending voice to the complexity of zooplankton-phytoplankton relationships that may exert equally important lateral control or multi-directional forces. Harper (1992) now defines biomanipulation as the deliberate alteration of one component of the food chain to promote changes in other components. Further discussions in this chapter assume this definition of biomanipulation, unless otherwise stated. Although alleviating the symptoms of eutrophication is still a common goal of biomanipulation (Hosper and Jagtman, 1990; Harper, 1992), a more comprehensive and ecologically responsible goal has made the list; that of promoting a diverse biological community (Perrow *et al.*, 1997).

2.3 The Aquatic Food-Web and Biomanipulation:

Food webs or food chains are visually attractive terms describing the network of consumer-resource interactions among species within an ecosystem. In general, aquatic food-webs consist of four or five trophic levels, with the lower levels comprised of producers and primary consumers that are more efficient at energy assimilation and conversion than higher, predatory organisms (Pimm, 1982; Polis and Winemiller, 1995). Food-web dynamics are the classic limnological approach to understanding population control through mechanisms of production (direct assimilation of nutrients) and consumption (grazing and predation) (Carpenter and Kitchell, 1992). Historically, population regulation within the aquatic community was considered unidirectional, that is, from nutrients to fish (Gophen, 1990) i.e. bottom-up control. Present theory has broadened to incorporate the opposite perspective, with proponents like S. R. Carpenter and B. Moss, arguing strongly for the role of top-down control. It was from this paradigm that the original concept of biomanipulation arose (Vermaat *et al.*, 1990; Moss, 1990). A third consideration, identified earlier in discussions of alternate stable state theory, is lateral control through intra-trophic-level competition (i.e. macrophyte-phytoplankton).

Top-down effects describe the direct suppression of prey species by a predator on the adjacent, higher trophic level and the indirect enhancement of species two trophic levels below with impacts percolating through the whole food-web. For example, a Danish study of young-of-the-year pike (top predator) demonstrated the top down trophic cascade by the pike preying on hatchling bream (secondary predator), a typically voracious feed on pelagic zooplankton, and with the resultant shift in zooplankton community composition to larger bodied species (Jeppesen *et al.*, 1997). Associated changes at lower trophic levels and the lake environment included a decrease in both chlorophyll-*a* and total phosphorus and an increased Secchi depth. Of course, should the top piscivore "fail" to exert sufficient pressure on the secondary predator (planktivore), zooplankton body size and abundance will both decrease due to size-selective predation. The eventual effect of this cascade, as observed for decades in European lakes, is the enhancement of phytoplankton populations that benefit from lower and less efficacious grazing pressure (Harper, 1992). With North American lake management goals often including a reduction in algal biomass and increased water clarity as priorities, a failure of top-down control measures can be both politically and biologically serious.

The complexity of zooplankton and phytoplankton interactions makes the application of the top-down model somewhat problematic at these lower trophic levels. The management directive gleaned from earlier sections was that a high nutrient regime imposed limitations on effective top-down control of phytoplankton (i.e. grazing), but the presence of dense and abundant macrophyte beds might mitigate these limitations (i.e. competition with macrophytes for light, lurking habitat for some piscivore species and reduced planktivore predation). If, however, nutrient concentrations are low to moderate, planktivorous fish

predation remains the key top-down force indirectly regulating phytoplankton biomass, thus adhering more closely to conventional top-down control pathways. Harper (1992) contends that the effectiveness of large-bodied zooplankton grazers, such as Daphnids (> 1mm), is so high under favourable conditions (i.e. low fish predation) that they should be able to maintain standing crops of algae below the carrying capacity of a lake as defined by its nutrient status. Jorgensen and de Bernardi (1998) agree that when planktivore predation is negligible, zooplankton grazing can control phytoplankton biomass.

Perhaps a more complete view of food-web dynamics is one that confers importance to both top-down and bottom-up control. Meijer *et al.* (1989) subscribe to this dual view, stating that large zooplankters can control algal biomass through grazing (top-down cascade) and that the removal of benthivorous fish, e.g. bream and carp, will reduce nutrient release from the sediment and thereby also help control algal growth (bottom-up cascade). These hypotheses are largely based on the results of biomanipulations of small, eutrophied ponds where all planktivores and benthivores were removed. However, the paper does not provide insight as to how long the acquired clear-water results lasted. One of the experimental systems cited by Meijer *et al.* (1989) was Round Lake, Washington (Shapiro and Wright, 1984), but in a later paper Round Lake had reverted to its turbid-state two years after biomanipulation (Gophen, 1990). An important point is that macrophyte beds were not replaced in the Round Lake experiment. An ideal food-web dynamic theory, as it pertains to biomanipulation, then follows this suggested pattern: significant planktivore/benthivore removal leads to increased zooplankton grazing (top down), decreased sediment resuspension (bottom-up), clearer water and a reappearance of submerged macrophyte dominance (Meijer and Hoesper, 1997).

Clearly, trophic interactions are now understood to be more complex and flexible than previously thought, with controlling forces exerted both from the nutrient level on up and the piscivore level on down. More detailed study of the zooplankton-phytoplankton relationships suggests even more complex interactions (particularly with blue-greens) that require physicochemical/biochemical considerations once thought minor. It is presently impossible to generalize to all blue-greens and all zooplankton situations (de Bernardi and Guissani, 1990).

2.4 Biomanipulation Techniques

2.4.1 Traditional Techniques: The most common biomanipulation technique is still piscivore stocking or reduction/removal of planktivores/benthivores to exert top-down control (Perrow *et al.*, 1997; Jeppensen *et al.*, 1997; Harper 1992; Shapiro, 1990). Catch restrictions on piscivores and chemically induced fish kills using agents like rotenone, are considered in-line with traditional techniques (Shapiro, 1990). Prior to

implementing any of these techniques, there are site, management and operational considerations that, if ignored, may negatively affect the outcome of a project.

From a management perspective, the cost of stocking is generally less than removal (Jagtman, 1990) and has been shown to be more economically feasible than engineered techniques (e.g. dredging, sediment capping, hypolimnetic treatments) (Sondergaard *et al.*, 1997; Kairsealo *et al.*, 1999). The participation of universities or appropriate government agencies could offset labour costs associated with seining or draining a lake to remove fish. The use of chemical agents to kill fish may avoid intense labour expenses, but may not be ecologically, socially or politically compatible with lake-user ideology (Shapiro, 1990). Similarly, differing political and social interests must be taken into consideration, as either or both of these factions can deem the removal of fish on a large scale unacceptable (Donabaum *et al.*, 1999). Among site-selection considerations, the candidate lake must be isolated from other water bodies, less a reduction in the fish populations be undermined by the premature introduction of populations from connected lakes or streams (Hosper and Jagtman, 1990). Also, lakes with $<50 \text{ kg ha}^{-1}$ of total fish stock (planktivores and benthivores) prior to reductions have a poorer chance at a timely shift to a clear-water state as compared to lakes with greater fish biomass (Hosper and Jagtman, 1990).

This suggests that biomanipulation needs to be functionally drastic to incite change. Arguably, then, one of the most important operational considerations is how drastically do fish populations need to be altered to make a functional difference? Conservative estimates suggest that 70 - 75% of planktivores need to be removed (Perrow *et al.*, 1997; Meijer and Hosper, 1997). In shallow Lake Wolderwijd/Nuldernauw, The Netherlands, this portion roughly translated to planktivore reductions to 20 kg ha^{-1} and benthivores to 25 kg ha^{-1} (Meijer and Hosper, 1997). Although earlier manipulations of this same lake reduced benthivores on the order of 50 kg/ha (Boers *et al.*, 1991), heavy wind-induced turbidity was not taken into account (Meijer and Hosper, 1997). Meijer *et al.* (1989) removed 44 kg ha^{-1} of planktivorous and 59 kg ha^{-1} of benthivorous fish in Lake Bleiswijkse Zoom, equivalent to almost all planktivores and 85% of benthivores. Kairesalo *et al.* (1999) attributed a collapse in cyanobacterial blooms in Lake Vesijarvi to the removal of nearly 400 kg ha^{-1} of planktivores, describing the treatment as "a large-scale disturbance of the ecosystem" that was required after traditional restoration techniques had no effect on cyanobacteria blooms. Perrow *et al.* (1997) recommend that fish removal be conducted in the winter or early spring, before ice melt. This allows fish capture to concentrate on areas of naturally high fish density (bottom waters) and capitalizes on the new conditions (i.e. low predation) coinciding with peak daphnia populations and clear water (pre-plankton growth). However, the authors caution that repeat manipulations should be anticipated if macrophytes do not colonize within the subsequent growing season.

Perrow *et al.* (1997) go on to suggest that the stocking of piscivores is an important but secondary technique, providing that the stocked fish density is sufficiently high. The study identified several Danish lakes that showed no functional response to pike stocking between 200 - 400 fish ha⁻¹, but positive results were noted in Lake Wirbel, Poland and Lake Lyng, Denmark when piscivores were stocked at numbers up to 3600 fish ha⁻¹.

Another biomanipulation technique that is less often applied, but still fits into the conventional top-down approach, is direct enhancement of zooplankton populations (Shapiro, 1990). Zooplankton are enhanced through the provision of an artificial predation refuge. Artificial predation refugia are classified as either "open" or "closed". Open ones resemble the physical structure of plants and are open to fish use as well; closed refugia are like cages that have mesh sizes to permit the free flow of water and zooplankton but exclude fish (Moss, 1990).

2.4.2 Alternative Technique: An alternate bio-technique called habitat enhancement (Shapiro, 1990) is not new, but with the recognition of macrophyte importance to system stabilization, it is receiving greater attention in the literature. The main component of habitat enhancement is macrophyte restoration/recolonization/selective removal in the euphotic zone. Ozimek *et al.* (1990) contends that the loss of macrophytes due to degradation affects the aquatic community at multiple levels, either directly or indirectly, and therefore lake restoration projects should include a component of macrophyte restoration. Galanti *et al.* (1990) explained that macrophyte alteration belongs within a broader realm of biomanipulation techniques since their selective removal from the littoral area reduces planktivore spawning sites and young-of-the-year predation refuge and therefore the recruitment of planktivores decreases. This decrease is the main thrust of classic biomanipulation. Similarly, the restoration or recolonization of macrophytes should also qualify as biomanipulation on the basis of providing a refuge for zooplankton and lurking habitat for piscivores, the two trophic levels that most influence planktivore abundance. Further, although benthivore abundance may not necessarily be affected, the sediment stabilization attributed to rooted macrophytes works to minimize benthivore influences. One could argue that macrophyte restoration achieves the same results as planktivore and possibly benthivore reductions, but follows a different functional pathway.

2.5 Goals and Objectives of Biomanipulation

So far I have identified, three main goals of biomanipulation from the literature, 1) quantitative reduction of planktivores and benthivores that, through feeding behaviour, disturb sediments to a high degree, 2) alleviating the more visual symptoms of eutrophication and 3) promoting biological diversity. Rather surprisingly, what is not in the literature is a goal of restoring community structure and ecological function.

Perhaps, this concept is actually embedded somewhere and intrinsic to the other goals, but it deserves a higher profile.

Unlike structure and function, system stability has figured prominently and explicitly as a desired feature of post-biomanipulation. The idea of stability carries the connotation of long-term, predictable relationships. As Moss (1990) has stated, biomanipulation results have to last. It can be concluded from section 2.1 that such lasting relationships require a functional dynamic to be set in motion to preserve and buffer stability, whether turbid or clear. Although a turbid state is stable, the complement of species is generally less than found in a clear-water state. Diversity is lowered and certain species-specific functions may not be performed (e.g. macrophyte-mediated sediment stability and refugia or piscivore predation pressure) due to either total absence of the "player" or remnant populations too small to influence the system's state. The dominant organisms in a turbid state are presumably functioning as their niche dictates, but it may be to the exclusion of or detriment to adjacent trophic levels. On the other hand, clear-water states appear to provide for a larger set of functional niches required to support a biologically diverse community, inclusive of those organisms that proliferate in a turbid system. It follows that, if biomanipulation is to result in a stable and biologically diverse system, then all the necessary biological functions and community structures that support such an ecosystem must also be (re) established.

It is quite common in the literature to have biomanipulation goals and objectives listed quantitatively. For example, the goals of planktivore and benthivore reductions were listed as percentages or ratios in the previous section. Jorgensen and de Bernardi (1998) referred to total phosphorus objective of less than $50 \mu\text{g L}^{-1}$ as signifying success and total phosphorus above $100 \mu\text{g L}^{-1}$ as failure when associated with large standing stocks of grazing resistant phytoplankton and dominant or near dominant populations of planktivorous fish. McQueen (1990) identifies a reduction of chlorophyll-*a* from $>150 \mu\text{g L}^{-1}$ to $50 \mu\text{g L}^{-1}$ as the main objective for some lakes while other areas in the world aim for reductions from $8 \mu\text{g}$ to $2 \mu\text{g L}^{-1}$. Less precisely stated objectives are still amenable to being quantified, such as the attainment of clear water or higher N: P ratios (Harper, 1992; Gohen, 1990; di Bernardi and Guissani, 1990; Meijer *et al.*, 1989). All of these measures support an overall objective of improved water quality, which feeds directly into the main goal of alleviating the symptoms of eutrophication. A critical review by DeMelo *et al.* (1992) used only quantitative objectives to compare biomanipulation results: fish densities or biomass, invertebrate densities, total zooplankton biomass or density or body size, total phytoplankton biomass or cell volume OR chlorophyll-*a* or primary production, Secchi depth, P and N concentrations and pH. Eventually, the number of objectives was reduced to abundance of piscivores, vertebrate planktivores, zooplankton and phytoplankton to better establish consensuality between different research groups.

There is difficulty in separating biomanipulation goals based on structural versus functional definitions. Structural goals alter existing biotic communities in an attempt to normalize existing functions (e.g.

increase zooplankton populations; increase grazing pressure). Functional goals require one or more species, plant or animal, to be re-introduced, thereby reinstating their associated ecological function (e.g. reestablish macrophytes; return of littoral habitat, competition with algae, sediment anchoring, etc.). There is definite overlap, with a return to a functional balance as the common ground. The tendency to break down larger, holistic goals into more quantifiable components may not be as appropriate for structural or functional considerations as compared to goals related to biological diversity or eutrophication. For example, lake management objectives in The Netherlands encompass the whole-lake perspective with specific reference to structure and function as comprehensive goals (Hosper and Jagtman, 1990). Moss (1999) contends, "Unless function... is understood, there cannot be sensible management nor predictable restoration". Many studies in the literature identify a main goal of biomanipulation as increasing zooplankton abundance (structural) to promote heavy grazing on over-abundant algal species (functional), but do not specify an amount of "increase" (Jorgensen and de Bernardi, 1998; Sondergaard *et al.*, 1997; de Bernardi and Guissani, 1990; Ozimek *et al.*, 1990). This may be because appropriate numbers of zooplankton, as a management tool, have not been identified.

Population or biomass increases and decreases can be quantified as can grazing rates, but what is really being evaluated is whether or not zooplankton *function* has been restored or enhanced (i.e. phytoplankton population control). There is also a growing trend in the literature towards re-establishing submerged macrophyte species as a main goal of biomanipulation (Gohen, 1990; Beklioglu and Moss, 1996; Perrow *et al.*, 1997). This goal can technically be measured through percent cover or number of plant species, however, these quantitative results do not report on whether macrophyte function (e.g. sediment anchoring, refugia, nutrient cycling) has been returned to the system, which is the reason researchers advocate macrophyte restoration.

Very general biomanipulation goals include "kick-starting" the system from one stable state to another or "forcing" the lake to switch from the turbid-water to the clear-water state (Hosper and Jagtman, 1990). Both of these statements are large enough in meaning to encompass all the other goals, and for the same reason not particularly useful on an applied level. Goals and objectives are needed to evaluate the absolute and relative success of a project. Measuring biomanipulation goals quantitatively may be necessary to help compensate for the uniqueness of each lake project, allowing for more meaningful comparison between lakes. However, McQueen's example of two differing chlorophyll-*a* objectives demonstrates the present lack of consistent international standards that, unless rectified, will reduce the usefulness of quantified goals between lake systems. Perhaps there is wisdom in initially bracketing biomanipulation success within the broader terms biological structure and function, and then considering the system part-by-part as quantified parameters become available and/or are standardized.

2.6 Maintaining the Desired Results of Biomanipulation

There are numerous examples of biomanipulated lakes that show initial change, but then return to pre-manipulated states within one or two years. As mentioned earlier, Round Lake, Washington results lasted only two years. It is postulated, introduced northern pike did not have proper macrophyte spawning habitat to maintain populations sufficiently high to exert predation pressure on planktivores (Shapiro, 1990). Likely for the same reasons, clear-water results in Lake Lyng, Denmark lasted only for the duration of the season the work was undertaken, despite 6 years of northern pike stocking (Sondergaard *et al.*, 1997). Immigration by extra-system fish countered the initial achievement of clear water and returned Lake Helgetjern to its turbid state (Perrow *et al.*, 1997). In all cases, more thoughtful planning might have produced longer lasting results. The Round Lake and Lake Lyng projects fell short of achieving the important goal of ecosystem function by not considering the need to re-instate macrophytes. The site selection criterion of a closed-system was not adhered to for Lake Helgetjern. How much longer might these lakes have remained "manipulated" had these measures been taken? It is difficult to speculate, but Jeppesen *et al.* (1997) contends that changes made to either plant or fish stock abundance must be sustained, or else feed back will return the system to its original turbid state. According to McQueen (1990), biomanipulated lakes need continual management (harvesting) of planktivore populations to maintain target populations of large-bodied zooplankton, as stocked piscivore populations are unable to do this maintenance function alone. If this is true, then lake managers can never hope to attain permanency in a biomanipulated system. I suggest a two-part solution, which may help propel biomanipulation strategies through the stability threshold and keep them there.

As discussed previously, biomanipulation theory is based on top-down, trophic cascades, a by-product of food-web biology. Critics of biomanipulation argue that the true nature of food-web interactions does not, in fact, support the application of top-down biomanipulation. In a review of 50 biomanipulation papers, DeMelo *et al.* (1992) found a strong predator-prey relationship high up in the food chain (piscivore-planktivore-zooplankton) but at lower trophic levels, the relationship weakened. Similarly, in a review of 36 biomanipulation projects McQueen (1990) suggested that the effects of changes at the top of the food web decreased as they cascade down through to phytoplankton. This suggests any top-down manipulation can only be expected to influence a portion of the biotic community, leaving the system vulnerable to lower trophic forces that may or may not promote/streamline stability. Unpredicted food-web interactions have forestalled any general acceptance that biomanipulation can work. For example, the introduction of pike to trigger a trophic cascade in several Danish lakes failed, as cannibalism during winter months devastated the small, stocked pike populations (Jeppesen *et al.*, 1997). Many North American planktivores are large, deep-bodied centrarchids or clupeids that may not be susceptible to piscivore predation beyond a certain age class (Gophen, 1990).

Biomanipulation, among other things, aims to reduce dominant phytoplankton crops through 1) nutrient reduction, 2) grazing or 3) limiting light. The degree of influence of trophic interactions on phytoplankton abundance may not be determinable. DeMelo *et al.* (1992) concluded that research to date can not differentiate direct and indirect effects of nutrient additions, macrophyte competition and other physical-chemical factors from those of direct manipulation. The present limited knowledge of food-web interactions at the lower trophic levels sustains a degree of skepticism around the practice of biomanipulation on highly eutrophied, algal-dominated systems.

2.6.1 Possible Solutions to Extend the Duration of Biomanipulation Results: Before attempting to define solutions, understanding the entire problem is important. Biomanipulation is no exception, especially since multiple causes can have the "same" effect. Nevertheless, macrophytes appear to be an important driver of system states and this is where I suggest rehabilitative efforts start.

2.6.1.1 Solution Part 1: A system may need priming for macrophyte regeneration or re-introduction. If nutrient levels are high ($>100 \mu\text{g TP L}^{-1}$), loadings have to be lowered and biological control over phytoplankton enhanced. In these circumstances, there is an overwhelming consensus in the literature that a combination of abatement of both external and internal loading (bottom-up) and classic biomanipulation (top-down) strategies be applied together (Moss, 1990; McQueen, 1990; Houser and Jagtman, 1990; Gophen, 1990; Agusti, 1992; Harper, 1992; Sondergaard *et al.*, 1997; Jeppesen *et al.*, 1997; Donabaum *et al.*, 1999). Biomanipulation alone is risky from a phytoplankton-control perspective. There are also cautions over using just nutrient abatement strategies. Early lake studies by Moss (1990) have demonstrated that nutrient abatement alone it is not likely to be able to shift the botanical community from phytoplankton to macrophytes without the use of biomanipulation. Kairesalo *et al.* (1999) reported that even after reducing the external phosphorus load of Lake Vesijarvi, Finland from $2.1 \text{ g P m}^{-2} \text{ year}^{-1}$ to $0.15 \text{ g P m}^{-2} \text{ year}^{-1}$, cyanobacterial blooms continued unabated. Recent macrophyte-centred investigations by Beklioglu and Moss (1996 and 1996b) found that nutrient control treatments alone have not generally resulted in significant phytoplankton biomass reduction or macrophyte regeneration in Britain. Houser and Jagtman (1990) agreed with the other authors, stating that nutrient reducing strategies on their own can not effectively compete with the in-place structure and functioning of a food chain operating within a phytoplankton dominated system. A slightly different argument, though no less compelling, is one that combines the supposition by Perrow *et al.* (1997) that many failures of biomanipulation are due to insufficient removal of planktivores ($<75\%$) with Gophen's (1990) concern that improving water quality through piscivore predation pressure alone may only occur if planktivore populations are first reduced to ecologically unacceptable numbers.

An important and obvious question is how low nutrient concentrations need to be reduced in anticipation of macrophyte growth? Numerous studies in Denmark suggest that long-term results can be achieved if nutrient abatement programs can reduce and stabilize total phosphorus concentrations to 50-100 $\mu\text{g L}^{-1}$ (Jeppesen *et al.*, 1997). Reductions into this range are considered realistic for at least some eutrophied urban ponds in Southern Ontario (Grenadier Pond and Lake Wilcox) that do not presently exceed 80 $\mu\text{g L}^{-1}$ mean annual total phosphorus. This level is low enough to allow substantial macrophyte growth in at least one study. Lago di Candia, Italy, is a stable clear-water, macrophyte dominated shallow lake with total phosphorus values ranging between 30-60 $\mu\text{g L}^{-1}$ (biomanipulation on Lago di Candia involved macrophyte harvesting) (Galanti *et al.*, 1990). Grenadier Pond has similar mean and maximum depths as Lago di Candia (see Table 1.0 in next section), although restorative measures in Grenadier Pond are not complete.

2.6.1.2 Solution Part 2: Once nutrient levels are acceptably low and sustainable, large-scale macrophyte restoration can be done if abundant enough macrophytes appear to be the key to maintaining a clear water state (Jeppesen *et al.*, 1997; Beklioglu and Moss, 1996). McQueen's (1990) review of 36 biomanipulated lakes showed that maintenance of the clear-water state in shallow lakes was achieved ONLY when macrophytes were restored. Meijer *et al.* (1989) reached similar conclusions from a study that compared stabilizing mechanisms in deep and shallow lakes. In two different studies, six years apart on Lake Wolderwijd/Nulderauw, macrophytes were reported as integral to achieving success. In the first study, credit was due to both increased grazing pressure and increased sedimentation of detrital phosphorus, a direct result of an increase in submerged, rooted macrophyte growth (*Chara spp.*) (Boers *et al.*, 1991). The second report's main recommendation for ensuring a stable clear-water state was to further the expansion of *Chara spp.* (Meijer *et al.*, 1997).

Accepting that macrophytes are necessary for system stability, the second requirement is adequate abundance, but how much macrophyte coverage is enough? Estimates for minimal plant coverage required to either promote or sustain the clear-water state over moderate to high nutrient conditions have been reported for several lake biomanipulation studies. Hosper and Jagtman (1990) considered a "substantial and sufficient" lake bottom area coverage (LBAC) by macrophytes to be 25% to support pike populations large enough (25 kg ha^{-1}) to control planktivore abundance. As mentioned in the previous section, Lago di Candia had an over-abundance of macrophytes and its plant coverage was reduced to 35% LBAC without loss of other biota or destabilization of the clear-water state (Galanti *et al.*, 1990). In Little Mere, England, the macrophyte coverage extended from 33% to 44% LBAC with nutrient abatement, and was identified as the system's stabilizing factor (Beklioglu and Moss, 1996). Meijer and Hosper (1997) suggest that lakes smaller than Lake Wolderwijd (26.65 km^2), should have 50% LBAC to achieve desired results. The range of 25% to 50% LBAC of macrophytes is based on shallow lake studies only, assuming appropriate littoral slopes and shores protected from excessive wind/wave action. Hosper and Jagtman (1990) and Galanti *et al.* (1990) stressed submergent species as the favoured type of macrophyte cover, while two other studies

showed promising results with floating plants, like nymphaeids in Little Mere (Beklioglu and Moss, 1996) and Hudson's Bay (Timms and Moss, 1984). The importance of plant diversity was not addressed in any of these studies, perhaps because many restoration projects result in abundance of only one or two rooted plant species (i.e. *Elodea spp.* and/or *Chara spp.*) (Moss, 1990).

According to theory, restoration of macrophytes capitalizes on their function as a zooplankton predation refuge. The effectiveness of macrophytes as a refuge depends on density in addition to areal plant coverage and abundance of young fish populations (Beklioglu and Moss, 1996b). Jeppesen *et al.* (1997) measured *Potamogeton spp.* density in terms of Percent Water-Column Volume Infested with macrophytes (PVI), determining that beds having less than 10-15 PVI did poorly as a littoral zooplankton refuge, even in the presence of few 0+ and 1+ aged planktivores. However, the authors go on to suggest that only 3% of lake coverage needs to be occupied by dense beds to achieve an increase in pelagic zooplankton numbers and therefore an increase in overall grazing pressure. Perrow *et al.* (1997) suggest a more conservative PVI estimate of 30.

This paper has identified the ecological challenges to aquatic system rehabilitation posed by high nutrient status. A review of the literature reveals a strong belief that a lake's trophic state may ultimately determine the lake's response to biomanipulation and explains the wide range of outcomes (Gophen, 1990; Beklioglu and Moss, 1996; Hosper and Jagtman, 1990; Jorgensen and de Bernardi, 1998). At the forefront of discussion should be whether conventional methods can realistically decrease total phosphorus concentrations of a hypereutrophic lake system ($>150 \mu\text{g L}^{-1}$) to the 50-100 $\mu\text{g L}^{-1}$ range suggested by Jeppesen *et al.* (1997). If not, is there any evidence that macrophytes maintain clear-water in hypereutrophic conditions? The following are three case studies that describe the environmental histories and factors leading to macrophyte dominance co-existing with such high levels of nutrients and zooplanktivorous fish.

Case 1. In Hudson's Bay, England, Timms and Moss (1984) found that despite high mean total phosphorus concentrations ($186 \mu\text{g L}^{-1}$), clear water was maintained and phytoplankton growth was limited by zooplankton grazing, in the presence of zooplanktivorous fish, when a zooplankton refuge was also present in the form of large, dense waterlily stands (*Nuphar lutea* and *Nymphaea alba*).

Case 2. Carvalho (1994) documented a drastic loss of zooplanktivorous fish following loss of dissolved oxygen related to sewage effluent loading in Little Mere, England, a clear-water, floating-plant-dominated shallow lake. The same study measured the extreme rise in total phosphorus concentration ($2484 \mu\text{g L}^{-1}$) due to the effluent, but the coincident loss of predation pressure led to the re-establishment of large-bodied and efficient zooplankton grazers and the avoidance of potentially large summer algal blooms. The pre-existing aquatic plants survived the high nutrient loading, but the small, isolated colonies did not expand.

A later study by Beklioglu and Moss (1996) monitored changes after the sewage effluent was diverted in 1992/93 (i.e. nutrient abatement). Total phosphorus levels significantly declined to $185 \mu\text{g L}^{-1}$ (although they still remained high by other lake standards), dissolved oxygen increased, planktivores were re-established, zooplankton shifted to smaller species, but the clear-water state remained stable. Credit for maintaining clear water, despite persistently high nutrients and a return of fish predation, was given to the pre-existing and expanded large macrophyte stands (*Nymphaeoid spp.*, *Potamogeton berchtoldii* and *Elodea canadensis*), which provided refuge for weed-associated zooplankton (Beklioglu and Moss, 1996, 1996b).

Case 3. Lake Zwemlust, provided a good case study of macrophyte restoration and subsequent maintenance of clear-water state through the removal of bottom-stirring fish and subsequent decrease in available nutrients. Following the removal of planktivorous and benthivorous fish from the turbid and hypereutrophic Lake Zwemlust ($400 \mu\text{g TP L}^{-1}$), the macrophytes *Nuphar lutea* and *Chara globularis* were introduced and natural colonization of five submerged species also occurred, covering almost 100% of the lake bottom within three years of treatment, an increase from no macrophytes prior to biomanipulation (Ozimek *et al.*, 1990). In addition to increased macrophyte abundance, the clear-water state returned initially due to zooplankton grazing, less sediment resuspension and then transient N-limitation induced by newly established macrophytes, which reduced phytoplankton growth and maintained the clear-water state for the following two years of the study (Ozimek *et al.*, 1990; Meijer *et al.*, 1989).

Macrophyte dominance at high nutrient levels is not widely accepted as predictably attainable (Perrow *et al.*, 1997), despite the examples of lakes that have managed to do just that (e.g. Hudson Bay, Little Mere, Lake Zwemlust, Lago di Candia). At present, the mechanisms of macrophyte dominance or decline at high nutrient concentrations are not clearly or commonly understood, although speculation and operational theories have been discussed throughout this paper. Moss (1990) concludes that macrophytes can exist over a range of nutrient concentrations providing a descriptive scale of very low-low-moderate-high, suggesting that it may be problematic for macrophytes to exist, let alone dominate at the hypereutrophic end of the nutrient spectrum.

2.7 Application of Biomanipulation to Shallow Lake Systems

For a variety of reasons, top-down control is considered by researchers to be more relevant and stronger in small-volume, shallow lakes than in larger deeper ones (Gophen, 1990; Carvalho, 1994; Jeppesen *et al.*, 1997). Moss (1990) presents a rather simple, but elegant argument for this opinion, by recognizing that macrophyte dominance is only possible in shallow systems as the fraction of littoral habitat is considerably greater in shallow systems.

Predation pressures, thought to be heavier in shallow lakes, are singled out as the main food-web based argument. Trophic interactions at the benthic-planktivore and zooplankton levels differ between shallow and deep systems in ways that increase or maintain the predation pressure on zooplankton in shallow lakes (Jeppesen *et al.*, 1997). Zooplankton prey are more vulnerable to predation in shallow than deep areas (littoral) as they cannot vertically migrate (Gophen, 1990). Many pre-biomanipulated lakes lack the littoral vegetation that could afford similar refuge for zooplankton prey, and therefore top-down control is maximized if initiated at the start of a project. The threat of fish predation can be sustained longer in shallow than in deep systems because in shallow ones benthic-planktivorous fish can maintain their populations independent of littoral zooplankton availability as they also feed on benthos whose biomass is higher in shallow than in deep lakes. Any subsequent shift from benthic invertebrate prey to pelagic zooplankton prey will have a greater impact on zooplankton populations in shallow than in deeper systems because the zooplankton: zoobenthos biomass ratio increases with depth (Jeppesen *et al.*, 1997). Also, filamentous cyanobacteria are particularly buoyant and less likely to be lost to the hypolimnion of deep lakes through settling than are the other algal taxa. Therefore, in deep lakes, algal species susceptible to sinking (e.g. diatoms) are physically removed from the competition with blue-greens whereas shallow lakes may provide a chance for species survival or co-dominance (Jeppesen *et al.*, 1997).

The paper by Jeppesen *et al.* (1997) is based on data from 233 Danish lakes, enclosure experiments, full-scale experiments and published empirical data. A total of seven lake projects were detailed in this paper. All of them showed positive, initial responses to biomanipulation and are of similar physical dimensions, allowing for comparison to this level of similarity. The surface area, mean and maximum depths (where available) of lakes profiled in this paper and another "successfully" biomanipulated lakes are provided in Table 2.0 for comparison to Grenadier Pond.

Table 2.0. Selected Characteristics of Shallow Urban Ponds

| Lake Name | Surface Area (ha) | Mean Depth (m) | Max. Depth (m) |
|---------------------|-------------------|----------------|----------------|
| 1. Round Lake | 12.6 | 2.9 | 10.5 |
| 2. Lake Bleiswijkse | 14.4 | 1.1 | NA |
| 3. Lake Zwemlust | 1.5 | 1.5 | 2.5 |
| 4. Lake Wolderwijld | 2665 | 1.5 | NA |
| 5. Little Mere | 2.8 | 0.7 | 2.6 |
| 6. Hudson's Bay | 1-120 | 1-2 | NA |
| 7. Lago di Candi | 149 | 3.8 | 7.7 |
| 8. Alte Donau | 160 | 2.3 | 6.8 |
| 9. Grenadier Pond | 18.3 | 3.0 | 6.0 |

NA = not available

2.8 Conclusions

A total of seven main conclusions and findings are provided regarding alternate stable state theory, biomanipulation theory, application and the important role of aquatic macrophytes.

1. Stabilization of the turbid-water state within a high nutrient regime is attributed to dominant populations of self-regulating phytoplankton favoured by low zooplankton populations and/or ineffective zooplankton grazers and continual resuspension of shallow, surficial sediments by wind and wave action which can be exacerbated by large populations of bottom-stirring fish (e.g. carp). All this combined with heavy planktivore predation pressure may prevent a shift to macrophyte dominance even if nutrients are significantly lowered.
2. Stabilization of the clear- water state is mainly attributed to well established and abundant macrophyte communities (rooted-submerged and floating) that directly limit nutrient availability to phytoplankton, indirectly support continuous piscivore-predation and grazing pressures beyond the high-nutrient barrier and reduce the impact of physical disturbance through sediment anchoring by roots.
3. For biomanipulation to take full advantage of alternate stable-state and food-web theories, both top-down and bottom-up dynamics must be considered, recognizing that trophic interaction complexity appears greatest at the zooplankton-phytoplankton levels.
4. The three traditional biomanipulation techniques are I) piscivore stocking, II) planktivore reduction/removal, and III) direct zooplankton population enhancement. An alternative bio-technique that is garnering greater support is habitat enhancement. In some cases, the removal of benthivores that stir-up sediments as they forage and/or are non-native to the system may also be targeted for removal.
5. There are four main goals of biomanipulation: I) quantitative reduction of planktivores and/or benthivores, II) alleviation of the symptoms of eutrophication, III) promotion of biological diversity and IV) a return of balanced community structure and ecological function.
6. A potential strategy for successful, long-term biomanipulation results involves a combined approach of nutrient abatement and biomanipulation applied in two stages. The first is priming the lake for macrophyte regeneration or re-introduction by lowering high nutrient levels to intermediate concentrations ($50 - 100 \mu\text{g TP L}^{-1}$) while simultaneously enhancing control over phytoplankton through increased grazing pressure and planktivore reduction. The second stage involves large-scale aquatic macrophyte restoration, with coverage somewhere in the range of 25% - 50% of the lakebed

area. Macrophyte species diversity is important, however successfully biomanipulated lakes in North America and Europe were dominated by two rooted submergent species (i.e. *Elodea spp.* and *Chara spp.*) and/or floating nymphaeids.

7. Shallow lakes are appropriate for attempting food-web based manipulations.

2.9 References

- Agusti, S., Duarte, C.M. and D.E. Canfield Jr., 1992. Self-regulation, bottom-up and top-down control of phytoplankton communities: a reply to the comment by Kamenir. In *Limnol. Oceanogr.* 37(3): 683-687
- Balls, H., Moss, B. and K. Irvine, 1989. The loss of submerged plants with eutrophication I. Experimental design, water chemistry, aquatic plant and phytoplankton biomass in experiments carried out in ponds in the Norfolk Broadlands. In *Freshwater Biology*, 22:71-89
- Beklioglu, M. and B. Moss, 1996. Existence of a macrophyte-dominated clear water state over a very wide range of nutrient concentrations in a small shallow lake. In *Hydrobiologia* 337: 93-106
- Beklioglu, M. and B. Moss, 1996b. Mesocosm experiments on the interaction of sediment influence, fish predation and aquatic plants with the structure of phytoplankton and zooplankton communities. In *Freshwater Biology* 36: 315-325
- Blindow, I., 1992. Long- and short-term dynamics of submerged macrophytes in two shallow eutrophic lakes . In *Freshwater Biology* 28:15-27
- Boers, P., Van Ballegooijen, L. and J. Uunk, 1991. Changes in phosphorus cycling in a shallow lake due to food web manipulations . In *Freshwater Biology* 25: 9-20
- Carmichael, W.W. and P.R. Gorham, 1977. Factors influencing the toxicity and animal susceptibility of *Anabaena flos-aquae* (Cyanophyta) blooms . In *J. Phycol.* 13: 97-101
- Carpenter, S.R. and R.C. Lathrop, 1999. Lake restoration: capabilities and needs. In *Hydrobiologia* 395/396 : 19-28
- Carpenter, S.R and J.F. Kitchell, 1992. Trophic Cascade and biomanipulation: Interface of research and management – A reply to the comment by DeMelo *et al.* In *Limnol. Oceanogr.* 37(1): 208-213
- Carvalho, L., 1994. Top-down control of phytoplankton in a shallow hypertrophic lake: Little Mere (England). In *Hydrobiologia* 275/276: 53-63
- Crowder, A. and Painter, D.S., 1991. Submerged macrophytes in Lake Ontario: Current Knowledge, Importance, Threats to Stability, and Needed Studies. In *Canadian Journal of Fisheries and Aquatic Science.* 48 (1539-1545).

De Bernardi R. and G. Giussani, 1990. Are blue-green algae suitable food for zooplankton? An overview. In *Hydrobiologia* 200/201: 29-41

De Melo, R. , France, R. and D. McQueen, 1992. Biomanipulation: Hit or myth? In *Limnol. Oceanogr.* 37(1): 192-207

Donabaum, K., Schagerl, M. and M. Dokulil, 1999. Integrated management to restore macrophyte domination. In *Hydrobiologia* 395/396: 87-97

Galanti, G., Guilizzoni, P. and V. Libera, 1990. Biomanipulation of Lago di Candia (Northern Italy): a three year experience of aquatic macrophyte management. In *Hydrobiologia* 200/201: 409-417

Gartner Lee Limited (GLL), 1995. *Recommmendations for the Rehabilitation of Grenadier Pond, Wendigo Creek and Associated Wetlands.* Consultants report for the Department of Parks and Recreation, City of Toronto, Ontario.

Gartner Lee Limited (GLL), 1996. *The Rehabilitation of Lake Wilcox.* Consultants report for the Town of Richmond Hill, Ontario.

Gophen, M., 1990. Biomanipulation: retrospective and future development. In *Hydrobiologia* 200/201: 1-11

Grimm, M.P., 1989. Northern pike (*Esox lucius* L.) and aquatic vegetation, tools in the management of fisheries and water quality in shallow waters. In *Hydrobiol. Bull.* 23: 59-65

Harper, D., 1992. *Eutrophication of Freshwater: principles, problems and restoration.* Chapman and Hall, 1st edition.

Hosper, S.H. and E. Jagtman, 1990. Biomanipulation additional to nutrient control for restoration of shallow lakes in The Nertherlands. In *Hydrobiologia* 200/201: 523-534

Jeppesen, E., Jensen, J.P., Sondergaard, M. and T. Lauridsen, 1997. Top-down control in freshwater lakes: the role of nutrient state, submerged macrophytes and water depth. In *Hydrobiologia* 342/343: 151-164

Jorgensen, S.E. and R. de Bernardi, 1998. The use of structural dynamic models to explain successes and failures of biomanipulation .In *Hydrobiologia* 379: 147-158

- Kairesalo, T., Laine, S., Luokkanen, E., Malinen, T., and J. Keto, 1999. Direct and indirect mechanisms behind successful biomanipulation. In *Hydrobiologia* 395/396:99-106
- Meijer, M.L. and H. Hoser, 1997. Effects of Biomanipulation on the large and shallow Lake Wolderwijd, The Netherlands. In *Hydrobiologia* 342/343: 335-349
- Meijer, M.L., Raat, A.J. and R.W. Doef, 1989. Restoration by Biomanipulation of Lake Bleiswijkse Zoom (The Netherlands): First Results. In *Hydrobiol. Bull.* 23: 49-57
- McQueen, D.J., 1990. Manipulating lake community structure: where do we go from here? In *Freshwater Biology* 23: 613-620
- Moss, B., 1999. Ecological challenges for lake management. In *Hydrobiologia* 395/396: 3-11
- Moss, B., 1990. Engineering and biological approaches to the restoration from eutrophication of shallow lakes in which aquatic plant communities are important components. In *Hydrobiologia* 200/201: 367-377
- Narumalani, S. Jensen, J.R. Althausen, J.D. Burkhalter, S. and H.E. Mackey Jr., 1994. Found at URL www.odyssey.maine.edu/gisweb/spatbd/acsm/ac94055.html
- Ozimek, T., Gulati, R.D. and E. van Donk, 1990. Can Macrophytes be Useful in Biomanipulation of Lakes? The Lake Zwemlust Example. In *Hydrobiologia* 200/201: 399-407
- Perrow, M., Meijer, M.L., Dawidowicz, P. and H. Coops, 1997. Biomanipulation in shallow lakes: state of the art. In *Hydrobiologia* 342/343: 355-365
- Phillips, G., Bramwell, A., Pitt, J., Stantsfield and M. Perrow, 1999. Practical application of 25 years' research into management of shallow lakes. In *Hydrobiologia*, 395/396 : 61-76
- Phillips, G.L., Eminson, D. and B. Moss, 1978. A Mechanism to Account for Macrophyte Decline in Progressively Eutrophicated Freshwaters. In *Aquatic Botany*, 4:103-126
- Pimm, S., 1982. *Food Webs.* Chapman and Hall
- Polis, G. and K. Winemiller (eds), 1995. *Food Webs and Integration of Pattern and Dynamics.* Chapman and Hall.

Shapiro, J., 1990. Bio-manipulation: the next phase - making it stable. In *Hydrobiologia* 200/201: 13-27

Shapiro, J. and D. I. Wright, 1984. Lake Restoration by bio-manipulation: Round Lake, Minnesota, the first two years. In *Freshwater Biology* 14: 371 -383

Sondergaard, M., Jeppesen, E. and S. Berg, 1997. Pike (*Esox lucius* L.) stocking as a bio-manipulation tool 2. Effects on lower trophic levels in Lake Lyng, Denmark. In *Hydrobiologia* 342/343: 319-325

Timms, R.M. and B. Moss, 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. In *Limnol. Oceanogr.* 29(3): 472-486

Vermaat, J.E., Hootsmans, M.J.M., and G.M. van Dijk, 1990. Ecosystem development in different types of littoral enclosures. In *Hydrobiologia* 200/201: 391-398

Wium-Andersen, S.; Anthoni, U.; Christophersen, C. and G. Houen, 1982. Allelopathic effects on phytoplankton by substances isolated from aquatic macrophytes (Charales). In *OIKOS* 39: 187-190.

Chapter 3 : Comparison of Littoral Sediment Cores with and without Macrophyte Growth in Biomanipulated Grenadier Pond

3.0 Introduction

The majority of aquatic macrophyte studies have centred on how sediment characteristics can affect aquatic macrophyte growth, diversity and distribution (Masden, *et al.*, 1993 Barko *et al.*, 1991; Barko and Smart, 1986; Carignan and Kalff, 1980). In this study, the interest is reversed: if and how the presence of rooted macrophytes is affecting the characteristics of the near-surface sediment of the littoral zone in shallow, productive Grenadier Pond, Canada. Macrophyte-related effects on sediment (with possible pond-wide implications) are of particular interest to both lake managers and researchers, as this study helps in the assessment of the value of re-vegetating the littoral areas as a nutrient control strategy.

The extent to which littoral zone dynamics govern nutrient cycling, a prominent issue in eutrophication reversal, is still under study, however it is at least in part a function of both macrophyte density and seasonal bed stability (Petticrew and Kalff, 1992). The macrophyte community is presumed to operate in a series of feedback loops (Chapter 2):

1. trapping of particulate sources of nutrients and organic material;
2. stabilizing of sediments and reduction of resuspension and transport to deeper, potentially anoxic regions of the lake (a driver for internal loading);
3. luxury uptake of nutrients (rendering them unavailable to phytoplankton, at least temporarily);
4. provision of predation refugia that protect large-bodied zooplankton, which graze on phytoplankton, from planktivorous fish; and
5. provision of predation refugia for young piscivores, who are important regulators of planktivores

In addition to anchoring sediments, stabilizing underwater slopes and reducing wind-turbulence at the sediment level (Petticrew and Kalff, 1992), macrophytes also function to provide habitat areas for invertebrates (benthic and plant-associated) and various age classes of fish (Masden *et al.*, 1993).

The efficacy of littoral zone retention of nutrient laden sediments and organic material has a temporal component, with functional contributions from macrophyte beds occurring in the growing season (spring through summer) (Kairesalo and Matilainen, 1994). Phosphorus removal from sediment, associated with

aquatic plant uptake, has also been strongly linked to the growing season. The magnitude of these potentially opposing processes (i.e. retention and regeneration) within a similar timeframe needs to be addressed in any experimental design in order to form an adequate hypothesis around plant-mediated nutrient levels. Smith and Adams (1986) demonstrated that seasonal uptake of phosphorus by rooted macrophytes (e.g. *Myriophyllum spp.*) was low in the summer months, and highest in early spring and fall, for both roots and shoots. Their study suggests that this cycle can be generalized to other species – on this basis I decided to sample in August, when macrophyte shoot density should be at a maximum and phosphorus sequestration intermediate.

Hypothesis Statements: If macrophytes are fulfilling their hypothesized ecological roles, then I expect: 1) higher sediment nutrient concentrations in vegetated sites – including remnant wetland stations, 2) higher organic matter content in vegetated sites, 3) more consolidated sediments in vegetated sites, and 4) a higher diversity and abundance of macrobenthic invertebrates in vegetated sites. An additional element in Grenadier Pond is the heavy salt loading associated with road salting practices. I expect sediment chloride concentrations to be highest in those stations closest to the storm water outfalls and Catfish Pond drainage but am unsure to what extent Cl⁻ loadings might affect macrophyte densities.

If physical re-establishment of macrophytes is accompanied by re-establishment of ecological function, the above parameters should also vary between newly planted macrophyte beds and long-established macrophyte stations. I expect newly planted sites to be more similar to established sites and less similar to non-vegetated sites.

3.1 Field Methods and Materials

3.1.1 Study Area Description and Location Map: See Chapter 1.

3.1.2 Littoral Habitat Types: In August 1998, three different littoral habitat types were identified around the pond based on presence/absence of rooted emergent vegetation and age class of present rooted emergent vegetation. Three habitat categories were assigned and are summarized in Table 3.1: established vegetation (EV) having > 40 years persistence; newly planted vegetation (NP) with one year or three years of growth; and non-vegetated sites (NV), having no significant emergent plant growth in over 40 years. The timeframe for EV and NV habitats was based on the last known shoreline alteration activity, which took place in 1955 and 1956 (Zimmerman *et al.*, 1986; Chapter 1) and a series of aerial photo interpretations. The timeframe for growth in NP habitats was based on plantings directed by lake management activity in 1995 and 1997.

Table 3.1 Descriptions of Habitat Types

| <i>Habitat Type</i> | <i>Station Name</i> | <i>Location on Pond</i> | <i>Description</i> |
|---------------------------------|---------------------|-------------------------|--|
| Established Vegetation | EV-1 | SW Shore | Remnant Wetland that includes the outfall from Catfish Pond. |
| | EV-2 | NW Shore | No known shoreline or littoral alteration; proximal influence from 2 storm water outfalls; natural ravine slope and vegetation (e.g. overhanging trees). |
| | EV-3 | NE Shore | No known shoreline or littoral alteration; proximal influence from Wendigo Creek outflow; paved path approximately 15m from water's edge. |
| Newly Planted Vegetation | NP-1 | SW Corner | Littoral zone and shoreline regraded and planted in 1997 (1 year growth); incorporates the pond outfall. |
| | NP-2 | East Cove Embayment | Littoral zone and shoreline regraded and planted in 1997 (1 year growth). |
| | NP-3 | East Central Shore | Littoral zone and shoreline regraded and planted in 1997 (1 year growth). |
| | NP-4 | SE Corner | Littoral zone and shoreline regraded and planted in 1995 (3 years growth). |
| Non-vegetated | NV-1 | East Shore | Littoral and shoreline plants removed in mid-1950's and replaced with hard, concrete edge and rip-rap stones; possible regrading; paved path approximately 15 m from water's edge. |
| | NV-2 | South Central Shore | Littoral and shoreline plants removed in mid-1950's and replaced with hard, concrete edge and rip-rap stones; possible regrading; gravel path adjacent to water's edge. |

3.1.3 Littoral Habitat Stations (LHS): Using the three littoral habitat types (EV, NP, NV), the entire pond perimeter and storm-water settling pond was "mapped". From this habitat map nine LHS were identified: three EV stations, four NP stations and two NV stations (Figure 3.1). Each LHS was set as a 60 m length, as this corresponded to the total lengths of the four NP stations. In cases where more than 60 m of

appropriate habitat was available (i.e. EV and NV stations), different shoreline orientations (e.g. west shore vs. east shore) and relative position to similar habitat was considered in an attempt to minimize bias.

3.1.4 Sediment Core Extraction Site Location: Cores were taken at 10 m intervals from each LHS. Six cores were taken in the majority of sites, but extremely hard and rocky sediments at NP-3, NV-1 and NV-2 reduced the number of successful extractions at these stations and a measuring error increased the number of cores from EV1. In the end, I had 21 cores from the three vegetated sites, 22 cores from the four newly planted sites and eight cores from the two non-vegetated sites. Originally, two criteria were used to preselect the location of sampling sites from the seven vegetated stations: 1) the interface between the plant bed and open water and 2) the maximum water depth associated with the photic zone (90 cm). Photic zone depth was assumed to correlate with the extent of rooted emergent plant growth and was determined by averaging Secchi depth readings across the period from 1985 to 1997 (Olden, 1998; GLL, 1996; Zimmerman *et al.*, 1986). However, upon inspection, rooted emergent plant growth in Grenadier Pond never extended beyond a water depth of 55 cm, so only the plant-open water interface was necessary to identify extraction sites (cores were taken just inside the plant bed). The average distance from the shoreline to core extraction sites was approximately 1.0 m. Under conditions where plant beds were not continuous and an "open area" coincided with a core extraction site, average distance from the shore was used to identify the core extraction site. This rule was also applied to the non-vegetated habitat stations.

I wanted a nested sampling design that incorporated four spatial scales ranging from within a core (smallest), between replicate cores, between stations of similar habitat and between different treatment sites (largest), which would permit a hierarchical, two-factor nested analysis of variance (Morrisey *et al.*, 1992).

A final note pertaining to core extraction sites is that in September 1997, the modified weir was installed and put into operation, lowering the 1998 summer pond level by 20 cm. This effectively meant that the littoral zone extended farther into the pond than in recent decades. This meant that some aquatic plant beds were actually stranded "onshore" with their associated sediments exposed to the air during the summer sampling. Therefore these areas were excluded from the study, as potential influences on air-exposed sediment were not considered comparable to those expected for flooded sediment.

3.1.5 Sediment Core Extraction: Sediment cores were taken with plastic corers, 4.8 cm inner diameter. The top 5 cm of sediment was extruded with a piston into a plexiglass calibration tube of the same diameter. The extruded core was sliced horizontally and then, by inserting a blade into the calibration tube,

and sliced vertically. Effort was made to keep the integrity of the core halves, but sediment consistency sometimes precluded a perfectly clean cut. Both halves were carefully transferred to separate, pre-labeled whirl paks – one for benthic invertebrate analysis the other for chemical/physical analyses. The remaining, deeper core section (5-15 cm) was also transferred to a whirl pak. All samples were kept on ice until return to the laboratory where they were stored in the dark at 4 °C until analysis. Kahle's solution (includes 10% formalin) was added to whirl paks to preserve samples destined for benthic invertebrate identification.

While in the field the following sub-sample site characteristics were measured and recorded: core sediment texture and colour, number of emergent, submergent and floating macrophyte stems within a 0.25m² quadrat (encompassing the core extraction site), littoral slope, width of plant bed from shoreline to open water and water depth at core extraction site. Macrophytes were identified in the field where possible, with samples collected when necessary. Photographs of all core-sampling sites were also taken (selected photographs in Appendix I).

3.2 Laboratory Methods

3.2.1 Physical and Chemical Parameters: The upper physico-chemical core samples were homogenized within the whirl paks, with the majority used for water content analysis. These samples were transferred to a weighing dish, weighed and then oven dried at 105°C for 24h to determine water content as a percentage of wet mass (%SAT) (Murdoch *et al.*, 1997). Approximately 10 g of dried sample was then sent to an external lab (Barringer Labs) for determination of total phosphorus (TP) concentrations ($\mu\text{g g}^{-1}$ dw) using an acid digest of HCl. The remaining dry sample was re-weighed and ashed at 500°C overnight (minimum 12 hrs) to determine organic matter content as percent loss on ignition (%LOI) (Murdoch *et al.*, 1997). This process was repeated for the lower core samples. The remaining wet sediment, from both upper and lower cores, was combined and returned to cold storage. The composite sample was later air-dried for determination of chloride concentrations by aqueous extract (Page, 1982) and measurement at the ANALEST facility using an isocratic elution with 3.5 mM sodium carbonate and 1mM sodium bicarbonate at a flow rate of 1 mL/min. Detection was by conductivity detector with ion suppression (see Appendix II for chloride calibration data).

3.2.2 Benthic Invertebrate Identification: Initially, some Kahle's preserved samples were treated with Rose Bengal to facilitate sorting, but the process did not prove to be valuable and was discontinued after five samples. Within four months after field collection, the preserved samples were washed and sorted from the sediment using a 0.5 mm mesh then stored in 80% ethanol until identification to family where possible, but sometimes only to order or phylum. The various taxa were identified using a dissecting microscope and the following references: Merritt and Cummins (1996); Clarke (1981); Burt (1970); Malakhov (1994).

3.2.3 Aquatic Plant Identification: All aquatic macrophyte samples were taken to the Royal Ontario Museum (ROM) Herbarium and identified with the assistance of ROM Staff and Hellquist and Crow (1984). Plant samples were dried, pressed and housed at the Herbarium.

3.2 Results and Statistical Analyses

3.3.1 Total Phosphorus, Loss on Ignition and Water Content: The chemical and physical data collected from the sediment cores are summarized in Table 3.2(a), (b) and (c). Average concentrations or values have been calculated for the younger sediment (0-5 cm) and older sediment (5-15 cm) for each LHS (some data points are missing for a variety of reasons). I believe that the regrading of all NP stations prior to planting must have mixed some surficial native sediments (approximately top 5 cm) with out-sourced material, but I do not know the ratio of "new material" to native sediment and the precise content of new material is not known. Sod bundles were used as a growth medium at NP-4, but not at the other NP stations. I tried to use the vertical profile of "younger" to "older" sediments from undisturbed sites to isolate potential effects of this shoreline regrading/mixing on the younger sediment taken from the NP stations.

Average TP concentrations from 2 of the 3 EV stations increased with core depth/age, as did all of the TP average values for the four NP stations. This pattern has been commonly reported in the literature (Drake and Heaney, 1987). The upper 0-5cm of sediment in the NV stations consisted only of rocks (2-6 cm diameter) that precluded analyses of TP (and other parameters). All one could say was that TP concentrations in the older sediments of NV stations were also consistently higher than found in the younger sediments of other cores.

There are no observable trends in the vertical distribution of average %LOI values for either EV or NP habitats. As stated above for TP, average %LOI values in the older sediments of NV stations are higher than the rock-dominated younger sediments.

Intuitively, %SAT values are expected to decrease with the core depth, due simply to gravity/compression if nothing else. The vertical pattern of average %SAT values within all the EV stations and NP-3 agree with this prediction. However, NP-1, NP-2 and NP-4 all show an increase in water content with core depth. Although analyses for %SAT was not determinable for the NV habitats, the relatively large pore space between rocks in the upper top 0-5 cm is assumed to have "held" more water than the finer sediments associated deeper within those cores.

3.3.1.1 Analysis of Variance: Differences in values of TP, %LOI and %SAT among replicate habitat stations (factor 1) and between stations of different habitat type (factor 2) were analyzed using *Systat 9.0 for Windows*, General Linear Model analyses of variance for a two-factor nested design. This statistical

package was chosen as it allows for uneven sample sizes at the replicate-core level. All other parameters were also analyzed using this software package and nested design. The original data were tested for normality by way of frequency histograms. Both TP and %SAT are reasonably normal, but %LOI, benthos abundance and shoot density all exhibit left-hand skewness. Despite the deviation from normal, these data were not transformed for the analysis of variance, as this test is considered quite robust (D. Jackson, Per. Comm.). To lend support to this decision, the residuals for all data sets were also tested for normality (using frequency histograms), as they better reflect how normally distributed (or not) are the data (D. Jackson, Per. Comm.). The degree of skewness of the original data was not observed in the residuals, which were reasonably normal.

I excluded data from NP-2 and NP-4 from statistical analyses. NP-2 failed to support plant growth (discussed in the Aquatic Macrophyte Results section) and NP-4 is the only station with three years of growth. Neither of these two conditions is replicated in the study design and it seemed inappropriate to include them. This situation does not preclude using all the data for comparative or trend analyses on a qualitative or descriptive basis.

Differences in mean sediment core concentrations/values for TP, %LOI and %SAT were observed for the two factors examined (i.e. among treatments and among LHS within treatments) (Table 3.3). However, the pattern of differences associated with the two factors is not the same for each parameter. TP (0-5cm) is significantly patchy among LHS within treatments (small spatial scale) but not among treatments, while both %LOI and %SAT (0-5 cm) showed significant variation only between different treatments (larger spatial scale). %LOI and %SAT average values were lowest at NP stations (2.3% - 8.4% and 14.5% - 38.9%, respectively) and highest at EV stations (8.9% - 10.9% and 45.5% - 46.6%, respectively).

Older sediment TP concentrations (5-15 cm) were significantly different among treatments and among LHS within treatments, with the lowest average concentrations associated with the NV station (199.8 – 358.1 $\mu\text{g g}^{-1}$ dw). %SAT was only significantly different between NP and EV stations, with generally higher average values at the EV stations (14.7% - 33.3%). No significant differences were detected for %LOI despite a wide range of average core values

Table 3.2a: 1996 Physical and Chemical Parameters of Sediment Cores at Established Vegetation Sites

| Sub-sample | Core Depth | Habitat Stations: | | | | | | | | |
|------------|------------|-------------------|------------------|------|-------|------------------|------|-------|------------------|------|
| | | EV-1 | | | EV-2 | | | EV-3 | | |
| | | TP (µg/g dw) | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT |
| I | S | 690.0 | 17.3 | 67.7 | 413.0 | 6.4 | 47.8 | 290.0 | 6.8 | 21.1 |
| II | S | 708.0 | 26.6 | 72.9 | 388.0 | 4.0 | 30.9 | 440.0 | 7.4 | 49.3 |
| III | S | 622.0 | 8.6 | 63.5 | 389.0 | 7.9 | 42.9 | 407.0 | 6.8 | 58.2 |
| IV | S | 641.0 | 2.7 | 39.0 | 446.0 | 15.2 | 61.8 | 488.0 | 8.3 | 40.5 |
| V | S | 628.0 | 2.0 | 22.8 | 607.0 | 5.2 | 36.7 | 441.0 | 7.4 | 48.9 |
| VI | S | 590.0 | 1.3 | 24.2 | 332.0 | 23.6 | 53.0 | na | 14.5 | 61.6 |
| VII | S | 607.0 | 2.4 | 37.0 | | | | | | |
| VIII | S | na | 19.4 | 63.6 | | | | | | |
| IX | S | na | 11.1 | 6.5 | | | | | | |
| AVERAGE | S | 636.0 | 10.5 | 46.4 | 493.7 | 10.4 | 46.5 | 413.2 | 6.9 | 46.6 |
| STDEV | S | 47.2 | 9.6 | 26.5 | 119.0 | 7.6 | 11.1 | 74.7 | 2.9 | 14.6 |
| I | D | 595.0 | 4.7 | 39.6 | 567.0 | 6.7 | 40.6 | 373.0 | 1.6 | 14.4 |
| II | D | 685.0 | 4.7 | 29.5 | 667.0 | 4.2 | 24.6 | 371.0 | 4.1 | 26.2 |
| III | D | 632.0 | 1.9 | 24.2 | 504.0 | 7.8 | 32.3 | 538.0 | 4.5 | 28.3 |
| IV | D | 601.0 | 2.3 | 21.5 | 399.0 | 42.1 | 67.7 | 690.0 | 1.9 | 18.1 |
| V | D | 646.0 | 4.3 | 37.4 | 544.0 | 15.7 | 33.3 | 636.0 | 8.8 | 48.4 |
| VI | D | 627.0 | 1.4 | 20.2 | 379.0 | 11.6 | 47.8 | 589.0 | 7.2 | 44.5 |
| VII | D | 732.0 | 0.8 | 73.9 | | | | | | |
| VIII | D | 627.0 | 3.7 | 36.5 | | | | | | |
| IX | D | 599.0 | 20.5 | 19.6 | | | | | | |
| AVERAGE | D | 633.4 | 4.9 | 33.3 | 507.2 | 14.7 | 39.4 | 532.7 | 4.7 | 30.1 |
| STDEV | D | 46.6 | 6.0 | 17.3 | 122.2 | 14.0 | 11.9 | 194.2 | 2.9 | 14.1 |

S = shallow 0-5cm; D = deep 5-15cm

na = not available due to insufficient sample

Table 3.2b: 1996 Physical and Chemical Parameters of Sediment Cores in the Newly Planted Sites

| Sub-sample | Core Depth | Habitat Station: | | | | | | | | | | | |
|------------|------------|------------------|------------------|------|-------|------------------|------|-------|------------------|------|-------|------------------|-------|
| | | NP-1 | | | NP-2 | | | NP-3 | | | NP-4 | | |
| | | TP (µg/g dw) | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT |
| I | S | 480.0 | 1.2 | 10.3 | 412.0 | 1.6 | 10.2 | 444.0 | 6.3 | 24.1 | 664 | na | 23.21 |
| II | S | 466.0 | 1.2 | 9.2 | 583.0 | 8.3 | 51.3 | 773.0 | 3.5 | 22.1 | 641 | 17.96 | 18.47 |
| III | S | 212.0 | 1.7 | 13.4 | 826.0 | 30.6 | 70.9 | 681.0 | 2.1 | 27.4 | 568 | 4.86 | 20.04 |
| IV | S | 474.0 | 3.4 | 18.8 | 474.0 | 1.9 | 20.6 | 701.0 | 4.3 | 21.7 | 272 | 1.43 | 21.48 |
| V | S | 308.0 | 2.3 | 20.3 | 308.0 | 4.7 | 40.6 | na | na | na | 312 | 2.72 | 9.55 |
| VI | S | 361.0 | 3.8 | 14.3 | 446.0 | 3.3 | 38.7 | na | na | na | 465 | 1.92 | 12.44 |
| VII | S | | | | | | | | | | | | |
| VIII | S | | | | | | | | | | | | |
| IX | S | | | | | | | | | | | | |
| AVERAGE | S | 395.5 | 2.3 | 14.9 | 603.0 | 6.4 | 38.9 | 649.8 | 3.8 | 23.8 | 488.7 | 6.8 | 17.7 |
| STDEV | S | 108.1 | 1.1 | 4.7 | 178.9 | 11.2 | 21.6 | 142.7 | 1.3 | 2.6 | 193.4 | 6.9 | 6.4 |
| I | D | 494.0 | 1.2 | 10.6 | na | na | na | na | na | na | 461 | 3.81 | 16.5 |
| II | D | 497.0 | 1.4 | 8.4 | 587.0 | 13.4 | 56.7 | 789.0 | 3.9 | 20.7 | 703 | 2.97 | 21.82 |
| III | D | 303.0 | 1.8 | 7.9 | 842.0 | 15.6 | 65.9 | 782.0 | 4.5 | 24.5 | 540 | 2.42 | 20.12 |
| IV | D | 586.0 | 3.4 | 31.0 | 707.0 | 10.4 | 35.0 | 744.0 | 3.8 | 20.6 | 466 | 1.05 | 31.86 |
| V | D | 596.0 | 1.8 | 19.0 | 526.0 | 4.8 | 28.2 | na | na | na | 336 | 1.99 | 7.81 |
| VI | D | 732.0 | 6.6 | 18.0 | na | na | na | na | na | na | 664 | 2.16 | 16.6 |
| VII | D | | | | | | | | | | | | |
| VIII | D | | | | | | | | | | | | |
| IX | D | | | | | | | | | | | | |
| AVERAGE | D | 534.6 | 2.6 | 16.9 | 666.3 | 11.0 | 47.0 | 767.3 | 4.0 | 21.9 | 670 | 2.38 | 18.79 |
| STDEV | D | 142.8 | 1.7 | 6.9 | 149.0 | 4.7 | 18.2 | 24.0 | 0.4 | 2.2 | 129 | 0.92 | 6.01 |

S = shallow 0-5cm; D = Deep 5-15cm

na = not available due to lab error; nc = core not collected

Table 3.2c: 1996 Physical and Chemical Parameters of Sediment Cores at Non-Vegetated Sites

| Sub-sample | Core Depth | Habitat Stations: | | | | | |
|------------|------------|-------------------|------------------|-----------------|----------|------------------|-----------------|
| | | NV-1 | | | NV-2 | | |
| | | TP (µg/g dw) | %Cl ₂ | %BAT | TP | %Cl ₂ | %BAT |
| all cores | S | all rock | all rock | all rock | all rock | all rock | all rock |
| AVERAGE | S | na | na | na | na | na | na |
| I | D | 234.0 | 4.6 | na ¹ | 498.0 | 1.6 | na ¹ |
| II | D | 208.0 | 7.4 | na ¹ | 365.0 | 1.6 | na ¹ |
| III | D | 171.0 | 4.1 | na ¹ | na | 1.6 | na ¹ |
| IV | D | 188.0 | 2.9 | na ¹ | 191.0 | 1.5 | na ¹ |
| V | D | | | | | | |
| VI | D | | | | | | |
| VII | D | | | | | | |
| VIII | D | | | | | | |
| IX | D | | | | | | |
| AVERAGE | D | 198.6 | 4.7 | na ¹ | 358.0 | 1.6 | na ¹ |
| STDEV | D | 27.4 | 1.9 | | 156.3 | 0.0 | |

S = shallow 0-5cm; D = deep 5-15cm

all rock or na = insufficient sample; na¹ = not available due to lab error

among treatment sites (Table 3.2). Overall, there were more significant differences among treatment sites (larger spatial scale) than among LHS within treatments (smaller spatial scale) (four of the six analyses).

Morrisey *et al.*, (1992) suggest that small spatial scale patchiness or localized heterogeneity (i.e. between cores within replicate habitat stations), can mask differences at larger scales, particularly when the power of the test becomes low. This could be the case with TP (only one and two degrees of freedom) and perhaps %LOI in older sediment (two and four degrees of freedom), which had a marginal p-value (0.059).

3.3.1.2 Regression Analysis: Simple linear regression analysis was applied to decide whether relationships between TP, %LOI and %SAT, commonly reported in the literature, were evident in Grenadier Pond and contribute to the explanation of patterns observed above. Originally, the data were not transformed and the regression analyses did not identify any correlations. Closer inspection of the data using frequency histograms, showed TP and %SAT data to be reasonably normally distributed, but %LOI data were skewed to the right. Transforming the TP data ($\log(x)$) did not improve the normality, so it was not changed. Using an arcsine(x) transformation improved the normality of %SAT and effectively normalized the %LOI data.

Interpretation of the correlation (i.e. r^2 -value) must still be done cautiously since 1) sample size (n) is very low, less than 30 for every case, and as low as seven for the NV station and 2) a ratio, even when it is the variable of interest, as it is in this case, can lead to "spurious correlations" (Jackson and Somers, 1990). Figures 3. (a) to (m) illustrate the regression results with predicted values overlying the transformed and untransformed data.

TP concentrations are often directly proportional to %LOI (Barko and Smart, 1986). %SAT has also been positively correlated with %LOI (Cyr, 1998; Barko and Smart, 1986). It might then logically follow that TP concentrations would correlate positively with %SAT.

The EV stations do not show any correlations between TP and %LOI or TP and %SAT at either depth of sediment (Figures 3.2 a, b, f and g). Figure 3.2 (j) shows a strong positive correlation between %LOI and %SAT for the top 0-5 cm of sediment ($r = +0.72$), but the data spread is diffuse with only 12 data points, making the high r -value suspect. The older sediment does not reveal a strong correlation ($r = +0.08$), however outlying data points may be pulling the regression line and perhaps masking a stronger relationship (Figure 3.2k).

The general interpretation of the regressions for the NP stations is that none of the expected relationships are demonstrated, with perhaps the exception of %LOI and %SAT in the younger core segment ($r = +0.54$; Figure 3.2 (l)). Figures 3.2d), and i) suggest stronger correlations ($r = +0.81$ and $+0.71$, respectively), however

Table 3.3 Summaries for Analyses of Variance for Selected Parameters*

| Parameter | Source of Variation | df | Mean Sq. | F-ratio | P-value |
|-----------------------------|---|----|-----------|---------|---------|
| a) TP (0-5 cm sediment) | among treatments (ev & np) | 1 | 544.03 | 0.04 | 0.842 |
| | among LHS within treatments (ev, np) | 4 | 85053.51 | 6.34 | 0.001 |
| | error within an LHS | 28 | 13406.8 | | |
| b) TP (5-15 cm sediment) | among treatments (ev & np & nv) | 2 | 317262.83 | 26.59 | 0.000 |
| | among LHS within treatments (ev, np, nv) | 5 | 45481.61 | 3.81 | 0.008 |
| | error within an LHS | 33 | 11930.49 | | |
| c) %LOI (0-5 cm sediment) | among treatments (ev & np) | aq | 307.64 | 7.41 | 0.011 |
| | among LHS within treatments (ev, np) | 2 | 5.4 | 0.13 | 0.941 |
| | error within an LHS | 26 | 41.5 | | |
| | | | | | |
| d) % LOI (5-15 cm sediment) | among treatments (ev & np & nv) | 2 | 106.83 | 2.47 | 0.101 |
| | among LHS within a treatment (ev, np, nv) | 4 | 110.31 | 2.55 | 0.059 |
| | error within an LHS | 31 | 43.22 | | |
| e) %SAT (0-5 cm sediment) | among treatments (ev & np) | 1 | 3418.08 | 14.88 | 0.001 |
| | among LHS within treatments (ev, np) | 3 | 216.69 | 0.94 | 0.435 |
| | error within an LHS | 23 | 229.69 | | |
| f) %SAT (5-15 cm sediment) | among treatments (ev & np) | 1 | 1359.61 | 7.55 | 0.011 |
| | among LHS within treatments (ev, np) | 3 | 114.09 | 0.63 | 0.600 |
| | error within an LHS | 25 | 180.15 | | |

* analyzed using SYSTAT 9.0, General Linear Model, nested design

outlying data points appear to be influencing the regression line in both cases. A very diffuse spread of only eight data points makes definitive interpretation problematic.

The only regression analysis applied to the NV stations was TP concentrations vs %LOI for older sediment (Figure 3.2e). No correlation was identified.

3.3.1.3 Comparison of TP, % LOI and %SAT in Grenadier Pond Sediment to Other Similarly Impacted Ponds: TP, %LOI and %SAT are, for this paper, the main indicators of rooted macrophyte influence on surficial sediments. Values for TP, %LOI and %SAT in Grenadier Pond were compared with those reported from other shallow, eutrophic ponds. Figures 3.3 (a), (b) and (c) use maximum and minimum values for the comparison. I included two oligo-mesotrophic ponds for greater contrast. All values are from the littoral zone of these systems. Grenadier Pond has the lowest sediment TP concentrations of any of the other eutrophic lakes, with even maximum values an order of magnitude lower (Figure 3.3a). The range in organic matter content (%LOI) is comparable to those of other eutrophic ponds, and considerably higher than lakes of lower nutrient status (Figure 3.3b). The eutrophic Onondaga Lake appears closer to mesotrophic levels in organic content as its sediments are reportedly sandy (Masden *et al.*, 1993). Grenadier Pond has a wider absolute range in sediment water content (%SAT) than any other eutrophic pond or the lesser-enriched systems (Figure 3.3c). The large range in %SAT may reflect the wide range of sediment textures found in the Grenadier, a subject presented in the next section and discussed later.

3.3.2 Sediment Texture: Table 3.4 summarizes the qualitative descriptions of the sediment in the nine habitat stations and the two core depths (0-5cm and 5-15 cm). The number of cores with a given sediment description is also listed for each core depth, but similar numbers or placement on Table 3.4 does not necessarily reflect a continuous description through one or more cores. The 1996 data from Grenadier Pond were taken prior to some biomanipulation activity and are not representative of disturbed sediments occupying NP-1, NP-2 or NP-3 stations.

There are four general trends identified in the sediment texture data:

1. Increased consolidation of sediment with core depth (not all cores, but at all stations);
2. Woody debris and/or fibrous organic material is essentially found only at the three EV stations, with the noted exception of NP-4, which has leafy detritus and fibrous material, perhaps reflecting the use of sod material as a growth medium at this location;
3. The EV stations are the most heterogeneous, with an average of 3 and 3.3 different sediment descriptions required to characterize the range of sediments occurring in the upper 0-5 cm and deeper 5-15 cm of the cores, respectively;

Figure 3.2a) Established Vegetation Stations, 0-5cm Sediment, TP vs %LOI

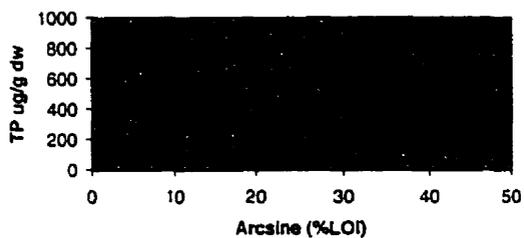


Figure 3.2b) Established Vegetation Stations, 5-15cm Sediment, TP vs %LOI

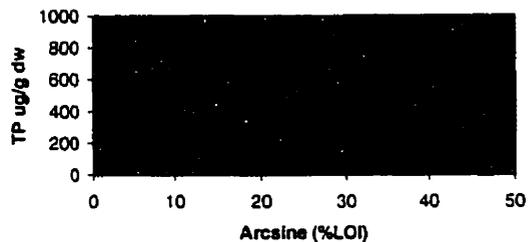


Figure 3.2c) Newly Planted Stations, 0-5cm Sediment, TP vs %LOI

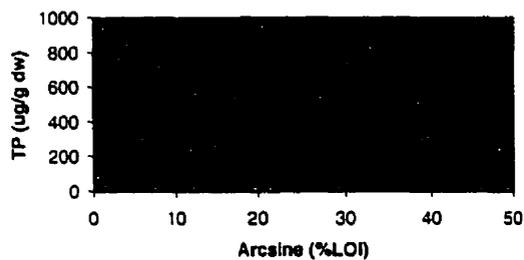
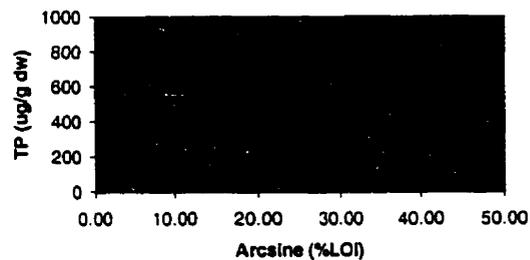


Figure 3.2d) Newly Planted Stations, 5-15cm Sediment, TP vs %LOI



r^2 values:

Figure 3.2a) 0.01

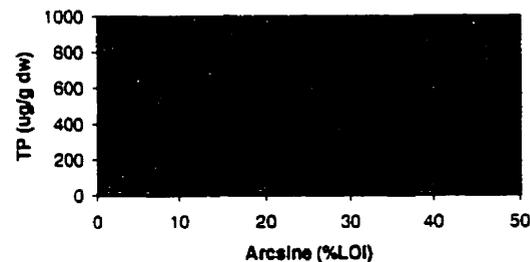
Figure 3.2b) 0.22

Figure 3.2c) 0.05

Figure 3.2d) 0.65

Figure 3.2e) 0.30

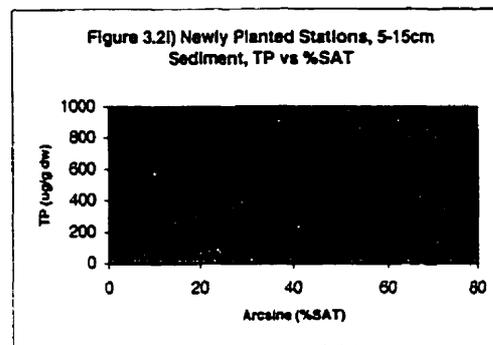
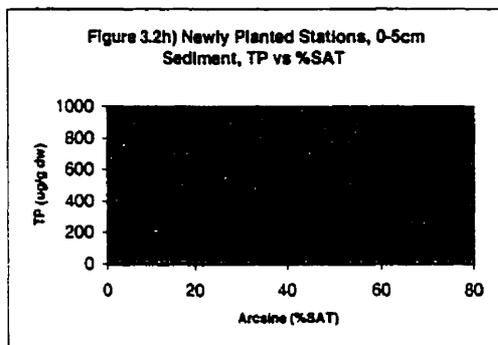
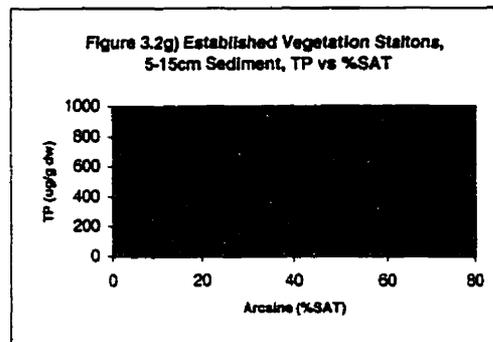
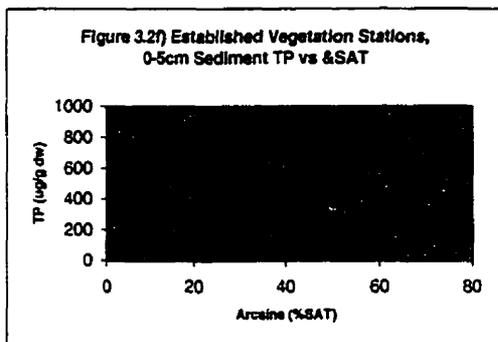
Figure 3.2e) Non-Vegetated Stations, 5-15cm Sediment, TP vs %LOI



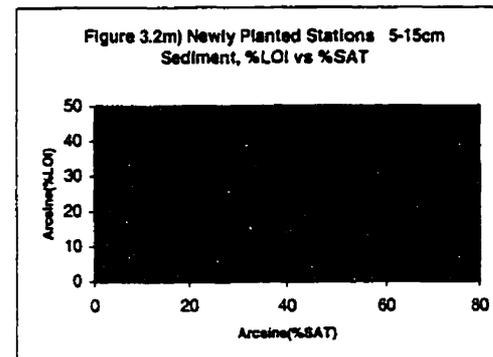
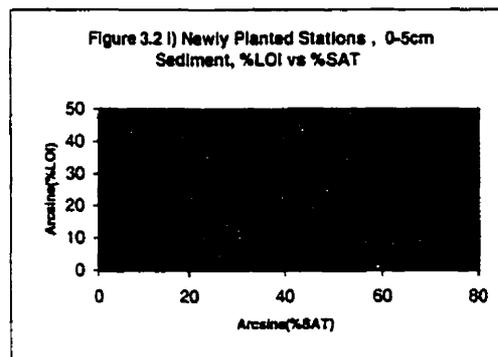
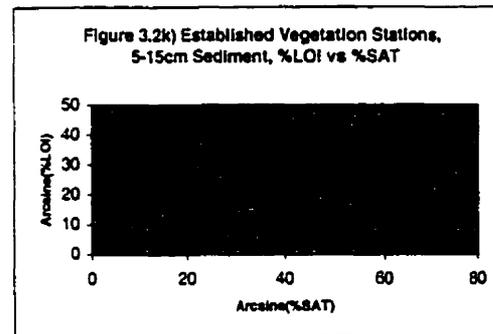
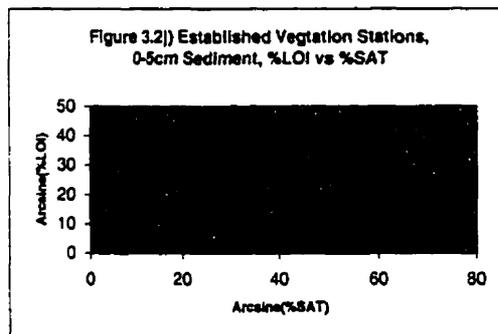
Figures 3.2 (a) - (e): Simple Linear Regression Analyses for TP (untransformed) vs. %LOI (transformed).

Observed values (diamond shape)

Predicted values (line)



r^2 values: f) 0.02, g) 0.001, h) 0.30, i) 0.50



r^2 values: j) 0.53; k) 0.08; l) 0.30; m) 0.0032

Figures 3.2 (f) - (i): Simple Linear Regression Analyses for TP (untransformed) vs. %SAT (transformed).
Figures 3.2 (j) - (m): Simple Linear Regression Analyses for %LOI (transformed) vs. %SAT (transformed).
Observed values (diamond shape)
Predicted values (line)

Figure 3.3a) Comparison of [TP] in Littoral Sediment of Grenadier Pond and Other Eutrophied Lakes

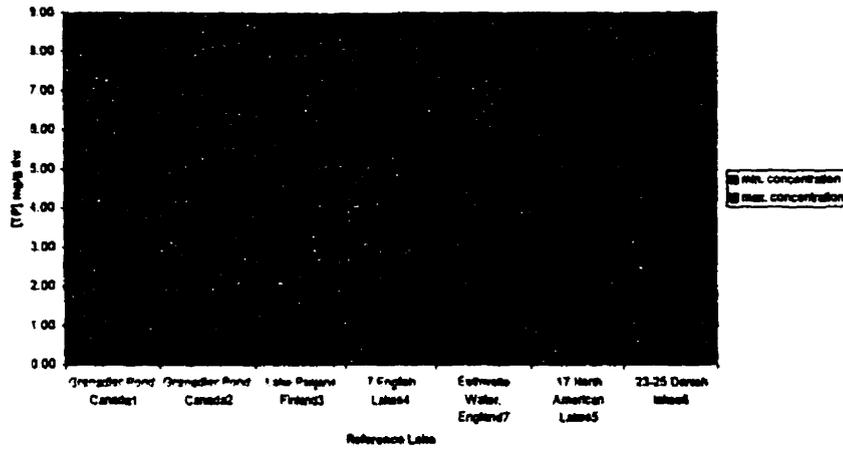


Figure 3.3b) Comparison of %LOI in Littoral Sediment of Grenadier Pond and Other Eutrophic or Mesotrophic Lakes

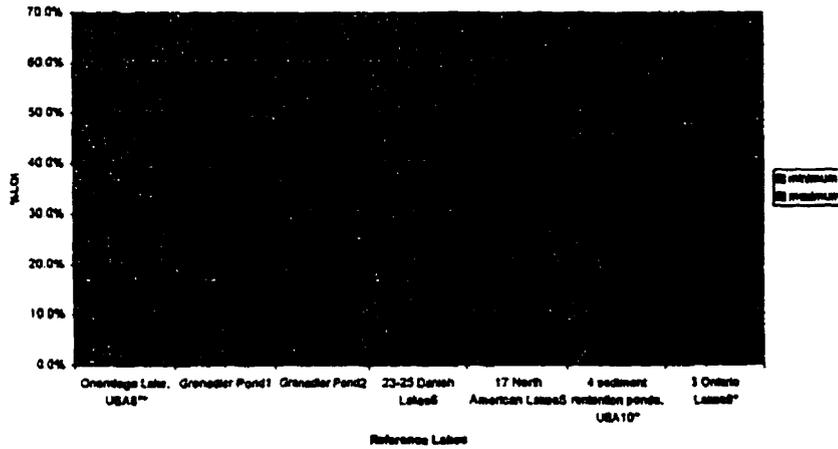
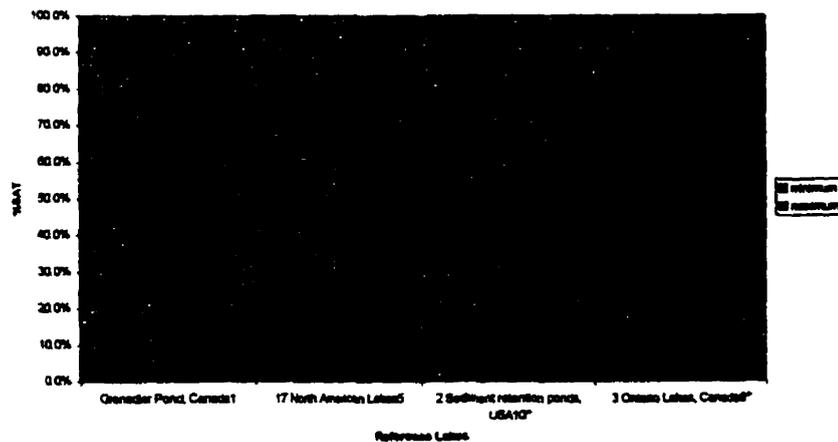


Figure 3.3c) Comparison of %SAT in Littoral Sediment of Grenadier Pond and Other Eutrophic or Mesotrophic Lakes



Figures 3.2(a-c): Study references are as follows 1) Tu, M.Sc. Research, 1996; 2) GLL, 1996; 3) Kairesalo and Mattilainen, 1994; 4) Phillips et al, 1994; 5) Barbo and Smart, 1998; 6) Sondergaard et al, 1998; 7) Drake and Hearney, 1987; 8) Meraden et al, 1993; 9) Cyr, 1990; 10) Allan et al, 1997
 * Oligo-mesotrophic lakes
 ** Large proportion of sandy sediment

4. In contrast to the EV station variability, the NV stations are uniformly algae-covered-rock within the top 0-5 cm of all the cores; descriptions of the deeper core sections are all very similar.

3.3.3 Benthic Macroinvertebrates: The greatest diversity of benthic macroinvertebrates occurs in the EV stations with 12 different taxa being represented as compared to 10 for both NP and NV, respectively (Table 3.5). In terms of total number of animals per LHS, NV-1 and NV-2 have the highest numbers at one to two orders of magnitude greater than any of the EV or NP stations (Figure 3.4a). The high numbers the NV stations are associated with chironomid abundance. When comparing just the EV and NP stations, total numbers appear similar, but the various taxa are more evenly distributed through areas of established vegetation, with worm and insect species dominating the newly planted areas (Figure 3.4b).

3.3.3.1 Analysis of Variance: Five of the taxa in the samples were consistently present and represented different relative abundances (low, moderate, high). These, along with the total number of animals were analyzed by two-factor nested analyses of variance; Table 3.6 summarizes the results. The total number of individual animals differed significantly between the different habitat stations reflecting the great abundance of chironomid species at the NV stations. Similarly, chironomid abundance was significantly higher at the NV stations. Amphipod abundance was significantly lower at the NP stations compared to the EV or NV stations, but nematode abundance was significantly higher. Significant localized patchiness was also detected for nematodes. Review of nematode distribution in Table 3.5 suggests that the localized heterogeneity is among the NP stations. Neither oligochaete nor snail abundance was significantly different among treatment sites or among LHS within treatments.

| Table 3.4: Littoral Sediment Description for Grenadier Pond 1996* and 1998 | | | | | |
|--|--|---|-----------------|---|-----------------|
| LHS | Description of Proximal Littoral Sediment (0-10cm), 1996* | Description of Sediment Cores, 1998 | | | |
| | | 0-5 cm | Number of Cores | 5-15cm | Number of Cores |
| EV-1 | Black organic silt (0-5 cm), light brown organic material marbled with black organic silt (10+ cm) | Dark brown, organic, peaty, unconsolidated | 1 | Dark, organic, woody debris, peaty, fine sand, unconsolidated | 2 |
| | | Dark brown, organic, sandy-silt, unconsolidated | 1 | Dark, organic, peaty, fine sand, moderately consolidated | 1 |
| | | Dark brown, organic, woody debris, sandy-silt, unconsolidated | 7 | Light grey, woody debris, sandy-silt, consolidated Light-dark brown, silty, consolidated | 1 5 |
| EV-2 | Dark brown/black organic silt 1.7% very coarse sand, 8.1% coarse sand, 15.3% medium sand, 27.9% fine sand, 14.9% very fine sand, 31.2% silt, 0.81% clay | Light brown, organic fibrous material, woody debris, silty-sand, unconsolidated | 2 | Dark brown, peaty, woody debris, unconsolidated | 3 |
| | | Dark, organic fibrous material, woody debris, sandy, moderate consolidation | 4 | Very dark brown, silty-sandy muck, moderately consolidated | 3 |
| EV-3 | NA | dark, organic fibrous material, woody debris, sandy-silt, moderately consolidated | 2 | dark, fine woody debris, sandy-silt, consolidated | 2 |
| | | dark, organic, sandy-silt, moderately consolidated | 2 | dark, organic, sandy-silt, consolidated | 2 |
| | | grey-brown, silty-sand, gravel, consolidated | 1 | grey-brown, silty-sand, gravel, consolidated | 1 |
| | | grey-brown, some woody debris, silty-sand, consolidated | 1 | grey-brown, some woody debris, silty-sand, consolidated | 1 |
| NP-1 | NA | Red-brown, coarse sand-gravel (50%), fine sand-silt (50%), crumbly | 2 | Red-brown, coarse sand-gravel (25%), fine sand-silt (75%), moderate consolidation | 2 |
| | | Brown, fine-coarse sand, gravel, minimal silt, crumbly | 4 | Grey-brown, fine sand-silt (90%), coarse sand (10%), moderate consolidation | 4 |
| NP-2 | Light brown, organic fibrous material, site of highest littoral %LOI 0.1% v. coarse sand, 3.9% coarse sand, 14.1% med. Sand, 26.6% fine sand, 12.3% v. fine sand, 41.3% silt, 1.8% clay | Red-brown, fine & coarse sand, pebbles, rocks, crumbly | 1 | Only 4 cores collected at this depth | |
| | | Dark brown/black, fine sandy-silt, organic muck, unconsolidated | 5 | Dark brown/black, fine sandy-silt, moderately consolidated | 4 |
| NP-3 | NA | Dark brown, fine sand (25%), coarse sand (25%), pebbles (50%), organic bits, moderate consolidation | 1 | only 3 cores collected at this depth | |
| | | Grey-brown, fine sandy-silt, unconsolidated | 3 | Grey-brown, fine sandy-silt, moderate-high consolidation | 3 |
| NP-4 | NA | grey-brown, some fibrous organic material, sandy-silt, moderately consolidated | 2 | grey-brown, coarse-fine sandy-silt, moderately consolidated | 2 |
| | | light brown, fine sandy-silt, clayey, consolidated | 1 | light brown, fine sandy-silt, clayey, consolidated | 1 |
| | | brown, coarse sand, gravel, some silt, leafy detritus, moderately consolidated | 3 | brown, coarse sand, gravel, some silt, lots leafy detritus, consolidated | 3 |
| NV-1 | NA | large rocks (2-6 cm diameter) covered in algae, crumbly | 4 | brown, fine sandy-silt, some coarse sand, gravel and rock, moderate consolidation | 1 |
| | | | | mainly large rock with some sandy-silt | 1 |
| | | | | black muck, silty-sand, large rock, unconsolidated | 1 |
| | | | | dark brown, large rocks, some silty-sand, unconsolidated | 1 |
| NV-2 | NA | large rocks (2-6 cm diameter) covered in algae, crumbly | 4 | grey, gravel with silt layer, unconsolidated | 2 |
| | | | | dark grey, gravel with coarse sand-silt, unconsolidated | 1 |
| | | | | grey-brown, gravel with coarse sand-silt, unconsolidated | 1 |

* = GLL, 1996; sediment characteristics are pre-biemanipulation of the littoral zone

Table 3.5 Benthic Macroinvertebrate Diversity in Top 0-5cm of Littoral Sediment in Grenadier Pond

| Invertebrate Name | Classification Level | Common Name | LHS: | EV1 | EV2 | EV3 | NP4 | NP1 | NP2 | NP3 | NV1 | NV2 |
|---------------------------------------|-----------------------------|---------------------|-------------|------------|------------|------------|------------|------------|------------|------------|------------|-------------|
| Insecta | | | | | | | | | | | | |
| <i>Chironomidae</i> larvae | family | midge | | 23 | 15 | 16 | 13 | 21 | 12 | 60 | 473 | 1046 |
| <i>Tipulidae</i> | family | crane fly | | 0 | 0 | 0 | 0 | 2 | 0 | 1 | 0 | 0 |
| <i>Plecoptera</i> nymph | order | stonefly | | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| <i>Odonata</i> nymph | order | damselfly | | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| <i>Coleoptera</i> | order | beetle larvae | | 3 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 1 |
| Annelida | | | | | | | | | | | | |
| <i>Oligochaeta</i> | class | (true) aquatic worm | | 8 | 7 | 12 | 33 | 7 | 1 | 10 | 1 | 1 |
| <i>Tricladidae</i> | family | flatworm | | 3 | 0 | 0 | 1 | 5 | 0 | 0 | 27 | 86 |
| <i>Nematoda</i> | phylum/class | roundworm | | 1 | 0 | 0 | 11 | 24 | 12 | 6 | 0 | 1 |
| Crustacea | | | | | | | | | | | | |
| <i>Amphipoda</i> | order | scud | | 13 | 6 | 3 | 4 | 0 | 0 | 1 | 20 | 4 |
| Mollusca | | | | | | | | | | | | |
| <i>Physidae</i> | family | tadpole snail | | 2 | 2 | 8 | 3 | 0 | 1 | 0 | 8 | 0 |
| <i>Lymnaeidae</i> | family | pond snail | | 4 | 7 | 6 | 1 | 1 | 2 | 0 | 10 | 0 |
| <i>Planariidae</i> | family | ramshorn snail | | 2 | 14 | 13 | 3 | 0 | 0 | 0 | 1 | 0 |
| <i>Sphaeriidae</i> | family | finger nail clam | | 2 | 3 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| <i>Hydrobiidae</i> | family | spire snail | | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 |
| Total Number of Invertebrates: | | | | 63 | 55 | 59 | 69 | 60 | 29 | 78 | 542 | 1139 |

Figure 3.4a) Distribution of Benthic Invertebrates For All Littoral Habitat Stations

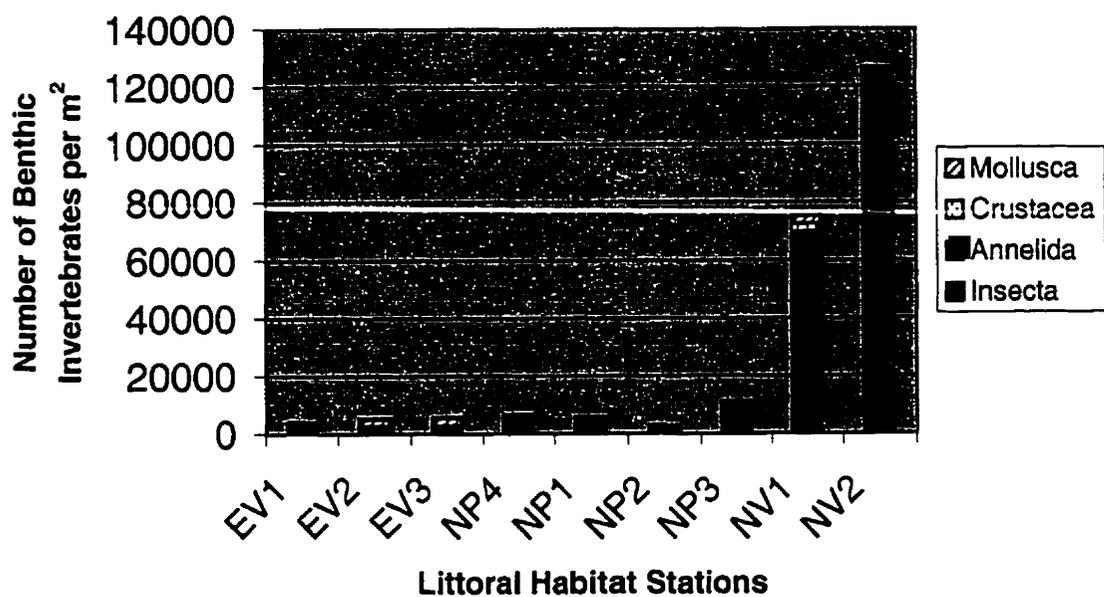


Figure 3.4b) Distribution of Benthic Invertebrate for Vegetated Stations Only

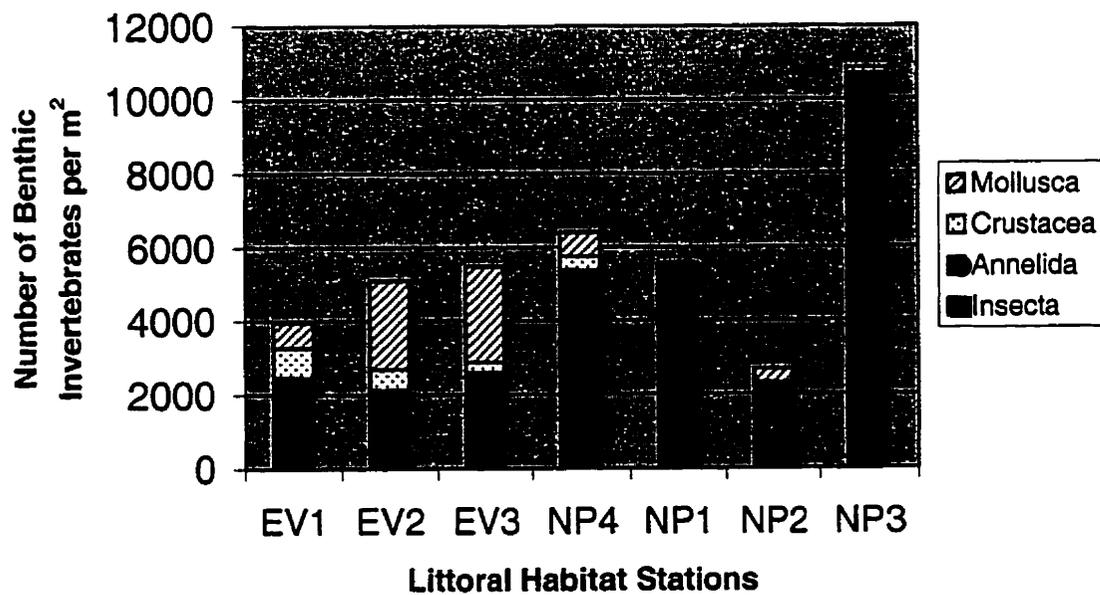


Table 3.6. Summaries of Analyses of Variance for Selected Benthic Invertebrate Taxa*

| Taxa | Source of Variance | df | Mean Sq. | F-ratio | P-value |
|---------------------|--|----|----------|---------|---------|
| Total # Individuals | Among treatments (ev & np & nv) | 2 | 63117.49 | 18.65 | 0.000 |
| | Among LHS within treatments (ev, np, nv) | 5 | 5702.24 | 1.68 | 0.166 |
| | Error within an LHS | 33 | 3385.11 | | |
| Total Chironomids | Among treatments (ev & np & nv) | 2 | 57679.49 | 18.89 | 0.000 |
| | Among LHS within treatments (ev, np, nv) | 5 | 4615.44 | 1.51 | 0.214 |
| | Error within an LHS | 32 | 3053.47 | | |
| Total Oligochaetes | Among treatments (ev & np & nv) | 2 | 2.07 | 0.5 | 0.609 |
| | Among LHS within treatments (ev, np, nv) | 5 | 3.17 | 0.77 | 0.578 |
| | error | 33 | 4.11 | | |
| Total Nematodes | Among treatments (ev & np & nv) | 2 | 22.84 | 16.99 | 0.000 |
| | Among LHS within treatments (ev, np, nv) | 4 | 4.55 | 3.39 | 0.022 |
| | Error within an LHS | 28 | 1.34 | | |
| Total Amphipodes | Among treatments (ev & np & nv) | 2 | 2.68 | 0.9 | 0.417 |
| | Among LHS within treatments (ev, np, nv) | 4 | 0.87 | 0.29 | 0.881 |
| | Error within an LHS | 28 | 2.97 | | |
| Total Snails | Among treatments (ev & np & nv) | 2 | 41.06 | 1.88 | 0.172 |
| | Among LHS within treatments (ev, np, nv) | 4 | 14.09 | 0.64 | 0.636 |
| | Error within an LHS | 26 | 21.88 | | |

* = Analyses run using SYSTAT 9.0, General Linear Model, Nested Design

3.3.4 Aquatic Macrophytes: The diversity of aquatic macrophytes is comparable between the EV and NP stations, with seven and six species, respectively, although the mix of species present is different for these two habitat types (Table 3.7). Two species are submersed (*Ceratophyllum demersum* and *Potamogeton pectinatus*). This value is low when compared to 31 eutrophic lakes from around the world where the average number of submersed species was 4.4 (Masden *et al.*, 1993). The NV stations are obviously devoid

Table 3.7. 1998 Aquatic Macrophyte Inventory on Grenadier Pond for All Littoral Habitat Stations

| Category | Species | Common Name | LHS: | | | | | | | | | TRCA, 1998* |
|------------------------|--------------------------------|--------------------------------|--------|--------|---------|---------|---------|-------|--------|------|------|-------------|
| | | | EV-1 | EV-2 | EV-3 | NP-4 | NP-1 | NP-2 | NP-3 | NV-1 | NV-2 | |
| Deep Emergent Marsh | <i>Typha latifolia</i> | Broad-leaved Cattail or common | x (74) | NF | x (39) | NF | x (38) | x (1) | x (1) | NF | NF | x |
| | <i>Scirpus validus</i> | Soft-stem Bulrush | NF | NF | NF | NF | NF | NF | NF | NF | NF | x |
| | <i>Acorus calamus</i> | Sweet Flag | NF | x (85) | x (102) | x (50) | x (132) | NF | x (73) | NF | NF | x |
| | <i>Iris pseudacorus</i> | Yellow Iris or Yellow Flag | NF | NF | NF | NF | NF | NF | NF | NF | NF | NF |
| | <i>Iris versicolor</i> | Blue Iris or Blue Flag | NF | NF | NF | NF | NF | NF | NF | NF | NF | x |
| | <i>Sparganium eurycarpum</i> | Great Bur Reed | NF | NF | NF | x (143) | x (11) | NF | x (49) | NF | NF | NF |
| Shallow Emergent Marsh | <i>Lythrum salicaria</i> | Purple Loosestrife | NP | NP | NP | NP | NP | NP | NP | NP | NP | x |
| | <i>Lycopus europeus</i> | European Waterhorehound | NP | NP | NP | NP | NP | NP | NP | NP | NP | NF |
| | <i>Lycopus americanus</i> | Cut-leaved Waterhorehound | NP | NP | NP | NP | NP | NP | NP | NP | NP | NF |
| | <i>Polygonum lapathifolium</i> | Nodding or Pale Smartweed | NP | NP | NP | NP | NP | NP | NP | NP | NP | NF |
| | <i>Polygonum amphibium</i> | Smartweed | NP | NP | NP | x* | NP | NP | NP | NP | NP | NF |
| Floating | <i>Nymphaea odorata</i> | Fragrant waterlily or white | x | x | x | NF | NF | x** | NF | NF | NF | x |
| | <i>Nuphar variegatum</i> | Yellow waterlily | x | NF | NF | NF | NF | NF | NF | NF | NF | x |
| | <i>Spirodela polymiza</i> | Greater Duckweed | NF | NF | NF | NF | NF | NF | NF | NF | NF | NF |
| | <i>Lemna minor</i> | Duckweed | x | x | x | NF | NF | x** | NF | NF | NF | x |
| Submerged | <i>Ceratophyllum demersum</i> | Coottail | x | x | x | NF | x | x | x | NF | NF | x |
| | <i>Potamogeton pectinatus</i> | Sago Pondweed | NF | x (3) | NF | NF | x (17) | NF | x (1) | NF | NF | x |
| | <i>Potamogeton crispus</i> | Crispy Pondweed | NF | NF | NF | NF | NF | NF | NF | NF | NF | NF |
| | <i>Potamogeton filiformis</i> | | NF | NF | NF | NF | NF | NF | NF | NF | NF | NF |
| | <i>Elodea nuttallii</i> | Nuttall's Waterweed | NF | NF | NF | NF | NF | NF | NF | NF | NF | NF |

NF - not found in 1998 study, but were historically present in Grenadier Pond

NP - not part of study (i.e., study restricted to certain species, e.g. strictly aquatic OR to certain lake area, e.g. plant-openwater interface)

x - identified within study site

x* - found in study site, although not expected for this habitat

x** - observed on open water, beyond study site

(#) - number of rooted emergent or rooted submergent macrophyte shoots

TRCA, 1998* = general pond survey; plant numbers not available

of vegetation, but a fairly extensive monoculture of submersed plants (*Potamogeton pectinatus*, TRCA, 1998) was observed in deeper water beyond the NV-1 study area. Three of the seven species found at the EV stations were floating species (*Nymphaea odorata*, *Nuphar variegatum* and *Lemna minor*) and, within the limits of the study sites, these were associated only with EV stations. They were however observed on the open-water in other areas around the pond. Of the six plant species found at the NP stations, two were unique to these stations, a deep emergent marsh variety (*Sparganium eurycarpum*) and a shallow emergent marsh plant (*Polygonum amphibium*). These two species were planted in NP stations (City or Toronto, 1995). It should be noted that *Polygonum amphibium* (smartweed) is often classified as a terrestrial plant (D. Metsger, Per. Comm.) and therefore I excluded it from any further analysis.

I was particularly interested in macrophyte-sediment relationships, so only rooted emergent and submergent species were selected for further consideration. Four plants were identified from the areas sampled (*Typha latifolia*, *Acorus calamus*, *Sparganium eurycarpum* and *Potamogeton pectinatus*). Figure 3.5(a) illustrates the relative abundance of these four plant species across the three habitat types. *T. latifolia* and *A. calamus* alternate between dominant and co-dominant growth for the EV stations. While *T. latifolia* is present in the NP stations, it is relatively rare. *A. calamus* and *S. eurycarpum* appear as the co-dominant species for three of the four NP stations. Aquatic macrophytes did not regenerate at NP-2, with the exception of a single *T. latifolia* shoot. *T. latifolia* is exclusively dominant at the Catfish Pond outfall (EV-1). *T. latifolia* is co-dominant with *A. calamus* at EV-3 and NP-1, proximal to Wendigo Creek and Catfish Pond inflow, respectively. Areas where *A. calamus* is dominant (EV-2, NP-3, NP-4) are furthest from the various pond inputs. There may be a correlation with water quality and/or sediment quality controlling the species composition in these various areas (Barko and Smart, 1986). Organic content and tolerance of sediment salinity are possible starting points and will be discussed further in sections 3.4.2 and 3.4.6, respectively.

3.3.4.1 Statistical Analysis: I found it interesting that highest average density of rooted macrophyte shoots occurred within the NP stations (Figure 3.5b). Intuitively, I would have expected the longer timeframes for growth to result in more plants per unit area. However, different plant species (e.g. cattails, sweet flag, great bur reed) may have different optimal shoot densities that dictate the observed distribution patterns (Figure 3.5a). Statistical analysis of the variance in mean shoot densities confirms that shoot density at NP stations is significantly higher than at the EV stations ($p < 0.05$) (Table 3.8).

Figure 3.5a) Relative Abundance of Rooted Emergent and Rooted Submergent Macrophyte Shoots

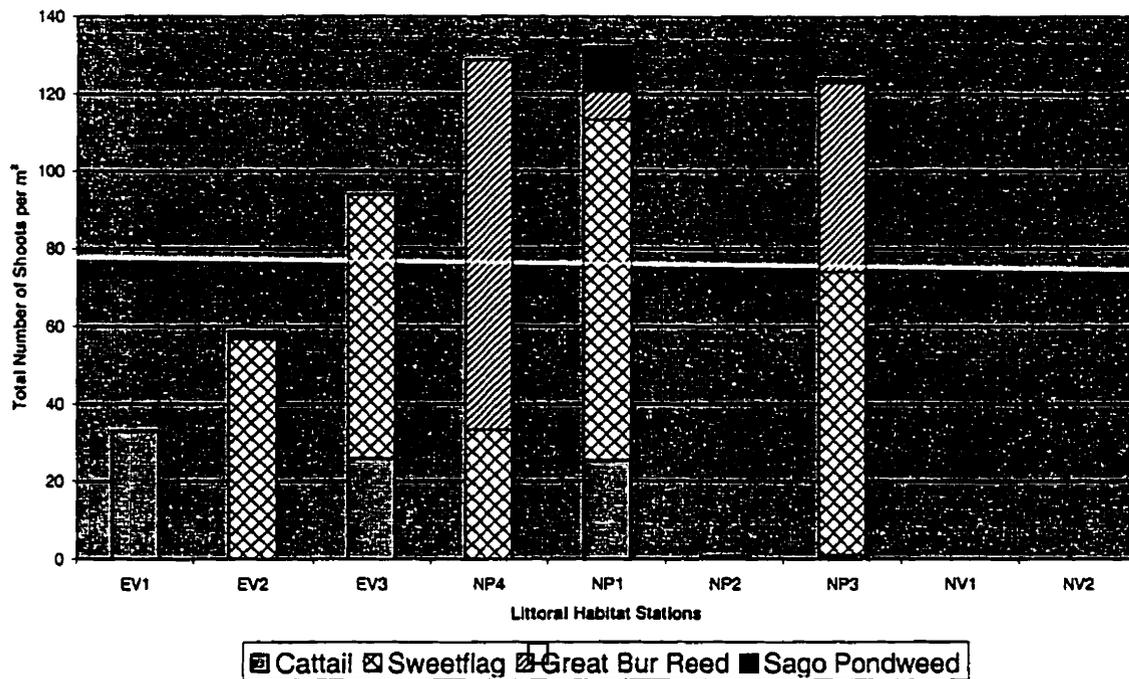


Figure 3.5b) Average Density of Rooted Emergent and/or Rooted Submergent Macrophyte Shoots

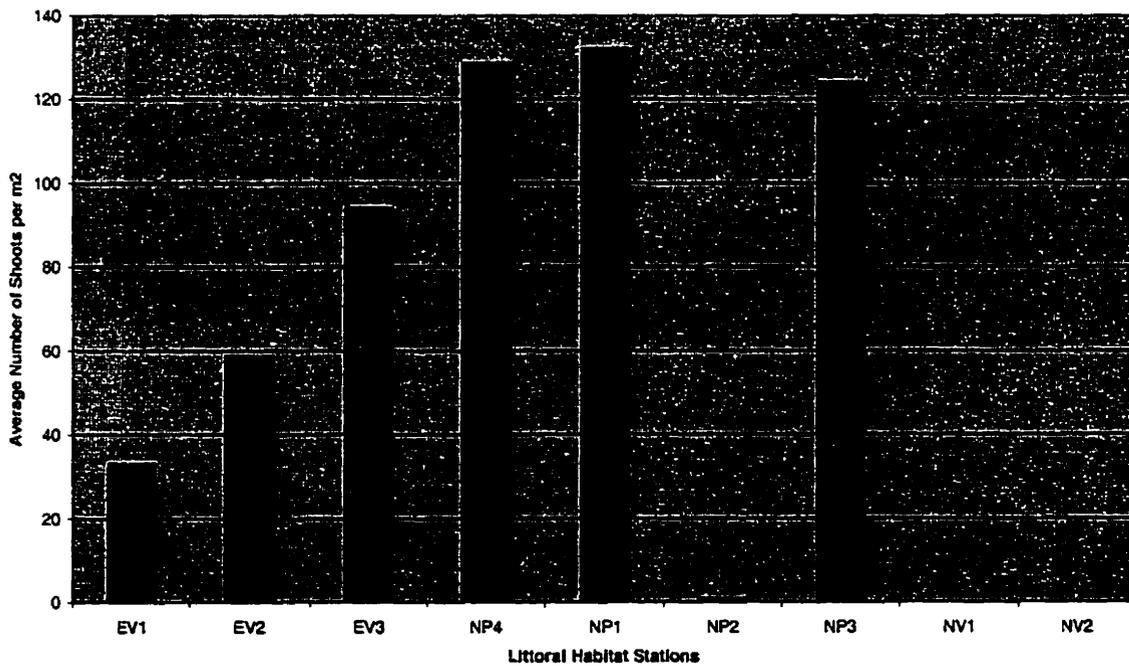


Table 3.8. Summary of T-Test for Rooted-Emergent and -Submergent Macrophyte Shoot Densities*

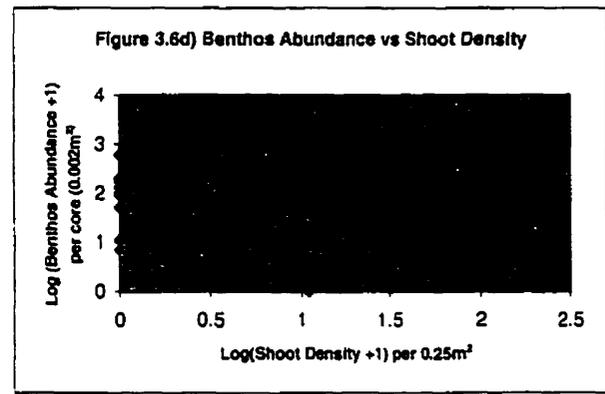
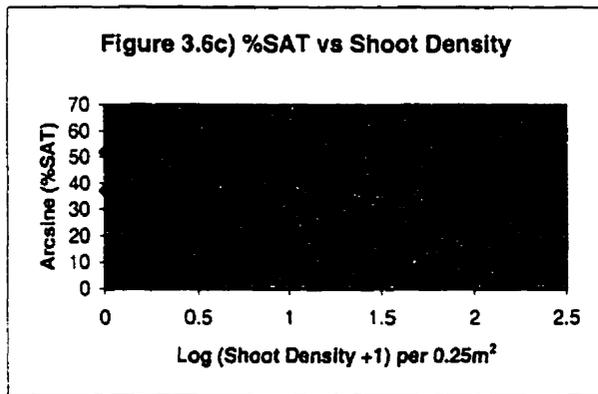
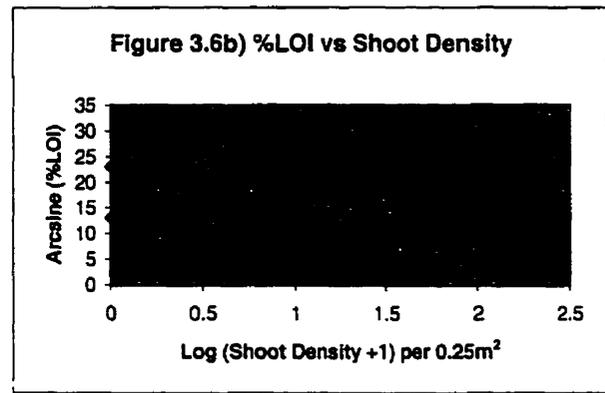
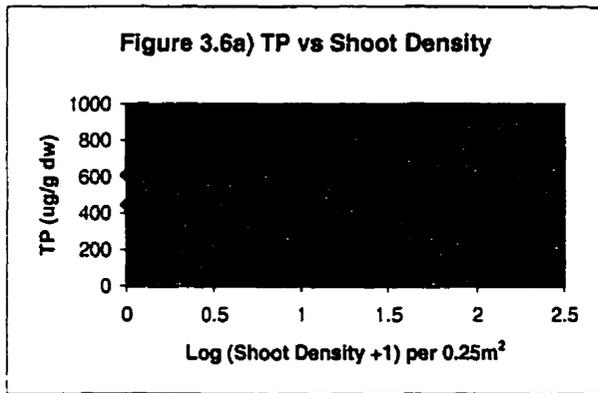
| Parameter | Source of Variance | df | Mean Sq. | F-ratio | P-Value |
|---------------|--------------------------------------|----|----------|---------|---------|
| Shoot Density | Among treatments (ev & np) | 1 | 1440.11 | 4.63 | 0.041 |
| | Among LHS within treatments (ev, np) | 3 | 282.35 | 0.91 | 0.451 |
| | Error within an LHS | 26 | 311.05 | | |

* = Analyses run by SYSTAT 9.0, General Linear Model, Nested Design

3.3.4.2 Regression Analysis: The shoot density and benthic data had to be transformed ($\log(x+1)$) to achieve reasonably normal distributions. Sediment data for TP, %LOI and %SAT was not determinable for the upper 5 cm of sediment. As a result, NV stations were not included for these three regressions. Figures 3.6 (a) – (d) illustrate the results of the regression analyses. TP concentrations are unchanged despite increasing macrophyte density (Figure 3.6a). No obvious correlation with shoot density is apparent for either %SAT or %LOI, which all display a scatter of data points. The apparent negative correlation ($r = -0.57$) between number of benthos and shoot density is highly suspect as the analysis is likely being confounded by the differences in substrate between NV stations (rocks/algae) and EV and NP stations (sediment/macrophytes).

3.3.5 Chloride: Average sediment chloride concentrations per LHS show a clear trend around Grenadier Pond (Figure 3.7). The west shore stations (EV-1 and EV-2) have the highest levels (>300 mg/L aq. extract); the east shore stations (EV-3, NP-1, NP-2, NP-3, NP-4, NV-1) represent moderate levels (91.2 – 190.3 mg/L aq. extract); and the south shore stations (NP-1 and NV-2) have the lowest range in concentration (50.5 – 91.6 mg L⁻¹ aq. extract). The orientation of the “corner” stations (i.e. NP-1 and NP-4) was determined based on where the majority of core samples were taken (west, south or east shore). Statistical analyses were not undertaken with the chloride data in conjunction with other parameters, as the rationale for measuring sediment chloride levels is separate from the rest of the study (see section 3.0). A discussion of observed trends is considered appropriate and adequate.

Proximity to storm water outfalls does not seem to account for the observed trend in sediment chloride concentrations, with perhaps the exception of EV-1, which encompasses the outfall from Catfish Pond. EV-1 has an average chloride concentration of 308 mg L⁻¹ aq. extract and Catfish Pond is known to be



Figures 3.6 (a) - (d): Simple regression analyses with untransformed (TP) and transformed (%LOI, %SAT, Benthos Abundance, Shoot Density) data. There are no strong positive correlations in the data. NP-2 and NP-4 are excluded from all analyses. NV-1 and NV-2 are excluded from Figures (a), (b) and (c) only.

Observed values (squares)
Predicted values (line)

meromictic as a result of salt loadings from road drainage (Zimmerman *et al.*, 1986). *T. latifolia* is dominant or co-dominant where chloride concentrations are highest (EV-1 and EV-2, respectively); *A. calamus* occurs within every chloride range, but is only dominant where chloride concentrations are moderate-high (NP-4, NP-3 and EV-2).

3.3.6 Summary: Average TP concentrations in littoral sediments of Grenadier Pond are lower than those typically found in other shallow, eutrophied ponds. Organic content is within the expected range for eutrophic ponds, while the sediment water content has a larger than expected range. Sediment core concentrations of TP vary significantly within the same type of habitat, but average TP is not any higher in the EV stations than in the NP or NV stations. TP concentrations increased with depth through the cores for all vegetated stations. %LOI varies more between habitat types as does %SAT, with average values of %LOI (both core depths) being lowest at the NV stations, moderate at the NP stations and highest at EV stations. None of the expected correlations were observed between these three parameters with the exception of a positive relation between %LOI and %SAT in the EV stations and, to a lesser extent, NP stations. On its own, %SAT generally behaves as predicted, with the majority of core samples decreasing in %SAT with increasing core depth. Sediment texture appears most variable among the EV stations, but follows the predicted pattern of becoming more consolidated with depth. The benthic invertebrate community shows a distinct abundance of chironomids in the NV stations. Substrate in these NV stations is radically different from that found in either EV or NP habitats, but identification down to species level would be required to properly probe for reasons behind the marked colonization (e.g., algal grazing vs. predacious chironomids). Although no significant difference in total numbers of animals is observed between the EV and NP stations, the relative abundance of selected species is different. Benthic macroinvertebrate diversity is highest within the EV stations and lowest within the NV stations with the NP habitat more closely resembling the EV compliment. Average densities of rooted aquatic macrophytes are highest in the NP stations, but the pattern may be linked more to the plant species than the age class of macrophyte beds. General macrophyte diversity is greatest in EV stations, but rooted-macrophyte diversity is slightly higher in NP stations (4:3). Regression analysis does not reveal any strong correlations between macrophyte shoot density and the four main parameters under study: TP, %LOI, %SAT and benthic macroinvertebrate abundance. Sediment chloride concentrations appear to vary spatially around Grenadier Pond, with highest concentrations along the west shore, moderate levels contained to the east shore and lowest levels along the south shore.

3.4 Discussion

From the literature, aquatic macrophytes are expected to play an important role in littoral zone dynamics and lake recovery from eutrophication. "Use" of macrophytes in Grenadier Pond is an appropriate

biomanipulation strategy (Chapters 1 and 2). Chapter 2 provided details on aquatic macrophyte function and mechanisms of operation, which formed the basis of this recommended strategy. The results presented here suggest that Grenadier Pond's within-sediment and sediment-macrophyte dynamics are not typical of other shallow, eutrophied pond ecosystems bringing into question whether standard biomanipulation approaches will produce predictable outcomes. The remainder of this chapter explores other possible factors, perhaps in combination with rooted macrophytes, that might explain the observed trends in [TP], %LOI, %SAT and benthic macroinvertebrate distribution/abundance in Grenadier Pond. Rooted macrophytes may still influence sediment habitat/structure in this system, but if their role has been marginalized, then rehabilitation results using this technique may be similarly limited.

3.4.1 Total Phosphorus, %LOI and Macrophytes: TP concentrations in the littoral sediment appear to be behaving independently from two factors expected to directly influence TP sediment concentrations: %LOI and Macrophyte Shoot Density. In the original hypothesis, TP concentrations were expected to be highest in the EV stations with a declining gradient to the NP and NV stations (see section 3.0). The fact that mean TP concentrations in Grenadier Pond do not follow this distribution pattern, despite moderate-high shoot densities in the EV stations, introduced a level of uncertainty about our understanding of Grenadier's littoral system dynamics. A paper by Kairesalo and Matilainen (1994) suggests the outer littoral zone (described as the emergent plant- open water interface, and the location of core sampling for this study) functions more as a source of P to the pelagic during the growing season than does the inner littoral zone. They further suggest that net accumulation of "new" sediment also only occurs in the inner-most littoral zone and only during the growing season. Another study (Benoy and Kalff, 1999) agrees with the seasonal entrapment of particulate P and stresses that P will return to the system upon senescence. Perhaps the hypothesis of higher TP concentrations within EV stations should have been contingent on a spatial factor that I did not consider in the development of the original sampling design. Casual observations during sampling, but which cannot be confirmed, suggest that shoot density increases with proximity to the shore, particularly at EV-1. This is not likely to be the case with NP stations of 1-year growth, as macrophyte-bed widths there were substantially smaller than any of the EV stations resulting in little observed difference between the "outer" and "inner" littoral zone.

At times other than the growing season, wind and wave action largely determine the dispersal and distribution of detritus and dissolved and particulate nutrients (Kairesalo and Matilainen, 1994; Barko and Smart, 1986; Chapter 2). Cyr (1998) agrees that wind energy distributes sediment, but contends that in shallow, unstratified aquatic ecosystems, this process of resuspension and redistribution is random and occurs year round. If so, the patchiness of TP and %LOI may be a result of shoreline exposure to varying fetches, particularly wind-weighted fetches. Littoral slope gradient is another physical factor that has been identified as causing local, spatial patchiness in nutrients (Lehmann, 1996). However, the regrading process accompanying plant regeneration provides similar slopes between vegetated sampling stations.

The combined influences of wave action and water quality on sediment-nutrient retention could also be factors affecting TP sediment concentrations. For example, removal of CO₂ by photosynthesis raises water column pH, but the buffering capacity of the sediments creates a steep pH gradient at the sediment-water interface which is only disrupted by wind-induced currents (Drake and Heaney, 1987). The impact of wave action in the littoral is reported as inversely proportional to rooted-macrophyte density (see section 3.0), but some degree of physical disturbance would still occur. This could increase pH at the sediment-water interface and increase P release, even under aerobic conditions (Drake and Heaney, 1987). Perhaps localized patchiness of TP is an indirect measure of P-release caused by fluctuations in pH. It is not known if increases in pH can be related to plant density, but the presence of rooted macrophyte beds do not have a direct effect on P-release rates from sediment (Beklioglu and Moss, 1996).

It is well documented that high total iron (Fe) and magnesium (Mg) in the sediment can influence P-release rates (Sondergaard *et al.*, 1993; Nurnberg and Manning, 1991). Phillips *et al.* (1994) suggests that when iron (III) is present in excess, phosphate will be trapped in the upper oxic layer of sediments of low to moderate organic content. This upper core layer may not have been consistently at a single oxidation-reduction potential or it may have had variable Fe/Mg concentrations, which could contribute to observed distribution patterns of TP measured in Grenadier Pond. A review of the literature did not find any discussion of spatial heterogeneity in Fe/Mg deposits or anoxia

Another possible reason for not observing expected behaviour maybe analytical. Barko and Smart (1986) suggest that TP as a nutrient source to plants varies with sediment density. If TP sediment concentrations were based on volume instead of dry mass, a correlation between TP and macrophyte growth (Barko and Smart, 1986), and perhaps shoot density, might have been observed. However, this seems unlikely since there is no correlation between shoot density and %SAT, which is an estimate of sediment density.

Finally, near-surface littoral sediments in the vegetated stations of Grenadier Pond were enriched with TP relative to the deeper horizon. I take this as confirmation of macrophyte function, as work on other lakes has demonstrated that macrophyte beds serve as "sites of net sediment accumulation over hundreds if not thousands of years" (Petticrew and Kalff, 1992). The study by Drake and Heaney (1987) challenges the singularity of this supposition by postulating, "remobilization due to mineralization reactions taking place in the sediment column may also play a major role" in vertical distribution of P through the sediments. Further study would be required to determine the relative magnitude of these separate processes in P-sediment distribution in Grenadier Pond.

3.4.2 Organic Content and Macrophytes: As predicted, organic content was highest in the EV stations in optimal quantities (10%) for plant growth (Barko and Smart, 1986). No correlation between %LOI and shoot density was observed. The actual source of organic content in the EV stations is likely woody debris and fibrous plant material, found throughout the sediment cores. As in the discussion of TP distribution, wind-wave action is considered. Past data on the prevailing winds on Grenadier Pond suggests they come from the southeast (i.e. coming up from Lake Ontario) (Zimmerman *et al.*, 1986) and therefore, wave action maybe depositing or holding woody debris and detritus along the west shore.

Favourable wind direction becomes a more likely candidate for controlling organic matter distribution (at least on the west shore) if shoot densities are NOT actually increasing over time in proportion to organic matter accumulation. Maximum shoot density (as defined by the number of shoots per unit area) may be pre-determined by spatial "preferences" of specific plant species. Competition for sediment resources or sunlight may have resulted in optimal "spacing" between neighbouring plants, but this may not translate to optimal conditions for particulate entrapment. Kairesalo and Matilainen (1994) have estimated ranges in macrophyte shoot density, according to relative position within the littoral zone, that perform the function of particulate entrapment: 500-700 shoots m^{-2} (inner-mid littoral zone) and 100-300 m^{-2} (outer littoral zone). Average shoot densities in the various LHS were presented in Figure 3.5b. Converted here as a measure of shoots m^{-2} in the outer-littoral zone: 33 – 94 (EV Stations), 4 – 132 (NP Stations), 0 (NV Stations). All the established vegetation sites in Grenadier Pond are inadequate for sediment entrapment and the newly planted areas are only marginally "functional".

3.4.3 Water Saturation, Sediment Texture and Macrophytes: Although it's not entirely consistent, sediment water content (%SAT) does seem to behave as expected, that is, decreasing with increasing core depth. One wonders though, why saturation levels are highest in EV stations, when mature roots are "supposed to" help consolidate sediments (see section 3.0). Cyr (1998) concluded that rooted macrophyte beds in oligo-mestrophic lake littoral sediments have little effect on sediment water content. No correlation between %SAT and shoot density was found for Grenadier Pond. Cyr (1998) contends that water content among cores is quite variable if substrate type is not uniform. Although not statistically analyzed, substrate texture in Grenadier Pond appeared highly variable between cores of replicate sites (Table 3.4). Particle size for these sediments would need to be determined to postulate a causative relationship. There is little empirical evidence to support the statement that macrophyte consolidation of sediments is an outcome of littoral plantings in Grenadier Pond.

3.4.4 Macrobenthic Invertebrates, Macrophytes and Sediment: A clear progression between the three LHS exists for benthic macroinvertebrates: the benthic community of the NP stations more closely resembles the assemblage at the EV stations than those of the NV stations. These results are comparable to

those of Lake Tohopekaliga, Florida (Butler *et al.*, 1992), and provide additional insight to this study's findings. Butler *et al.* (1992) investigated benthic macroinvertebrate colonization of restored macrophyte beds that, as part of the management plan, sustained an extended drawdown period and littoral sediment modification. Invertebrates, associated with both macrophytes and sediments of restored and undisturbed areas, were collected after one and two years of new plant growth.

Butler *et al.* (1992) found that after one year of plant growth, the taxonomic diversities of sediment-dwelling macroinvertebrates between undisturbed (equivalent to established vegetation sites in Grenadier Pond study) macrophyte beds and restored (equivalent to newly planted sites in Grenadier pond) areas were similar (48 and 47 taxa, respectively). Although the number of different taxa found in Grenadier Pond is considerably lower, the relative yields are essentially the same. After two years of plant growth, the number of taxa declined in both areas (undisturbed sites were reduced to 12 taxa, within the same order as found in Grenadier Pond). In terms of invertebrate density, the restored areas were much higher than undisturbed areas for both years. Chironomidae and Oligochaeta together comprised 1/2 of the total number of invertebrates in the restored sites. These same families were also very prominent in the NV stations of Grenadier Pond (approximately 2/3). The restored areas, with highest invertebrate densities, were sandier than the woody-detrital substrates of the undisturbed sites. Although Grenadier Pond substrates were highly variable even among replicate LHS, the EV stations have more woody debris than the NP stations. From their results, Butler *et al.* (1992) made the connection between rooted-macrophytes, sediment texture and benthic invertebrate colonization, which is critical to this study: "firm substrates, such as sand generally harboured increased invertebrate abundance, particularly when interspersed by plant roots". In summary, Butler *et al.* (1992) attributed the observed differences between macrophyte areas to changes and differences in rooted macrophyte densities, sediment texture, concentration of dead vegetation, ratios of air-breathing to gill breathing invertebrates and fish predation.

Other studies have noted the importance of sediment texture, organic content (e.g. woody debris), low sediment disturbance (e.g. areas protected from wave action by rooted plants and/or lake morphology) (Cyr, 1998; Morrisey *et al.*, 1992) and fish predation (Meijer *et al.*, 1989) as controlling factors in the distribution and abundance of benthic macroinvertebrates. Bechara (1996) postulates that flooding, and the associated changes in water quality, has the most profound impact on benthic invertebrates, but during periods of low water (as in the Grenadier Pond study), local environmental conditions like substrate-vegetation variables may be as important as water level. In spite of a lack of correlation between benthic invertebrates and shoot-density, invertebrate distribution and diversity, together with sediment texture, suggest replanting is returning macrophyte function, with respect to macroinvertebrates, to Grenadier Pond.

3.4.5 Macrophyte Colonization: With relatively clear evidence for only one of the four predicted relationships between macrophytes and selected sediment parameters, it becomes necessary to consider the

macrophyte beds themselves as a source of variation. Chapter 2 identified lake bottom area coverage by macrophytes to be 25% to support pike populations large enough to control planktivore populations and up to 50% for plants to become a stabilizing factor for the clear-water state. In 1997, the lake bottom area coverage (emergents and submergents) for Grenadier Pond was 6.42 ha or 5.5% (TRCA, 1997). Macrophyte regeneration sites planted in 1998 would have slightly increased this value, but not by the 20% needed to just meet the minimal recommended coverage. It is possible that, at present, there simply is not enough macrophyte coverage in Grenadier Pond to perform the ecological functions and support the relationships predicted in this study.

3.4.6 Sediment Chloride Concentrations and Macrophytes: Sediment chloride concentrations from other urban ponds were not found in the literature for comparison to Grenadier Pond. Sediment conductivity for saline lakes was found to be in the range of 2.43 to 27.1 mS cm⁻¹ at 25°C, however only water column conductivities are available for Grenadier Pond, and these measurements are not considered equivalent (Hammer and Heseltine, 1988). Chloride concentrations in highway snowmelt (collected along right-of-way) have been measured as high as 9916 mg Cl⁻ L⁻¹ (Buttle and Labadia, 1999). Grenadier Pond is a known receiving water for urban road drainage presumably contaminated with salt from comparable deicing practices. Also, the chloride gradient evident in Grenadier Pond sediment makes within-system comparisons of macrophyte distribution and community composition relevant to a discussion of chloride as a controlling influence.

Hammer and Heseltine (1988) determined that macrophyte species richness decrease with increasing salinity. Increased water column salinity in eutrophic Lake Onondaga, New York, was considered, in large part, responsible for the limited colonization of aquatic macrophyte species (Masden *et al.*, 1993). Along similar lines, sediment chloride concentrations may account for the relatively low species diversity (seven) described in Grenadier Pond in 1998 (see Table 3.7). Of the limited number of macrophyte species identified by Hammer and Heseltine (1988) and Masden *et al.* (1993) as tolerant to significant increases in salinity, four occur in Grenadier Pond (greater than 50% of the total number of species found in Grenadier Pond): *Potamogeton pectinatus*, *Typha latifolia*, *Lemna minor*, *Ceratophyllum demersum* (listed in order of increasing sensitivity). *P. pectinatus*, *T. latifolia* and *L. minor* are even able to survive, to a limited extent, in true saline lakes (> 3g L⁻¹) (Hammer and Heseltine, 1988). It should be noted that the plant species *Scirpus validus* was identified in the TRCA (1998) aquatic macrophyte surveys (1995, 1997, 1998), but was not found within any of the LHS of this study. Several species of *Scirpus spp.*, occurring in western Canada, are considered very salt tolerant (Hammer and Heseltine, 1988).

The only major factor that affects aquatic vascular plants and their distribution in saline lakes is the total concentration of ions in the substrate (Hammer and Heseltine, 1988). Indeed, plant niche along coastal regions is primarily dependent on salinity tolerances in some plants (Haller, 1974). Only after salt-tolerance

is considered does the distribution of viable species seem to become dependent on other factors such as water turbulence, light, nutrients and temperature (Haller, 1974). The extent to which freshwater species follow this pattern was not found in the literature. Although the distribution of macrophytes in Grenadier Pond agrees with Haller's findings, the apparent response to chloride may be indicative of some chloride correlate rather than chloride itself or even to other factors such as sediment texture or susceptibility to waterfowl grazing. To explore these hypotheses, further studies on the Pond would need to be carried out. It is however interesting that only the plants known to be salt-tolerant in Grenadier Pond occupy littoral areas that span the chloride range and clear dominance of *T. latifolia* is coincident with the highest sediment chloride concentrations. Information on the specific salt-tolerance of *A. calamus*, another dominant and widely distributed macrophyte in Grenadier Pond, was not found in the literature.

Previous discussion suggested that physical forces (i.e. wind/wave action) play a strong role in the distribution of some sediment constituents in Grenadier Pond (i.e. TP and %LOI). The spatial pattern in the chloride data supports this argument, as the west shore would be the predicted area of sediment deposition given the prevailing winds from the southeast. This idea assumes that chloride accumulation is independent of other sediment characteristics. In contrast to this assumption, Barko and Smart (1986) positively correlated low conductivity in interstitial water of sediment with high organic content. Although statistical correlation was not performed on the chloride and %LOI data, the opposite relationship seems to be present in Grenadier Pond. Stations with the highest average chloride concentrations coincide with stations showing the highest average %LOI levels (EV-1 and EV-2; Figure 3.7 and Table 3.2a). This may be further evidence that wind/wave action may directly affect the distribution of chloride in sediments and indirectly macrophyte diversity and distribution in Grenadier Pond.

3.5 Conclusions

1. TP concentrations in littoral sediments are generally lower than found in other eutrophied ponds. %LOI is comparable to other eutrophied ponds. %SAT has a wider range than typically found in ponds of low, moderate or high nutrient status.
2. There is no evidence that macrophyte beds (established or newly planted) or organic matter content (%LOI) are significant influences on TP concentration or distribution in the littoral sediment of Grenadier Pond; wind and wave action are more likely to be the main mechanisms of nutrient redistribution in the littoral, since shoot densities are low. The opportunity for aquatic macrophytes to govern nutrient dynamics in the littoral of Grenadier Pond is limited. Other factors, such as pH or Fe/Mg may also be affecting P-release rates and therefore concentrations.

3. Plant density may be pre-determined by inherent spatial preferences of a given species. This serves to undermine (though not eliminate) the hypothesis that increasing organic matter content would necessarily be correlated to high plant density. When compared to estimates of “functionally adequate” shoot densities, all vegetated areas in Grenadier Pond, established and newly planted, appear largely “inadequate”. Although organic material is collecting within the plant beds, the extent to which macrophytes are responsible is unclear.
4. There is no evidence that sediment water content (%SAT) is a result of root-mediated sediment consolidation. The %SAT attributed to the various habitat stations can be explained by sediment texture. Therefore, the lack of correlation between plant density and %SAT does not necessarily reflect impairment of macrophyte function, but more likely the negligible role macrophytes play in Grenadier Pond sediment consolidation.
5. There is evidence that rooted-macrophytes in the NP stations are contributing to the provision of habitat for benthic invertebrates in ways more similar to established vegetation than non-vegetated habitat. Therefore, macrophyte function, in the context of providing benthic invertebrate habitat, appears to be returning to Grenadier Pond.
6. In general, macrophytes in Grenadier Pond appear to play a considerably smaller role in littoral dynamics than is suggested in the literature or indeed measured in other aquatic ecosystems. This appears to be the case because the total lake bottom coverage of macrophytes in Grenadier Pond may be insufficient (5.5%) to provide the desired functional role(s) attributed to littoral macrophyte colonies of greater abundance (25% – 50%). This is not saying that the restoration of littoral macrophytes was an ineffective management technique, but present goals of macrophyte restoration in Grenadier Pond may need to be modified to better reflect currently identified functions, i.e. benthic invertebrate habitat, potential retention of organic material. Alternatively, restoration efforts should continue to focus on creating/extending the littoral habitat and encouraging more extensive macrophyte coverage.
7. There is some evidence to suggest that the chloride concentration in littoral sediments may be a factor in the distribution and community composition of vascular macrophytes in Grenadier Pond, however other factors, such as sediment texture, selective waterfowl grazing or some correlate with chloride may also be influencing plant diversity.

3.6 Recommendations for Future Study

1. To improve the overall study, the sampling design could be modified to include areas of the macrophyte beds most likely to entrap particulate phosphorus and organic material (i.e. inner-littoral) and greater number of core-samples should be retrieved with emphasis on collecting even numbers of sub-samples, however, timing of the field sampling was justified (i.e. August).
2. Chapter 2 discussed an alternate theory suggesting that re-planting macrophytes may have less to do with nutrient "reform" and everything to do with providing a habitat refuge for large bodied zooplankton and subsequent food-web shifts. In light of this chapter's conclusions, it is recommended that future research investigate whether macrophyte beds in Grenadier Pond are functioning as a predation refuge for large-bodied zooplankton or how they might be made to function as such.

3.7 References

- Allan, C., Forsythe, R., and J. Diemer, 1997. *Piedmont N.C. Wet Retention Basins: Performance Factors, Sedimentation Dynamics, and Seepage Losses*, Water Resources Research Institute of the University of North Carolina, Report No. 309
- Barko J.W., Gunnison, D. and S.R. Carpenter, 1991. Sediment interactions with submerged macrophyte growth and community dynamics. In *Aquatic Botany*, 41:41-65
- Barko, J.W. and R.M. Smart, 1986. Sediment-related mechanisms of growth limitation in submersed macrophytes. In *Ecology*, 67(5):1328-1340
- Bechara, J.A., 1996. The relative importance of water quality, sediment composition and floating vegetation in explaining the macrobenthic community structure of floodplain lakes (Parana River, Argentina). In *Hydrobiologia*, 333:95-109
- Beklioglu, M. and B. Moss, 1996. Mesocosm experiments on the interaction of sediment influence, fish predation and aquatic plants with the structure of phytoplankton and zooplankton communities. In *Freshwater Biology*, 36:315-325
- Benoy, G.A. and J. Kalff, 1999. Sediment accumulation and Pb burdens in submerged macrophyte beds. In *Limnology and Oceanography*, 44(4):1081-1090
- Boers, P., Van Ballegooijen, L. and J. Uunk, 1991. Changes in phosphorus cycling in a shallow lake due to food web manipulations. In *Freshwater Biology*, 25:9-20
- Burt, D., 1970. *Platyhelminthes and Parasitism: An Introduction to Parasitology*. American Elsevier Publishing Company, Inc. New York, USA.
- Butler, R.S., Moyer, E.J., Hulon, M.W. and V.P. Williams, 1992. Littoral zone invertebrate communities as affected by a habitat restoration project on Lake Tohopekaliga, Florida. In *Journal of Freshwater Ecology*, 7(3):317-328
- ButtleJ.M. and C.F. Labadia, 1999. Deicing salt accumulation and loss in highway snowbanks. In *J. Environ. Qual.*, 28:155-164

Carignan, R. and J. Kalff, 1980. Phosphorus sources for aquatic weeds: water or sediment? In *Science*, 207:987-989

City of Toronto, 1997, *Plant Species List for Regeneration Sites*. Produced by the Department of Parks and Recreation. Unpublished.

Clarke, Arthur H., 1981. *The Freshwater Molluscs of Canada*. National Museum of Natural Sciences, Ottawa, Canada.

Drake J.C. and S.I. Heaney, 1987. Occurrence of phosphorus and its potential remobilization in the littoral sediments of a productive English lake. In *Freshwater Biology*, 17:513-523

Gartner Lee Limited (GLL), 1996. *Sediment Characterization Program Summary Report Grenadier Pond 1995-1996*. Consultant's Report 95-337, prepared for the City of Toronto, Parks and Recreation Department.

Haller, W.T., 1974. Effects of salinity on growth of several aquatic macrophytes. In *Ecology* 55:891-894

Hammer and Heseltine, 1988. Aquatic macrophytes in saline lakes of the Canadian prairies. In *Hydrobiologia*, 158:101-116

Hellquist, C.B. and G.E. Crow, 1984. *Aquatic vascular plants of New England*, Part 7

Jackson, D. and K.M. Somers, 1990. The spectre of "spurious" correlations. In *Oecologia*, 86:147-151

Kairesalo, T. and T. Matilainen, 1994. Phosphorus fluctuation in water and deposition into sediment within an emergent macrophyte stand. In *Hydrobiologia*, 275/276:285-292

Lehmann, A., 1996. Contribution of GIS to the study of submerged macrophyte ecology in Lake Geneva with a special focus in plant-sediment relationships @ <http://anthropologie.unige.ch/LEBA/Anthony.html>

Malakhov, V.V., 1994. *Nematodes Structure, Development, Classification and Phylogeny*. Smithsonian Institution Press, Washington, USA.

Masden, J.D., Eichler, L.W., Sutherland, J.W., Bloomfield, J.A., Smart, R.M. and C.W. Boylen, 1993. Submersed littoral vegetation distribution: Field quantification and experimental analysis of sediment types

from Onondaga Lake, New York., Technical Report A-93-14, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Meijer, M.L., Raat, A.J. and R.W. Doef, 1989. Restoration by biomanipulation of Lake Bleiswijse Zoom (the Netherlands): first results. In *Hydrobiol. Bull.* 23:49-57

Merritt, R.W. and K.W. Cummins, Editors, 1996. *An Introduction to the Aquatic Insects of North America*. 3rd Edition. Kendall and Hunt Publishing, Iowa, USA.

Morrissey, D.J., Howitt, L., Underwood, A.J. and J.S. Stark, 1992. Spatial variation in soft-sediment benthos. In *Marine Ecology Progress Series*, 81:197-204

Nurnberg, G.K. and P.G. Manning, 1991. Upwards migration of iron and phosphorus compounds in anoxic sediments from the mesotrophic lake on the Precambrian shield. Report prepared for Water Resources Branch, Ontario Ministry of the Environment, Queen's Printer for Ontario.

Smith, C.S. and M.S. Adams, 1986. Phosphorus transfer from sediments by *Myriophyllum spicatum*. In *Limnol. Oceanogr.*, 31(6):1312-1321

Sondergaard, M., Kristensen, P. and E. Jeppesen, 1993. Eight years of internal phosphorus loading and changes in the sediment phosphorus profile in Lake Sobygaard, Denmark. In *Hydrobiologia*, 253:345-356

Page, A.L., editor, 1982. *Methods of Soil Analyses, Part 2, Chemical and Microbiological Properties*. 2nd Edition. American Society of Agronomy, Inc. and Soil Science Society of America, Inc. Madison, Wisconsin, USA.

Pennak, Robert, 1953. *Freshwater Invertebrates of the United States*. The Ronald Press Company, New York.

Petticrew, E.L. and J. Kalff, 1992. Water flow and clay retention in submerged macrophyte beds. In *Can. J. Fish. Aquat. Sci.*, 49:2483-2489

Phillips, G., Jackson, R., Bennett, C. and A. Chilvers, 1994. The importance of sediment phosphorus release in the restoration of very shallow lakes (The Norfolk Broads, England) and implications for biomanipulation. In *Hydrobiologia* 275/276:445-456

Toronto and Region Conservation Authority (TRCA), 1998. *Grenadier Pond, 1998 Aquatic Macrophyte Survey Results*, November. Unpublished.

Toronto and Region Conservation Authority (TRCA), 1997. *Grenadier Pond, 1995 and 1997 Aquatic Macrophyte Survey Results*. Unpublished.

Zimmerman, A.P, Warren, L.A. and S.G. Smith, 1986. *The comparative limnology of Grenadier and Catfish (West) Ponds, High Park, Toronto, Ontario*. Unpublished.

Appendix I: Selection of Littoral Habitat Station Photographs