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EFFECTS OF SHORELINE LOGGING ON THE EPILITHIC ALGAL

COMMUNITY IN SMALL CANADIAN SHIELD LAKES WITH LOGGED

CATCHMENTS

by

Sandra R. Carr

A Graduate Thesis Submitted in Partial Fulfillment of the Requirements for

the Degree of Masters of Science in Forestry

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ABSTRACT

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Key Words: epilithic algae, littoral zone, lakes, Canadian Shield, shoreline logging, water turbulence, wind, nutrients, chl *a*, taxonomy

Chlorophyll *a*, nutritional status and community characteristics of epilithic algae in littoral zones of three small Canadian Shield lakes with logged watersheds were assessed. Sites were chosen with different shoreline treatments, logged or unlogged shoreline forest, to estimate impacts of shoreline logging on epilithic algal communities. All three lakes had logged watersheds, and L42 and L39 had experienced some shoreline logging. No shoreline logging had been done on L26, but two sites were used with characteristics similar to logged and unlogged shoreline forest. Chlorophyll *a* (chl *a*) was used to index algal biomass; carbon (C), nitrogen (N) and phosphorus (P) were used to measure nutritional status; and, community characteristics were evaluated using areal densities of taxonomic classes. Littoral zones were described using a water turbulence index and littoral wind measurements, mass measurements of epilithic material including organic and inorganic components and changes in light (theoretical) and temperature (actual) environments following shoreline logging. The only significant differences in measured variables were found at sites in L39, with higher levels of total and organic material, chl *a*, C and N at the site with logged shoreline forest. Further, Chlorophyceae, Bacillariophyceae, Xanthophyceae and Dinophyceae were higher and Cyanophyceae were lower in L39 at the site with logged shoreline forest. No differences were found between treatment sites in L42, or between sites in L26. Differences found in L39 may have been due more to inter-site variation than to shoreline logging. Overall, data did not show evidence of impacts to epilithic algal communities or littoral environments from shoreline logging, however, results were not conclusive given the preliminary nature and short time-span of the study.

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1 INTRODUCTION

1.1 BACKGROUND

1.1.1 Resource Management Context

Science-based sustainable forestry requires reliable, predictive models of aquatic responses to catchment disturbance (Whitehead and Robinson 1993). Such models will likely be provided by long-term, catchment-scale experiments, complemented by comparative studies of lower precision and explanatory power, but broader spatial relevance. Management of forested land for timber operations is one of the most prevalent land uses in northwestern Ontario. Along with impacts on terrestrial components of the logged landscape, there may also be impacts on wetlands, lakes, and streams. Impacts on aquatic systems may involve changes in the quantity and quality of sediment and nutrient inputs from the terrestrial environment (e.g. Lepisto *et al.* 1995), or greater exposure to wind and sun. Perhaps best understood are impacts from changes in water yield and timing as a result of altered evapotranspiration and drainage within the logged watershed (Bosch and Hewlett 1982; Swift *et al.* 1988). These input changes may be reflected in alterations to the chemical and physical environment, and to biotic components of aquatic systems.

The Province of Ontario developed the Timber Management Guidelines for

the Protection of Fish Habitat (OMNR 1988) and the Code of Practice for Timber Management Operations in Riparian Areas (OMNR 1991) to mitigate undesirable effects of timber management on fish and aquatic habitat. While the Guidelines and Code were designed to be conservative and reliable, they were based on scientific information available in the early 1980s, most of which was collected from the Pacific northwest and eastern deciduous forests of North America. Because the climate, topography, vegetation and geomorphology of northern Ontario are different from these regions, scientific information about land-water linkages in boreal shield ecosystems is required. The use of Ontario's forests is likely to remain a cornerstone of northwestern Ontario's economy, and natural resource planners and managers require reliable, detailed information about the relationships between timber management and aquatic ecosystems. Science-based planning guidelines will enable Ontario's boreal forests to be managed in a way that supports healthy aquatic ecosystems.

The Coldwater Lakes Experimental Watersheds (CLEW) research project was begun by the Centre for Northern Forest Ecosystem Research (CNFER), a research organization of the Ontario Ministry of Natural Resources (OMNR), to address the need for information about the impacts of logging in boreal forests on both aquatic and terrestrial systems. The CLEW research was initiated in 1990 to experimentally evaluate the effects of logging on

boreal lake ecosystems, and to provide information about the effectiveness of shoreline buffer strips in preventing those effects. A modular, multi-scale research approach has been implemented with three main elements designed to measure the responses to timber management of upland and aquatic ecosystems, with and without application of Ontario's standard shoreline buffer strip design:

1. Whole-basin experimental case studies involving intensive monitoring of a small number of headwater lakes and streams before and after timber management;

2. Small-scale study of sediment and nutrient transport to examine sources, dynamics and fate of sediment deposited in aquatic environments associated with timber harvest and site preparation in a number of small upland sub-catchments; and,

3. Comparative analysis of physical and biotic indicator variables in a variety of lakes with different timber management histories facilitating exploratory analysis of spatial and temporal trends in aquatic ecosystem response to timber management (CNFER Unpublished).

1.1.2 Watershed Ecosystem Context

The most important measurement of the terrestrial landscape, insofar as inputs to lakes and streams are concerned, is the catchment area or watershed (Likens and Bormann 1974). The catchment area is defined by the height of land around a lake or bounding a stream. It has been recognized that the catchment is the maximum or potential area of influence, while the actual area of influence is likely smaller than this (Steedman 1988). According to the variable source area concept, the actual area of influence is

dynamic, depending on precipitation, antecedent soil moisture and the size and duration of the precipitation event (Hibbert and Troendle 1988). Thus, the lake or stream is integrated with its catchment area, such that changes in land use lead to changes in the inputs to aquatic systems, thereby having impacts on the receiving water bodies (Likens 1985). This recognition has led to the concept of the watershed ecosystem as the basis for land-use planning, focussing on the watershed or catchment area as a unit of management (Davies 1991, OMEE/OMNR 1993). This management perspective recognizes the need for a more scientific approach to the task of managing our use of natural resources (see Lee 1993 for a full exposition of "adaptive management").

In addition to its relevance to management, the watershed concept provides a natural and tangible way to delimit, study, and classify ecosystems (Lotspeich 1980). The watershed concept is particularly useful to those researchers interested in the aquatic effects of land-use decisions, as at Hubbard Brook (Bormann and Likens 1979), the Experimental Lakes Area (Schindler 1987), and the Columbia River Basin (e.g., Simenstad et al. 1992). The need to predict the aquatic effects of land-use choices has tapped into the relatively new field of science-based (research-based) resource management. In Canada, research projects addressing timber management effects on land-water linkages (e.g., Forest Renewal BC, in Ontario OMNR/CNFER research,

SF-NCE (Sustainable Forests – National Centres of Excellence) and TROLS (Terrestrial and Riparian Organisms, Lakes and Streams) in Alberta, the Prince Albert Model Forest in Saskatchewan, and others) exemplify this.

Within the general concept of the watershed ecosystem, an understanding has developed about the importance of the land-aquatic interface or ecotone. This ecotone must be understood and managed to offset aquatic impacts of watershed land uses (e.g., Platts *et al.* 1987; Naiman and Décamps (eds.) 1990). The structure of the land-aquatic ecotone, riparian or shoreline zone, may be important in mitigating the intensity of effect that logging may have on an aquatic system (Riley & Mohr 1994). Maintaining the natural structure, vegetation and disturbance regime of a riparian or shoreline zone around a lake or stream may be a way of buffering aquatic systems from the acute environmental effects of logging (Ormerod *et al.* 1993).

1.1.3 Research Project Description

My research project investigated the impacts on epilithic algal communities of shoreline logging and is part of the CLEW research program. Specifically, I was interested in the nutritional status, organic and inorganic biomass, and taxonomic composition of epilithic algal communities between sites with intact shoreline forest and logged shoreline forest. Epilithic algae are often the predominant phototrophs in the littoral, or nearshore, zone of oligotrophic

(nutrient-poor) lakes containing few macrophytes where the benthic substratum is rock (Loeb *et al.* 1983; Wetzel 1983; St. Jonsson 1992; Hawes and Smith 1993; see Table 1). In Precambrian (Canadian) Shield lakes, they are important in the energy cycle of the whole lake (Stockner and Armstrong 1971).

This project was designed to evaluate the effectiveness of shoreline buffer strips in mitigating impacts of logging on epilithic algae by examining the algae at sites with different shoreline treatments. Results of this study may influence forest management practice, specifically regarding shoreline buffer strips designed to mitigate changes in littoral habitats and biota.

1.1.4 Biological Context

The littoral (near-shore) zone of a lake is an extremely heterogeneous and potentially productive area, lying at the interface of the terrestrial catchment and the pelagic (open-water) zone of a lake (Wetzel 1983). An important contributor to overall lake productivity, and to the regulation of lake metabolism (Loeb *et al.* 1983, Wetzel 1979 cited in Wetzel 1983), the littoral zone is subdivided into the **eulittoral** (encompassing that area between the highest and lowest water levels, influenced by breaking waves), and the **infralittoral**, which lies under water and is further subdivided into three zones (the upper, middle, and lower littoral) in relation to the distribution of

macrophytic vegetation (Wetzel 1983). For simplicity, "littoral zone" in this document, encompasses both the eulittoral and the infralittoral.

Being an interface between land and open water, the littoral zone may be highly susceptible to changes in inputs from the catchment, particularly in the absence of intact shoreline forest (Brown and Krygier 1970; Gregory *et al.* 1987; Mullen and Moring 1988; OMNR 1991; Ormerod *et al.* 1993; DeNicola and Hoagland 1996; France and Steedman 1996; France 1997; Rask *et al.* 1998). Attached algal communities growing within the littoral zone of lakes are excellent integrators and indicators of such changes in inputs (Fairchild *et al.* 1989, Hansmann and Phinney 1973, Noel *et al.* 1986, Rask *et al.* 1998, Rosemond *et al.* 1993, Shortreed and Stockner 1983). In fact, algae in general have become a much-used biomonitor due to their high turnover rate, which facilitates quick response measurements, their sensitive response to environmental change, and their high species diversity in undisturbed systems (Gregory *et al.* 1987; Round 1991; Dixit *et al.* 1992; Stewart 1995; Dokulil *et al.* 1997; Rask *et al.* 1998).

Due to the sometimes omnivore-rich character of littoral food webs (Lodge *et al.* 1994; Vander Zanden and Rasmussen 1996), attached algae are an important food source for invertebrates (Suren and Winterbourn 1992), snails, crustaceans (Lodge *et al.* 1994), and some littoral fish (France and

Steedman 1996). It is sensible to argue that changes in the attached algal community will have an effect throughout the littoral food web and possibly into the pelagic food web. Despite some researchers' emphasis on the cascading, or top-down, nature of littoral food web interactions (e.g., Lodge *et al.* 1994), the nature of the control is likely more a combination of bottom-up and top-down interactions (e.g., Rosemond *et al.* 1993). As well, although littoral and pelagic food webs are generally uncoupled (France 1995), in small lakes lacking deepwater pelagic forage fish, juvenile lake trout are dependent on littoral production, and thus could be affected by littoral food web changes (France & Steedman 1996).

1.2 LOGGING IMPACTS

1.2.1 Impacts of Logging on Aquatic Systems

After logging, water yield from watersheds generally increases (Bosch and Hewlett 1982; Nicolson 1988; Stednick 1996). Control of yield is largely through evapotranspiration, storage of water within the ecosystem, and the pathways by which water moves through and out of the ecosystem (Bormann and Likens 1979). Storage and movement of water through ecosystems are controlled by living biomass, soil depth and character (i.e. soil water storage and release characteristics), topography, and rainfall (Bormann and Likens 1979; Swift *et al.* 1988). Forested watersheds do not generally contribute a significant amount of overland runoff to streams and lakes, since virtually all

rain infiltrates through the soil and moves horizontally as subsurface flow or vertically to the water table (Swift *et al.* 1988), or is lost through evaporation (Bormann and Likens 1979). Harvesting of large portions of a watershed may change this characteristic. Timber harvesting can effectively enlarge the area of direct drainage adjacent to a lake or stream, increasing the area flushed more rapidly and frequently, particularly on more steeply sloped areas (Hewlett and Troendle 1975 cited in Winter 1985). These changes in water yield and timing have effects on the chemistry of water (organic and inorganic constituents) entering aquatic systems, and on particulate transport from watersheds into aquatic systems as a result of increased erosion of soil and possibly bedrock, and increased nutrient transport (Bormann and Likens 1979).

Although regional factors such as climate, geology, soil, and vegetation affect the details of aquatic ecosystem responses to logging, the following generalizations emerge from the scientific literature:

1. Following the harvest of a forested catchment:
   a) water yield and peak flow increase,
   b) erosion and sediment export increase,
   c) more nutrients are exported from the catchment, and
   d) water temperature may increase in streams and littoral areas of lakes;

2. The severity of these effects is proportional to the amount of land clearance;

3. Improperly designed logging roads and water crossings have

historically been an important source of eroded sediment;

4. Significant reversal of hydrologic effects (item 1, above) can be expected within 2 - 10 years after logging, as vegetation is re-established; and

5. Effects to biota depend on species and life stage, and are poorly understood (Steedman and Morash 2001).

In some regions, these effects are not observed until after soil scarification occurs (Rask *et al.* 1998). Logging may also increase wind exposure, which can increase water turbulence and thereby affect oxygen stratification on very small lakes (Rask *et al.* 1993).

Practical prediction of some aquatic ecosystem responses to logging remains highly speculative and anecdotal. Hydrologic (Stednick 1996), thermal (Brown and Krygier 1970; Steedman *et al.* 1998), and geochemical (Schindler *et al.* 1980; Hornbeck *et al.* 1986) effects are relatively well-understood. Effects on aquatic habitat and biota have been studied less intensively, for a shorter time, and may be inherently more complex and less predictable than physical-chemical phenomena. Further, in higher trophic levels of the food chains (e.g. zooplankton, fish), the biological interactions are very important in controlling the dynamics of populations, which makes the assessment of the effects of catchment manipulations very difficult (Rask *et al.* 1993, Hill *et al.* 1995). In lower trophic levels (e.g. algae, plants), the interaction of top-down and bottom-up effects is important (Rosemond *et al.* 1993, Hill *et al.*

1995), complicating interpretation of catchment manipulations.

Lakes are still under-represented relative to streams in the world literature on forestry effects. However, Finnish lake catchment logging experiments have been reported (Rask *et al.* 1993; Lepisto *et al.* 1995; Rask *et al.* 1998,), and a number of comparative and experimental studies in Canada are now being reported (e.g. Carignan *et* al 2000; Evans *et* al 2000; Scully *et al.* 2000; Steedman and Kushneriuk 2000).

Removal by logging, fire or blowdown of shoreline and upland forest around boreal lakes has been associated with a variety of effects. These include increased wind stress and increased autumn mixing depth (Rask *et al.* 1993; Scully *et al.* 2000), increased phytoplankton and attached algal biomass and production (Rask *et al.* 1993), shifts in phytoplankton community composition (Rask *et al.* 1998), reduced inputs of terrestrial plant material (France *et al.* 1996), and increased littoral insolation and diurnal temperature fluctuations (Steedman *et al.* 1998 and Steedman *et al.* 2001). Other effects of shoreline logging that have not yet been documented for boreal lakes, but can be inferred from stream and river studies, include reduced littoral habitat complexity, and altered nutrient and energy flows (e.g. Lynch and Corbett 1982; Plamondon *et al.* 1982; Meehan 1991). Many studies have reported increased nutrient mobility (e.g. N, P and cations) in disturbed forest soils

and headwater streams (Plamondon *et al.* 1982; Schindler *et al.* 1980; Nicolson *et al.* 1982; Nicolson 1988; Bayley *et al.* 1992). Finnish clearcutting experiments that included scarification have provided evidence of moderate and temporary nutrient enrichment in boreal forest lakes following catchment disturbance (Rask *et al.* 1993; Rask *et al.* 1998). There is good evidence, however, that increased nutrient loadings following forest disturbance such as wildfire or clearcutting depend on lake or catchment morphology (e.g., Schindler *et al.* 1980; Nicolson *et al.* 1982; Nicolson 1988; Rask *et al.* 1993; Carignan *et al.* 2000).

1.2.2 Impacts of Logging on Attached Algae

Studies focussed on the impacts of logging on attached algae were conducted mainly on attached algal communities in streams of the Pacific Northwest (Hansmann and Phinney 1973; Shortreed and Stockner 1983) and New England (Noel *et al.* 1986; Mullen and Moring 1988). The impacts of logging on attached algae in lakes have only begun to be addressed, as in studies from the managed forests of Finland (e.g., Rask *et al.* 1993; Rask *et al.* 1998), and from oligotrophic Canadian Shield lakes in Quebec (Planas *et al.* 2000). Given the importance of logging in northwestern Ontario, the dearth of research into its impacts on aquatic systems in general, and on attached algae in particular, should be addressed.

In general, a switch in attached algal community dominance from diatoms (Bacillariophyceae) to filamentous green algae (Chlorophyceae) was found after clearcut logging of the watersheds of streams in the Pacific Northwest and New England (Hansmann and Phinney 1973, Noel *et al.* 1986, Shortreed and Stockner 1983). If the watersheds were patch-cut or only partially cut, leaving a buffer strip of vegetation along the stream, no difference was found in attached algal community dominance either before or after logging, or in comparison to reference streams (Hansmann and Phinney 1973; Shortreed and Stockner 1983; Noel *et al.* 1986; Mullen and Moring 1988). Increased densities of attached algae have been found in streams draining clearcut watersheds in comparison to reference streams (Noel *et al.* 1986). In the Carnation Creek study on the west coast of Vancouver Island, which compared data from the same stream before and after logging, biomass and accumulation rates of attached algae were, while higher after logging, generally similar to those recorded prior to logging (Shortreed and Stockner 1983).

Where changes in attached algal community composition and biomass or density were found, these changes were associated with increases in light intensity, stream temperature, nutrient concentrations, suspended sediment loads and decreases in oxygen (Hansmann and Phinney 1973; Shortreed and Stockner 1983; Noel *et al.* 1986). All of these factors may have had direct

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effects on attached algal communities by changing available resources and habitat, and indirect effects through changes in nutrient cycling, detrital load and increased decomposition. A New England study (Mullen and Moring 1988) found a negligible effect on attached algae in streams after logging. This stability was attributed to several factors: light intensity was not increased since only about 5% of the riparian canopy was removed, small springs in the area may have helped to maintain a stable thermal regime, and only a small portion of the small, low-gradient watershed was actually logged (Mullen and Moring 1988). Thus, nutrients reaching the stream did not change (Mullen and Moring 1988). A study of Swedish and Antarctic lakes ranging along a productivity gradient found that water temperature was of minor importance in regulating attached algal communities (Hansson 1992). Findings from this study pointed to nutrient availability as a major factor determining attached algal biomass. Although one study of Swedish and Antarctic lakes found that water temperature seems to be of minor importance compared to nutrient availability in determining biomass of attached algae (Hansson 1992), in the case of very low water temperatures (0-4°C) found in Great Bear Lake in the Canadian Arctic, water temperature may become a limiting factor (Moore 1980).

1.2.2.1 Stress Responses and Ecology of Attached Algal Communities

Although there has been little work done on the effects of logging on attached

algae, a fair amount has been done on the effects of other anthropogenic

stresses. For example, responses to acidification, changes in the trophic level

of water bodies, and changes in light levels and UV radiation (e.g., Turner *et

al.* 1987, 1991; Fairchild and Sherman 1993; Marks and Lowe 1993; Rader

and Belish 1997) have been studied. However, demonstrating causality from

field distributional studies is constrained by interacting factors, and the often

poor understanding of the autecology of individual algal species (Cox 1993).

Shifts in species composition are strongly related to, for example, pH and

alkalinity (e.g. Fairchild & Sherman 1993), pH (e.g. Turner *et al.* 1991),

spectral irradiance (e.g. DeNicola *et al.* 1992), or the interacting effects of

nutrient availability and light levels (Marks & Lowe 1993). Changes in

biomass accumulation and density of the attached algae are also often seen

with changes in the aforementioned factors (e.g. Fairchild and Sherman

1993; Toetz 1995). There has been a great deal of difficulty, however, in

separating responses to environmental stress from responses to seasonal

(temporal) variation in environmental conditions. Dramatic resetting

responses to physical factors, notably strong wind events and water level

changes (e.g. Hawes & Smith 1993) also complicate research.

Responses to environmental stress may be more demonstrable in some

systems than in others. For instance, large seasonal variation in species

composition and biomass may not be seen in systems with a high density of

grazers, or where there is asynchrony in several growth-limiting factors (e.g. light and nutrients) such that no single factor has overriding control (Rosemond 1994). In such a system, particularly if there is also a relatively more stable physical environment, a response to environmental stress may be more demonstrable than in a system in which there is a lot of seasonal variation or a relatively less stable physical environment. A situation where herbivore density is high can result in attached algal biomass and productivity remaining relatively constant despite seasonal fluctuations in potentially limiting physical and chemical variables (Rosemond 1994). Something more subtle may be happening, such as a shift from nutrient limitation to light limitation for attached algae as system productivity increases and larger phytoplankton crops increase light attenuation, producing a shading effect (Hansson 1992). Further, water nutrient levels may be less important than sediment or interstitial nutrient levels (particularly phosphorus) to attached algae; although attached algae appear able to alternate between substratum (particularly epiphytic algae) and water as their nutrient source (Hansson 1992). Generally, controls on productivity, biomass and community structure are exerted by more than one factor, such as grazers and nutrients (Rosemond *et al.* 1993), grazers and light (Feminella *et al.* 1989, Hill *et al.* 1995), or nutrients, light and grazers (Hill *et al.* 1992).

The stage of maturation of the attached algal mat may also affect how the community responds to environmental stress. The mature attached algal mat is able to effectively create its own micro-environment influencing conditions such as internal temperature and chemistry (Sand-Jensen 1983), thereby becoming somewhat more resistant to changes in external environmental conditions. At the same time, the mature mat is a competitive environment, with conditions like oxygen, nutrients, and light becoming limiting over the growing season. Attached algal responses to environmental stress will vary depending on time of year, mat development, species composition, type of environment (relatively more stable or more variable), and the stress (Rott 1991). Often, to obscure the situation even more, adaptation, perhaps through morphological variability, rather than shifts in community variables may occur (Rott 1991).

In order to improve predictive ability, and to generalize about the highly variable attached algal communities with accuracy, a research program must be carefully designed. Multiple factors, both abiotic and biotic (as well as their interactions), should be measured. Community structure should be analysed at sufficient taxonomic resolution. Work should be done at spatial scales large enough to account for the community's spatial heterogeneity (cf. Cattaneo *et al.* 1993) and over long periods of time.

On a long-term scale, attached algal communities respond to environmental change primarily by shifts in species composition. However, any attached algal community is a stage in the continuum from freshly settled open substrata to a possible "climax", with the first stages of attached algal development characterized by generalists followed by increasing niche specialists (Rott 1991). It may be that studying the dynamics and structure of attached algal communities may depend on being able to handle each problem without interference from other influencing factors (Roos 1983). However, that approach does not emulate the situation in a natural environment and so may have limited usefulness. In reality, it is likely the interplay of multiple factors that is important as opposed to any single factor. Or, if there is a limiting factor, it may depend on the overall environment or the specific adaptations of species in that community.

1.3 OBJECTIVES

1.3.1 Major Questions

The major questions I will address are:

1. Are there differences in environmental parameters (water turbulence and light exposure) among the littoral zones at logged sites with different shoreline treatments (i.e. logged or unlogged shoreline forest?)

2. Are there differences among epilithic algal communities (with regards to nutritional status, organic and inorganic biomass, and taxonomic composition) at logged sites with different shoreline treatments?

3. If there are differences among sites with regards to environmental parameters and epilithic algal communities, are these differences explained through plausible mechanistic pathways?

1.3.2 Research Objectives

1. To identify different sites according to shoreline treatment in L39 and L42 at the CLEW study area.

2. To quantitatively characterize the littoral zones, according to physical and chemical features, at different sites in the three study lakes (L42, L39, L26).

3. To quantitatively characterize the epilithic algal communities, with regards to taxonomy and density, at the different sites by sampling rock substrates.

4. To characterize the epilithic algal community of the Coldwater Lakes area at sites with intact shoreline forest versus sites with logged shoreline forest.

1.4 VALUE OF WORK

Developing an ability to predict the general effects of clearcutting with

different shoreline treatments on epilithic algal communities will ultimately

be very important to maintaining habitat in boreal forest lakes whose

watersheds are available for commercial logging. Many of these lakes are

oligotrophic and often contain few macrophytes, in which case the epilithic

algae of the littoral zones are the predominant littoral primary producers

(Loeb *et al.* 1983). Epilithic algae in boreal lakes, then, are an essential food

source for invertebrates, small littoral fish, and, indirectly, juvenile pelagic

fish, such that changes in biomass or density of epilithic algae could result in

changes throughout these food webs. Therefore, if logging can be done so as to ease impacts on the epilithic algal component, other components of the littoral zone may not be impacted and suitable littoral aquatic habitat could be maintained.

One of the major benefits of this project will be in improved knowledge of the epilithic algal community; an important, but as yet little understood, component of boreal Shield lakes. Since similar types of research are ongoing in Alberta (TROLS and SF-NCE) and in Saskatchewan (in the Prince Albert Model Forest), exchange of findings among these projects and the Coldwater Lakes study will be broadened by this research. Finally, since research on attached algae is still at a relatively early stage (as compared to research on phytoplankton), further work in the area, done rigorously and with attention to technique, is a contribution to the field in general.

2 METHODS

2.1 COLDWATER LAKES STUDY AREA AND EXPERIMENTAL DESIGN

2.1.1 Coldwater Lakes Study Area Description

The study involves three lakes, L26, L39 and L42, and their catchments located in the Ontario Ministry of Natural Resources' (OMNR) Coldwater Lakes Experimental Watersheds (CLEW) area within the boreal-Great Lakes transition forest on the Canadian (Precambrian) Shield. Located approximately 200 km northwest of Thunder Bay, Ontario and about 70 km northwest of the community of Atikokan, Ontario, the study lakes are in a headwater area draining north into Pekagoning Lake (Figure 1). Soils are shallow with abundant bedrock outcrops, and relief within the catchments is generally not more than about 60 m. Catchment forests consisted of 75 - 100 year-old mixed-wood stands of black spruce (*Picea mariana* (Mill.) BSP.), jack pine (*Pinus banksiana* Lamb.), trembling aspen (*Populus tremuloides* Michx.) and white birch (*Betula papyrifera* Marsh.), with some eastern white cedar (*Thuja occidentalis* L.), red pine (*Pinus resinosa* Ait.) and white pine (*Pinus strobus* L.). These lakes support native populations of lake trout (*Salvelinus namaycush*), white sucker (*Catostomus commersoni*), and 5-7 species of smaller fish species (primarily Cyprinidae, Gasterosteidae, and Etheostomidae).
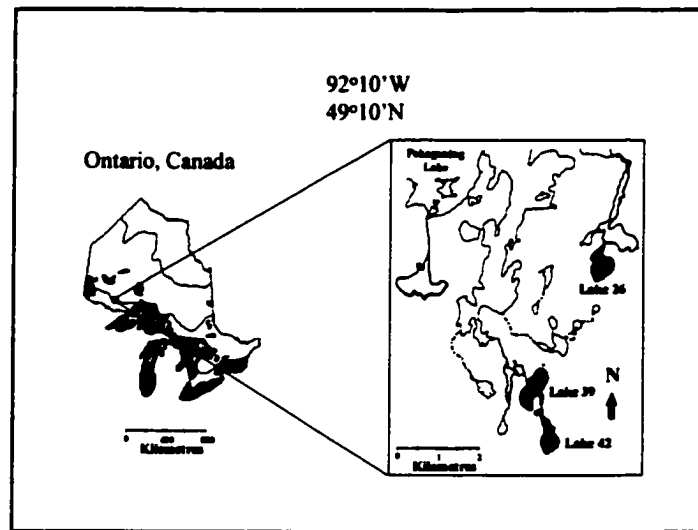
Figure 1. Locations of the three study lakes – L42, L39, L26 - in the
Coldwater Lakes Experimental Watersheds at 49°10'N, 92°10'W in the
province of Ontario, Canada.

Shoreline arboreal vegetation on L26 is dominated by white birch, black

spruce, and eastern white cedar, with some jack pine, and a few occurrences

of red maple (Acer rubrum L.) and white pine (Johnston et al. unpublished

data). L39 shoreline arboreal vegetation was dominated by black spruce with

some white birch and jack pine (Johnston et al. unpublished data). The

shoreline forest of L39 was more diverse than that of L26 with a few

occurrences of red pine, red maple, mountain ash (Sorbus americana Marsh.

and S. decora (Sarg.) Schneid.), mountain serviceberry (Amelanchier

bartramiana (Tausch) M.J. Roem.), and willow (Salix spp.). Johnston et al.

did not sample L42 shoreline vegetation, however, from visual estimations,

the shoreline arboreal vegetation appeared to be dominated by black spruce

and jack pine, with some white birch, and a few occurrences of red and white
pine.

The experimental lakes are small but typical of the deep, clear and
unproductive lakes that support lake trout (*Salvelinus namaycush*) on the
Precambrian Shield of North America. The lakes are dimictic (i.e., mixing
twice per year) and are thermally stratified from May to October. Because of
their relatively large volumes and small headwater catchments, the lakes
tend to have relatively long water renewal times of between 8 and 13 years
(L39 and L26 respectively) (Table 1). Lake surface area ranges from 26 ha
(L42) to 39 ha (L39) (Table 1). Total catchment area varies from 70 ha (L42)
to 194 ha (L39). The ratio of terrestrial catchment area to lake surface area
for the study lakes ranges from 1.6 (L42) to 2.6 (L26). Maximum depth
ranges from 18 m (L42) to 37 m (L26).

The study lakes are extremely dilute, with 1991 - 1995 mean pre-logging
conductivities of 14.4 μS cm^{-1} (L42) to 21 μS cm^{-1} (L26) (Table 1). With regard
to solute and nutrient content, the chemistry of the study lakes is relatively
homogeneous (Table 1). L26 is distinctive in its relatively high Ca
concentrations (2.3 mg L^{-1} compared to 1.3 and 1.6 for L42 and L39
respectively), while L42 has extremely low silicate concentrations (0.03 mg L^{-1} compared to 0.52 and 1.2 for L39 and L26 respectively). Total P lies

between 4 and 6 µg L^{-1} for all three lakes, and total N runs from 0.17 mg L^{-1} for L39 and L26 to 0.23 mg L^{-1} for L42, the pH is between 6.6 (L42) and 6.8 (L26).

Table 1. Physical, chemical, and biotic characteristics of the study lakes.

Experimental treatment	Lake		
	L26 moderate catchment disturbance	L39 intensive catchment and shoreline disturbance	L42 intensive catchment and shoreline disturbance
No. tributary lakes	0	1	0
Surface area (ha)	29	39	26
Max. depth (m)	37	23	18
Lake volume (10^6 m^3)	4.08	4.64	2.15
Total catchment area (ha)	106	194	70
Ratio: catchment area / lake surface area	2.6	1.9	1.6
Water renewal time (years) [a]	13.1	8.2	10.5
% Terrestrial catchment disturbed (1996, 1996+1998)	33, 45	62, 77	71, 74
% Shoreline zone disturbed (1996, 1996+1998)	0, 0	62, 62	42, 61
Estimated min. size of lake trout population[c]	428	265	273
Mean lake trout weight (g)	690	877	848
Total shoreline length (m)	2587	3594	3176
Littoral habitat area (% of lake surface <= 2m depth)	10	8	11
Littoral substrate (% littoral area as: rubble, boulder, bedrock, small inorganic, organic)[d]	6, 65, 18, 0, 10	11, 54, 24, 1, 11	20, 52, 8, 8, 11
Large wood debris (pieces/100 m shoreline[d]	15	14	10
Aquatic vegetation coverage (% littoral area)[d]	2	3	5
Avg. 1991-2000 avg. Sept. thermocline depth (m)	9.0	9.2	9.4
Avg. 1991-2000 May-Sept. Secchi depth (m)	8.0	7.0	6.5
Specific conductance (µS cm^{-1})[e]	21.0	16.4	14.4
DOC (mg L^{-1})[e]	2.2	2.4	2.5
Chlorophyll$_a$ (µg L^{-1})[e]	0.9	1.0	1.2
Silica (mg L^{-1})[e]	1.14	0.50	0.04
Total phosphorus (µg L^{-1})[e]	3	3	5
Total nitrogen (µg L^{-1})[e]	272	230	242
pH[e]	7.1	6.9	6.8

[a]based on runoff of 0.293 m/yr, 1995-1997 average (Beaty 1998 cited in Steedman et al. 2001))
[b]timber harvest on catchments of tributary lakes only
[c]no. unmarked fish encountered 1991-1998 in spawning aggregations
[d] CNFER unpublished data
[e]1991-2000 May-Sept. epilimnetic average

In the first three years after logging these lakes did not show much change in water quality (Steedman 2000) and recent data (to 2001) are consistent with these trends. Long water renewal times meant that only a small portion of each lake's volume was replaced by three years of post-logging runoff (Steedman 2000). The limited water quality changes observed in the study lakes indicated a slight nutrient enrichment after experimental logging, of a magnitude unlikely to induce important changes in lake biota such as phytoplankton or zooplankton (Planas *et al.* 2000; Patione *et al.* 2000).

The littoral zones of the three lakes share similar physical characteristics (CNFER unpublished data). The average length of geomorphologically homogeneous shoreline was similar in all three lakes (average segment length was between 53 and 59 m), suggesting that the lakes have similar spatial "texture" and pattern of shoreline geology and physiography. Lake shorelines were divided into sectors defined by lengths of homogeneous shoreline and littoral habitat. Shorelines and littoral habitats in all of the lakes are comprised mainly of coarse substrates and bedrock, with 10-20 % of the shoreline being organic sediments. In general, there is very little sand (0-6.3 mm diameter), gravel (6.4-75.9 mm diameter) or cobble (76-149.9 mm diameter) in the littoral zones (Table 1). Aquatic vegetation is not abundant, and is generally not found on more than 5 % of the shoreline (Table 1). Large wood debris (pieces 10 cm or more in diameter, and 1 m or more in length)

was common in the littoral zones, with 10-20 pieces per 100 m observed (Table 1). While similar, some differences in littoral substrate composition were found among the lakes. L42 has relatively more sand, gravel, cobble and rubble (150-303.9 mm diameter), and relatively little bedrock compared to the other lakes. The L26 littoral zone is dominated by small boulders (304-609.9 mm diameter), and has very little aquatic vegetation.

2.1.2 Coldwater Lakes Experimental Logging Design

In 1996, after five years of pre-disturbance monitoring, the catchments of L26, L39 and L42 were partially logged (33%, 62%, and 71% respectively) by commercial loggers using a tracked feller buncher, chainsaws and cable skidders. Shoreline forest was removed from about one-half of the shorelines of L39 and L42 (62% and 42%, respectively), while a shoreline buffer of 30-90+ m width was retained on L26 around the entire lake proportional to shoreline slope. The 1996 clearcuts were scarified in July and August 1997 by a skidder pulling chains and barrels. Slash piles around the lakes were burned in late October 1997.

The catchments of one intermittent tributary stream on each of L26 and L39, and a temporary 30 m - wide shoreline buffer on the southwest shore of L42 were left undisturbed in 1996, and harvested two years later in June 1998 bringing the percent of catchment deforested to 42%, 77%, and 74% (in L26,

L39 and L42 respectively), and the percent of shoreline forest harvested up to 61% on L42. The unharvested buffer on the southwest shore of L42 was used for detailed measurements of buffer strip effects on littoral wind energy and shading.

Precipitation, air temperature, wind speed and wind direction were monitored at an upland station east of L26 and on rafts in the middle of L26, L39, and L42. An additional monitoring raft was placed in the buffer strip area on the southwest shore of L42 during 1997 and 1998 to perform detailed monitoring of climatic variables before and after logging of shoreline vegetation. Limnological data were collected at weekly to bi-monthly intervals (depending on season and location), beginning in May 1991, at the deepest point of each lake (Steedman 2000).

2.2 THESIS STUDY DESIGN

My goal was to measure the effects of shoreline logging on the epilithic algal community in lakes with logged catchments, specifically L42, L39 and L26 in the Coldwater Lakes study area, in 1997 and 1998. I chose rock for sampling of attached algae since the predominant substratum of these lakes is rock and there are very few littoral macrophytes. Since the study is concerned with shoreline logging, sites were located in lakes with logged watersheds, but with both logged and unlogged shoreline forest. All study sites were on a

western shore with clean rubble-boulder substratum type (150-609.9 mm diameter rocks). I avoided littoral areas with a high detrital load and very little rock, and areas that were extremely exposed to wind, or were subject to very high sediment input before or after shoreline logging was done.

L42 and L39 had parts of the shoreline forest logged, so I chose sites along both logged and unlogged parts of the shoreline in both lakes (Table 2). The sites with intact shoreline forest on L42 were located on the southwestern shore (shoreline logged in 1998, sectors 18 and 22); sites with logged shoreline forest on L42 were on the northwestern shore (sectors 28 and 30). The site with intact shoreline forest on L39 was located on the northwestern shore (sector 38); the site with logged shoreline forest on L39 was on the southwestern shore (sector 23). L26 functioned as a reference lake since shoreline forest was left intact within at least 30 m of the water's edge during the logging of its watershed. However, two sites were chosen on L26 to contrast undisturbed shorelines with and without shoreline forest. The site without shoreline forest on L26 was located on the southwestern shore (sector 31); site with shoreline forest was located on the northwestern shore (sector 35). The site at sector 31 had a boulder field extending 10 m from the water's edge, while the site at sector 35 had forest right down to the water's edge. I used longitudinal comparisons of logged versus unlogged shoreline sites in both L42 and L39. In addition, because there was a strip of shoreline forest

on L42 left unlogged in 1997, which was then logged in 1998, I was also able

to use a before-after comparison on L42 (Table 2).

Table 2. Summary of experimental contrasts used in epilithic algae study.

Lake Contrasts	Sector	Shoreline Treatment	Years Sampled
L42 Comparison	18 & 22 vs.	unlogged	1997
	18 & 22	logged	1998
	28 & 30	logged	1997 & 1998
L42 Experiment	18 & 22 vs.	unlogged	1997
	18 & 22	logged	1998
L39 Comparison	23	logged	1997
	38	unlogged	1997
L26 Comparison	31	unlogged (boulder field)	1997
	35	unlogged (shoreline forest)	1997

2.2.1 Site Characterization

2.2.1.1 Water Turbulence

2.2.1.1.1 Gypsum Cylinder Preparation and Calibration.

Water turbulence was indexed from erosion of gypsum or Plaster of Paris

(hydrated calcium sulphate) cylinders suspended in the littoral zone of L42

and L39. Gypsum flux was calculated from weight loss combined with the

surface area of the gypsum cylinder. A strong linear relationship between

dissolution rate and water flow allows gypsum weight loss to be used as a

quantitative measure of current speeds, and index of hydrodynamic energy,

or as a qualitative indication of water flow (Petticrew and Kalff 1991).

Cylinders were prepared according to Petticrew and Kalff (1991) and R. Kushneriuk (personal communication). A mixture of Plaster of Paris and water was poured into cylindrical molds (9.5 cm long x 2 cm diameter) and left to set at room temperature overnight. A nylon cable tie used for numbering the cylinders and for suspending them in the water was inserted into the top of each cylinder before the mixture hardened. Once hardened, the cylinders were removed from the molds and oven dried at 40°C for a minimum of 3 hours. The ends of each cylinder were sealed with an oil-based paint to restrict erosion to the circumference of the cylinder.

Osika (1997) calibrated the cylinders in a recirculating flume channel at water velocities up to 12 cm/sec using a water temperature of 16-18°C according to Petticrew and Kalff (1991). Calibration demonstrated a positive linear relationship between cylinder diameter loss rate and water velocity (Osika 1997). Cylinders were left in the flume for up to 36 hours, removed, oven dried at 40°C, and reweighed (Osika 1997). Since the square root of the weight is approximately equivalent to the diameter, weight loss per hour was converted to diameter loss per hour by taking the difference of the square roots of the finish weight from the start weight (R. Kushneriuk personal communication). This conversion allows an estimate of surface area loss to be inferred from weight loss since surface area of newly made cylinders is relatively constant whereas weight is variable. From these calibrations, it

was found that cylinder diameter loss rate could be used to assess the cylinders' response to water flow.

Dissolution rates of gypsum cylinders are differently related to different flow variables in different flow environments (Porter *et al.* 2000). Further, dissolution rates are different in different flow environments (Porter *et al.* 2000). Therefore, it is important to choose a specific flow variable of interest, to determine the flow environment of interest, to choose an appropriate dissolution measurement duration and to establish regression relations that can be used to estimate the flow variable of interest from dissolution measurements (Porter *et al.* 2000). I was interested in turbulence (flow) intensity in what was primarily a mixed-flow environment. The dissolution measurement duration was chosen in order to integrate at least a full day of water turbulence and to ensure that gypsum cylinders remained symmetrical in order to maintain steady dissolution rates. I did not establish regression relations since I used this method qualitatively and did not attempt to quantify turbulence intensity.

2.2.1.1.2 Gypsum Cylinder Deployment.

After initial weighing, gypsum cylinders were hung on tripods and placed at both 0.5 m and 1.5 m depths at sample sites on L42 and L39. For the tripods

at the 0.5 m depth, two cylinders were hung on each tripod and two tripods were placed at each site, one on each side of the site. For the tripods at the 1.5 m depth, two cylinders were hung on each tripod, one cylinder at a depth of about 1.5 m, at the bottom of the water column, and the other cylinder at about 0.5 m depth.

Cylinders were left in the water for at least 24 hours and up to 72 hours. Less than 24 hours would not integrate a full day of water energy; more than 72 hours would have resulted in a reduced rate of weight loss as the surface area of the cylinder wore down and became unsymmetrical. After retrieving the cylinders, they were dried in an oven at 40°C for 48 hours and reweighed to assess weight loss.

2.2.1.2 Wind Monitoring

Wind was monitored from a raft stationed at Sector 22 on the southwest shore of L42 approximately 3 m offshore. An R.M. Young 05103 wind monitor was used mounted 1.5 M above the water surface on a raft. Data loggers sampled sensor readings at 5-s intervals and recorded hourly means and extremes. Littoral wind was monitored from June 3 to September 10 in 1997 and from January 1 to October 31 in 1998. I used data from the whole monitoring period in 1997 and from June 1 to September 30 in 1998.

2.2.1.3 Light Model

A detailed digital elevation model of the L42 catchment and forest canopy was created by merging elevation data from 1:20 000 scale Ontario Base Maps, custom aerial photogrammetry, SPOT 3 (Satellite Pour l'Observation de la Terre) satellite imagery of the 1996 clearcuts, and GPS (Global Positioning System) generated bathymetry and terrestrial feature identification (Steedman *et al.* 2001). Based on this terrain model, L42 summer-average direct solar radiation (kw m^{-2}, May 1 to October 31) was modeled in the ArcView geographic information system. Solar angle and littoral shade loss calculations were based on cloudless skies and simulated removal of an opaque forest canopy 20 m high.

2.2.1.4 Site Profiles

I constructed slope profiles for each study site by measuring depth and distance from shore in metres at three sampling spots at each site. Site profiles are shown in Appendix A.

2.2.2 Epilithic Algal Sampling and Analysis

2.2.2.1 Epilithic Algal Sampling

I sampled rock substrata *in situ* with a syringe brush sampler similar in design to that described by Loeb (1981). The brush sampled an area of 5 cm^2. When sampling, I used SCUBA without fins to minimize disturbance of

epilithic material. I chose rocks that were oriented either parallel to the water surface or east (facing away from the shore) up to an angle of about 10° (assessed visually). Sampling was done every two weeks from early June to late September in 1997 and 1998 in L42, and every four weeks from early June to late September in 1997 in L39 and L26. I sampled more intensively in L42 since that was the main experimental lake for the Coldwater Lakes study.

Three epilithic algal scrapings in 60 ml of lake water were collected from each of the three depths (2.25 to 1.5 m, 1 to 1.5 m, 0.3 to 1m) with as little disturbance of the surrounding material as possible. The three samples from each depth were composited to make one complete sample of 180 ml, representing 15 cm^2 of total rock surface area sampled from each depth. I also collected samples of lake water above rock surfaces once during 1997 to assess contamination of scrapings by water column material or disturbed bottom material. These samples were free of visible macroscopic material. Because of this low level of contamination, I considered there to have been no contribution of algal material from the surrounding water.

Epilithic scrapings were refrigerated with subsequent filtering completed within 48 hours of collection. Sample volumes were first diluted to 300 ml with lake water collected from each site on the sample dates, and

homogenized in a blender to break up larger clumps of material. The liquid sample was kept suspended in a beaker on a stirring plate from which each portion to be filtered was withdrawn. Twenty (20) ml of suspended sample were vacuum filtered through preignited (1 to 2 hours at 550°C) Whatman GF/C filters for particulate carbon (C) and nitrogen (N) analysis, 20 ml were filtered for particulate phosphorus (P) analysis, 180 ml were filtered for organic and inorganic matter content and 60 ml for chlorophyll *a* (chl *a)* analysis. Ten ml was withdrawn from the diluted sample before filtering, and was preserved and stored for taxonomic analysis. After analysis, all measurements were converted to areal concentrations (μg cm^{-2} or mg cm^{-2}). These were calculated from the volume of lake water passed through the filter paper, the diluted sample volume, and the area of lake bottom sampled (i.e. amount on filter divided by fraction of total sample volume passed through filter divided by total area sampled – e.g. 20 mg inorganic material on filter divided by 0.6 (180 ml filtered divided by 300 ml sample volume) divided by 15 cm^2 equals 2.2 mg of inorganic material per cm^2).

2.2.2.2 Epilithic Algal Analysis

2.2.2.2.1 Organic and Inorganic Material.

Mass of dried material was obtained to assess the total amount of material, the amount of organic material and of inorganic material at each site. The

amount of organic material is an indication of not only the amount of epilithic algae (living and dead), but also of material that has settled out from the water column (phytoplankton and zooplankton remains), as well as microscopic plant materials such as pollen, leaves, woody debris, microscopic animals, fungi and bacteria. The amount of inorganic material indicates the quantity of sediment and mineral material that has settled onto the bottom, either from being resuspended and moved into the littoral zone from elsewhere in the lake, or from being washed or blown into the littoral zone from the watershed.

Filtered organic and inorganic matter was dried at 60°C for 24 hours, as described in Aloi (1990), and then weighed to get total dry mass of material on the filter. They were then ignited at 550°C for one hour to remove organics without losing carbonates (Aloi 1990) and reweighed to get a mass of inorganic material. Mass of organic material was calculated as the difference between these two weights.

2.2.2.2.2 Carbon and Nitrogen Analysis.

Particulate C and N were measured to assess the available levels of these nutrients within the epilithic algal community. After filtering, filters for C & N analysis were vacuum dessicated in the dark for at least 24 hours and then stored in the dark at -10°C until analysis. Particulate C and N analyses were

conducted by the laboratory of the Freshwater Institute, Department of Fisheries and Oceans in Winnipeg MB.

Analysis of C and N involved combustion of the filters in an oxygen helium atmosphere at 700°C (Stainton *et al.* 1977) . The combustion products were then swept by helium carrier gas over a heated copper oxide-silver vanadate bed where they were oxidized to CO_2, H_2O, and oxides of N. The gas stream then passed over hot copper where oxides of N were reduced to N_2 and excess O_2 was removed. The remaining CO_2, H_2O, and N_2 in the helium carrier were then measured in a Perkin Elmer Model 240, a stop-flow type CHN analyzer. The gas mixture was introduced into a reservoir where CO_2, H_2O, N_2, and helium were mixed. The base mixture was then diverted through a series of columns that sequentially removed H_2O and CO_2. Carbon dioxide and N_2 were then measured by thermal conductance. C and N values were converted to mg cm^{-2}.

2.2.2.2.3 Phosphorus Analysis.

Particulate phosphorus (P) was measured to assess the available level of this nutrient within the epilithic algal community. Total phosphorus is used as the measure of available phosphorus in the absence of any other more suitable measure of available phosphorus (Turner 1993). Filters for P

analysis were placed damp into glass screw top vials and organic matter was ignited at 550°C to hydrolyze the phosphorus oxides to PO_4. Ignited filters were stored at room temperature until analysis. Particulate P analysis was conducted by the laboratory of the Freshwater Institute, Department of Fisheries and Oceans in Winnipeg MB.

The ignited filter was heated with dilute HCl, which extracts the phosphorus and converts it to orthophosphate. Phosphorus content was determined by a scaled down version of the soluble reactive phosphorus method using automated acid molybdate colorimetry (Stainton *et al.* 1977). P measurements were converted to µg cm^{-2}.

2.2.2.2.4 Chlorophyll *a* Analysis.

Chlorophyll *a* (chl *a*) was measured as an index of algal biomass in the particulate sample. Filters for chl *a* analysis were vacuum dessicated in the dark for at least 24 hours after which they were wrapped in aluminum foil, placed into black plastic film canisters and stored in the dark at -20°C. The quantity of chl *a* was measured using both fluorescence and reverse phase high performance liquid chromatography (hplc). Gross fluorescence overestimates chl *a* by measuring to both chl *a* and chlorophyll degradation products. The hplc procedure specifically measures chl *a*. Doing both

procedures obtains measures of both viable and nonviable algal pigments.
Chlorophyll *a* analysis was conducted by the laboratory of the Freshwater
Institute, Department of Fisheries and Oceans in Winnipeg MB.

The particulate matter collected on the GF/C filters was extracted in a
mixture of 68% methanol, 27% acetone and 5% distilled deionised water in
the dark at 4°C for approximately 16 hours. After extraction, the extract was
refiltered using an Acropore filter. A Shimadzu RF-551 PC
Spectrofluorometer Detector was used for gross fluorescence analysis. It was
equipped to provide excitation of the chl *a* molecule at around 440 nm and to
detect emission at around 660 nm. Chlorophyll *a* from *Anacystis nidulans*
(Sigma Chemicals) was used as a source of known concentration to calibrate
the fluorometer. The gross fluorescence and hplc chl *a* measurement
procedures were done using a LKB 2150 dual piston hplc pump and a
Shimadzu automatic hplc sampler injector. Data were collected and
calculated on a 486PC with JCL 6000 Chromatography data system, version
5.06. For hplc chl *a* measurements a Waters C18 Guard Pak pre-column and
a Waters Resolve 5μ spherical C18 reverse phase column were used. Chl *a*
measurements were converted to areal concentrations of μg cm^{-2}.

2.2.2.2.5 Taxonomic Analysis.

Preserved subsampled suspensions of epilithic algal scrapings were used for taxonomic analysis to quantify living algae. Algal subsamples were stored in glass vials, preserved in 0.3 ml of Lugol's solution and 0.2 ml of 10% formalin and refrigerated in the dark at 5°C. Algal enumeration and identification were done using a Leitz Diaplan research microscope under 200X magnification. I used a Sedgewick-Rafter counting cell, which measures 50 mm x 20 mm x 1 mm and holds exactly 1 ml of sample. Areal measurements of algal cells were recorded in one strip across the cell and approximated to known shapes to obtain a value in square micrometers for each taxon. This value was then multiplied by the microscope factor (18.35) to project values in μm^2 per ml for each taxon (Hopkins and Standke 1992). The microscope factor is obtained by dividing the total volume of the counting cell by the volume of one strip across the cell. Total μm^2 ml^{-1} were converted to areal densities by multiplying by 300 (diluted sample volume in ml) and dividing by 15 (total sampled area in cm^2). Abundance was expressed as μm^2 of algae per cm^2 of sampled area.

Areal abundance measurements should be viewed in relative terms rather than as absolute measurements. It is likely that many cells were obscured by clumps of material not broken up in the homogenization process used, thus not all living cells would have been identified and measured.

Taxonomy references used were Bourrelly (1968 and 1972) for

Chlorophyceae, Xanthophyceae, Dinophyceae and Chrysophyceae; Komarek

and Anagnostidis (1986 and 1989) and Anagnostidis and Komarek (1988) for

Cyanophyceae; and the PIRLA (Anderson *et al.* 1984-1986) collection of

diatom iconographs for the Bacillariophyceae. Prescott (1978) and Findlay

and Kling (1976) were used for general references and identification.

2.2.3 Data Analysis

In this study, lake treatments were not replicated, and, therefore, they did

not estimate between-lake variance associated with forestry impacts, i.e., the

responses of each lake to its forestry treatment were analyzed and

interpreted independently of the other lakes. However, inter-annual

variability estimates for L42, sectors 18 and 22, before and after shoreline

zone logging, were provided by the multi-year study design. Although L39

and L42 might appear to represent replicates of the shcreline logging

treatment, L42 is tributary to L39, and, therefore, the lakes are not

hydrologically independent.

Pre-logging and post-logging means and differences for masses of material,

particulate concentrations of nutrients and algal chl *a* and for algal densities

were calculated by repeated-measures analysis of variance (RM-ANOVA)

(SPSS Release 10.1.0 Standard Version), with shoreline logged and unlogged as "independent" cases comprising June to September monthly means or observations. Monthly means were made up of biweekly observations in L42 and monthly observations in L39 and L26. Since no differences were found between values at different depths, depth data were combined for each site and sampling date. These data were neither highly nor consistently skewed, and were not transformed prior to analysis. Seasonal averages within treatments and 95% confidence limits are displayed. Differences are recorded along with 95% confidence limits. Differences are considered significant where these confidence limits do not span zero. Bonferroni adjustment was used for all post-hoc comparisons.

Differences in the rate of gypsum cylinder erosion (indexing water turbulence) from L42 and L39 were assessed using box-whisker plots to show central tendency, data distribution and treatment effects. Littoral wind direction data from sector 22 in L42 were lumped into 8 major directions using azimuths and reported according to frequency of occurrence. Littoral wind velocity data from sector 22 in L42 were lumped into 8 major directions as with the wind direction data. Hourly average velocities were averaged over the whole recording season for 1997 (June 3 to September 10) and from June 1 to September 30 for 1998 to compare pre- and post-logging shoreline conditions.

3 RESULTS

3.1 LITTORAL WIND DIRECTIONS AND VELOCITIES FOR L42, SECTOR 22

In 1997, before shoreline logging but after watershed logging, littoral wind direction at sector 22, on the southwestern shore of L42, was predominantly from the west (about 32% of the time) (Figure 2). After shoreline logging in 1998, wind was predominantly from the southwest and the south (about 25% and 20% of the time respectively) (Figure 2).
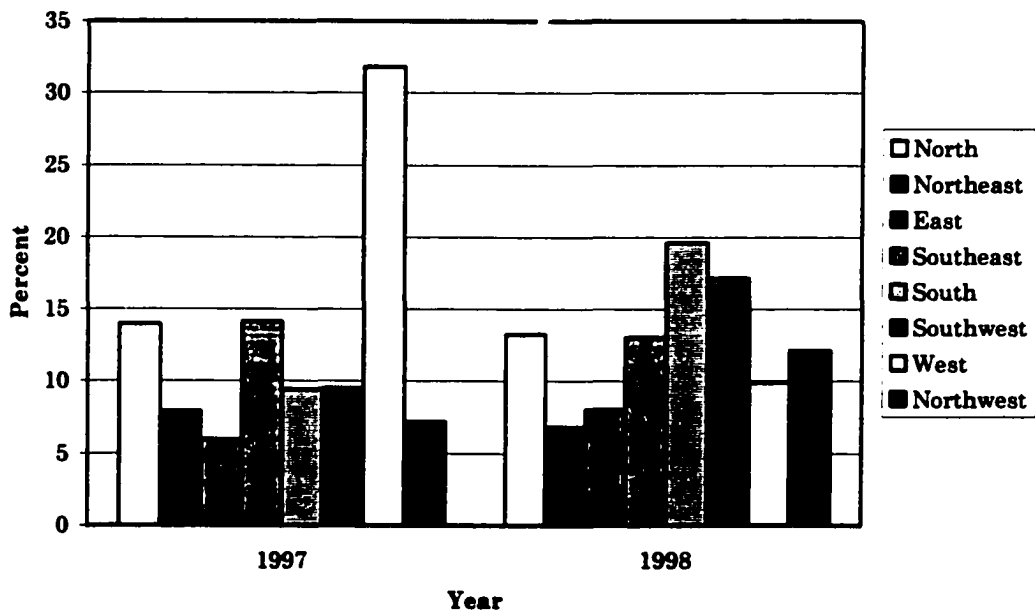
Figure 2. Littoral wind directions expressed as frequency percentages for sector 22 L42, June 3 through September 10, 1997, before shoreline logging, and June 1 through September 30, 1998, after shoreline logging.

After shoreline logging in 1998, average littoral wind velocities rose from the

southeast, south, southwest, west, and the northwest, doubling or almost

doubling relative to 1997 values, before shoreline logging (Figure 3). From

the southeast, south and the northwest, average velocities rose, respectively,

from about 1.3, about 0.9 and about 0.7 m sec^{-1} in 1997 to about 2.4, about 1.6

and about 1.4 m sec^{-1} in 1998. Average wind velocities from the southwest

and the west rose, respectively, from about 0.4 and about 0.4 m sec^{-1} in 1997

to about 0.8 and about 0.7 m sec^{-1} in 1998.
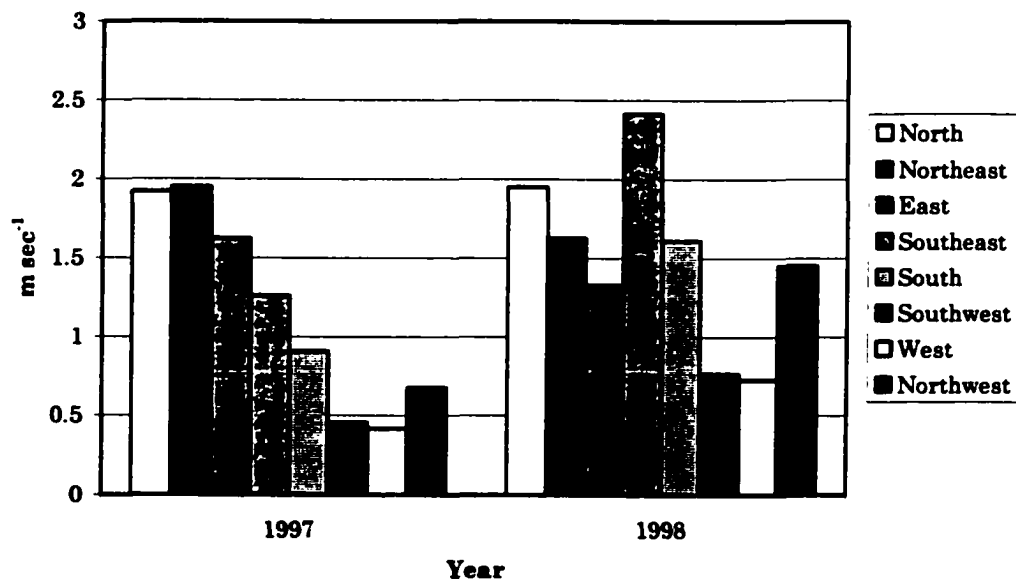
Figure 3. Mean littoral wind velocities (m s^{-1}) near Sector 22 in L42 from
June 3, 1997 to September 10, 1997, before shoreline logging, and from
June 1, 1998 to September 30, 1998, after shoreline logging. Velocity
means are based on hourly readings and are presented by wind direction
categories from North to Northwest derived from azimuths.

3.2 WATER TURBULENCE IN L42 AND L39

Water turbulence was measured after watershed logging, so references to logged and unlogged sites apply to the status of shoreline forest. Overall littoral water turbulence as indexed by gypsum cylinder diameter loss per hour was the same at logged and at unlogged sites in L42 (median diameter loss of 0.014 for unlogged and 0.015 for logged sites) and L39 (median diameter loss of 0.019 for unlogged and logged sites) (figures 4a and 4b). Shoreline water turbulence was the same at logged and at unlogged sites in both L42 and L39 (median diameter loss of 0.015 at unlogged sites and 0.016 at logged sites in L42; 0.020 at the unlogged site and 0.021 at the logged site in L39) (figures 4c and 4d).

Water turbulence about 2 m offshore was higher at the top of the water column than at a depth of 1.5 m (close to the lake bottom) in both L42 (median diameter loss of 0.012 at the bottom and 0.016 at the top) and L39 (median diameter loss of 0.012 at the bottom and 0.019 at the top) (figures 4e and 4f). However, distributions were still slightly overlapping for both lakes. Further, turbulence was higher at the top of the water column at logged sites than at unlogged sites in L42 (median diameter loss of 0.013 at unlogged sites and 0.017 at logged sites), but there was no difference between treatment sites in L39 (median diameter loss of 0.019 at the unlogged sites and 0.018 at logged sites) (figures 4g and 4i). At the 1.5 m depth, water

turbulence was the same at logged and unlogged sites in both L42 (median diameter loss of 0.011 at unlogged sites and 0.013 at logged sites) and L39 (median diameter loss of 0.013 at unlogged sites and 0.011 at logged sites) (figures 4h and 4j). Distributions were overlapping for both water column positions in both L42 and L39.
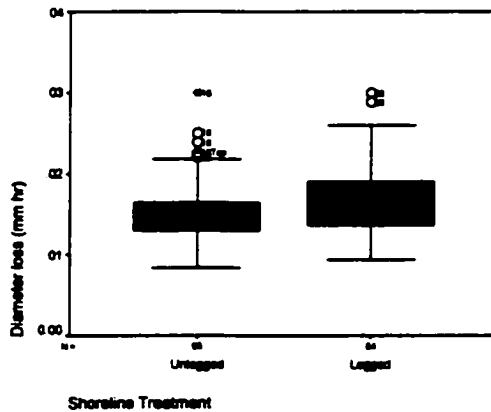
Figure 4a. Gypsum cylinder erosion in L42 at sites within different shoreline treatments.
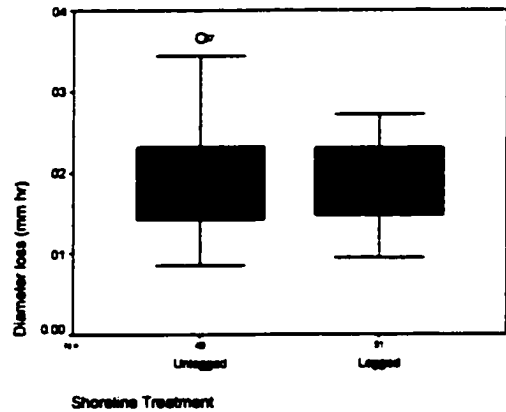
Figure 4b. Gypsum cylinder erosion in L39 at sites within different shoreline treatments.
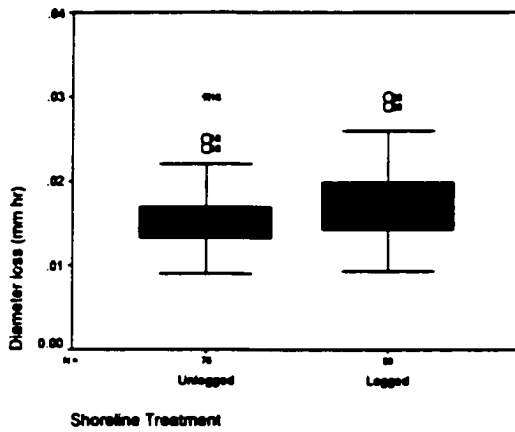
Figure 4c. Gypsum cylinder erosion in L42 at sites within different shoreline treatments at the shoreline.
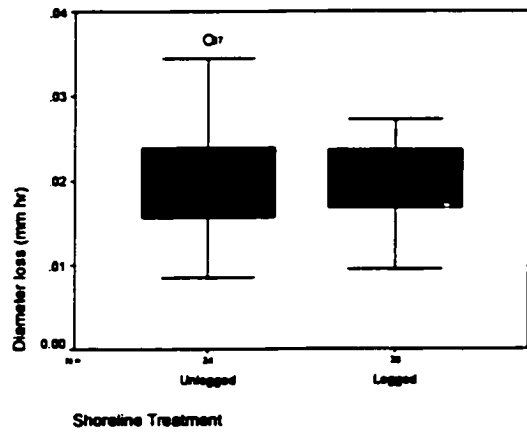
Figure 4d. Gypsum cylinder erosion in L39 at sites within different shoreline treatments at the shoreline.

Figure 4. Gypsum cylinder erosion of diameter loss (mm hr^{-1}) as an index of water turbulence at sites in L42 and L39 from July to September 1997. Boxplots are based on median, quartiles and extreme values. The box represents the interquartile range that contains 50% of values. The whisker lines extend from the box to the highest and lowest values, excluding outliers. The line across the box indicates the median.

Figure 4 (continued)
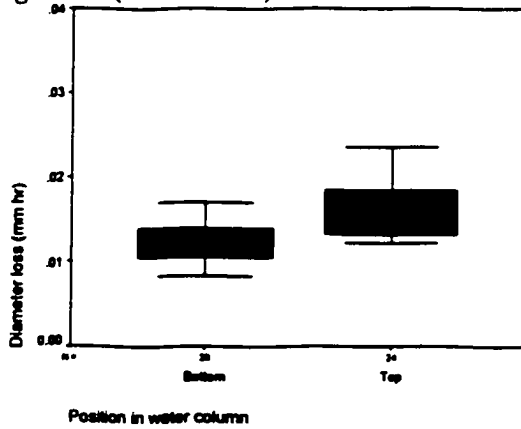
Figure 4e. Gypsum cylinder erosion in L42 at two positions in the water column, about 2 m offshore.
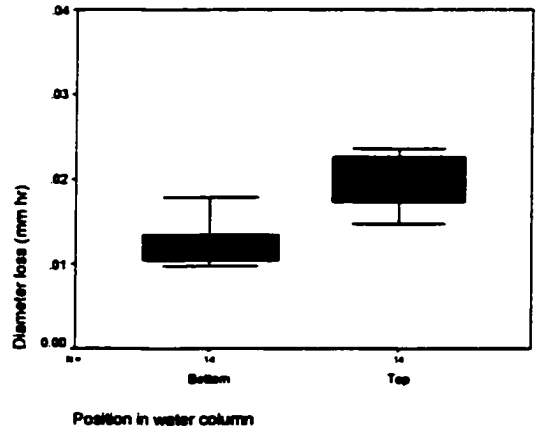
Figure 4f. Gypsum cylinder erosion in L39 at two positions in the water column, about 2 m offshore.
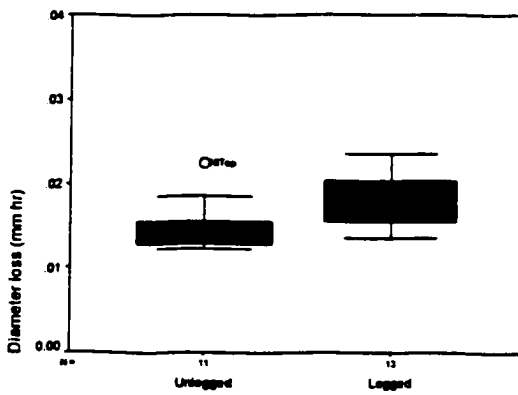
Figure 4g. Gypsum cylinder erosion in L42 at sites within different shoreline treatments at the top of the water column, about 2 m offshore.
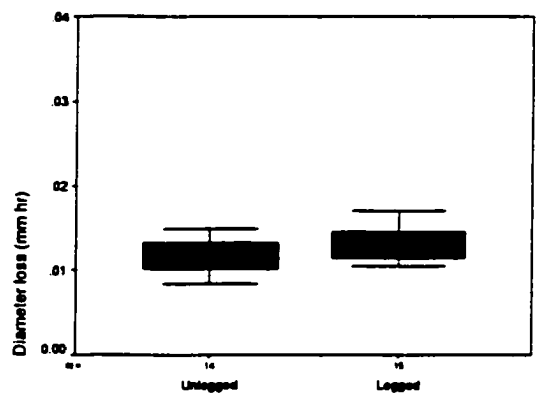
Figure 4h. Gypsum cylinder erosion in L42 at sites within different shoreline treatments at the 1.5 m depth, about 2 m offshore.
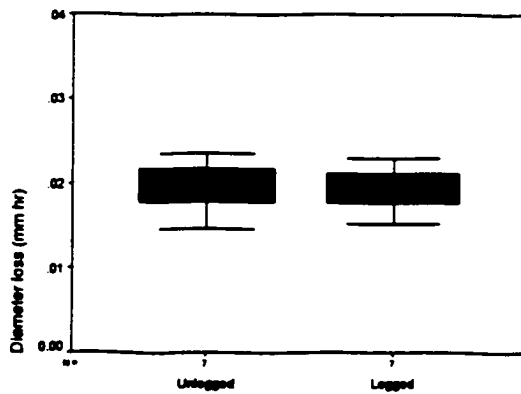
Figure 4i. Gypsum cylinder erosion in L39 at sites within different shoreline treatments at the top of the water column, about 2 m offshore.
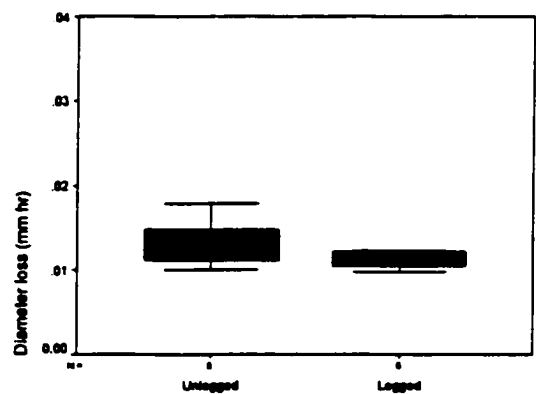
Figure 4j. Gypsum cylinder erosion in L39 at sites within different shoreline treatment sites at 1.5 m depth in the water column, about 2 m offshore.

3.2.1 Wind and Water Turbulence in L42, sector 22

No direct relationship was found between littoral wind velocity and gypsum cylinder diameter loss (indexing water turbulence) on L42 at Sector 22 (no data shown). Littoral wind velocity data were integrated over both 24-hour and 48-hour periods and wind directions were selected with the longest fetch. Wind data were compared with all gypsum loss data together, or separately by cylinder position (i.e. shoreline, offshore surface, offshore bottom). Nonetheless, there is evidence that cylinder diameter loss rate is positively related in a linear fashion to water velocity (Osika 1997, Petticrew & Kalff 1991) provided certain prerequisites are observed (Porter *et al.* 2000). Cylinders showed increased rates of erosion when placed at a depth of 0.5 m, where water turbulence would be expected to be higher, than at a depth of 1.5 m, where water turbulence would be expected to be lower.

3.3 BIOMASS AND NUTRIENTS

3.3.1 Mass of Organic and Inorganic Material

There was no difference in total, organic, inorganic or the ratio of organic to inorganic material in L42, either considering the whole lake or just sectors 18 and 22, before or after shoreline logging (Table 3). In L39, there was more total epilithic material and more organic material at cut sites than at uncut sites. There was no difference in the mass of inorganic material between the

cut and the uncut sites. There was no difference in biomass of material in

L26 between the two stations sampled.

Table 3. June to September total, organic, inorganic mass of epilithic material for L42 (1997 and 1998), L39 (1997) and L26 (1997), and for L42, sectors 18 and 22 before shoreline logging (1997) and after shoreline logging (1998). All masses are recorded in mg cm^{-2}. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. Asterisks identify significant differences where the 95% confidence interval does not span zero.

Site Contrasts	Index	Mean pre-logging value	Mean post-logging value	Mean difference after logging
L42		n=6	n=12	------
	Total Material	8.00 (6.01, 9.98)	7.75 (6.35, 9.16)	-0.25 (-2.68, 2.19)
	Organic Material	5.48 (4.56, 6.40)	4.86 (4.21, 5.51)	-0.62 (-1.74, 0.51)
	Inorganic Material	2.52 (1.33, 3.71)	2.89 (2.05, 3.73)	0.37 (-1.09, 1.83)
L42, s18&22		n=6	n=6	------
	Total Material	8.00 (5.85, 10.14)	7.42 (5.27, 9.57)	-0.58 (-3.61, 2.46)
	Organic Material	5.48 (4.32, 6.63)	4.95 (3.80, 6.10)	-0.53 (-2.16, 1.10)
	Inorganic Material	2.52 (1.49, 3.55)	2.47 (1.44, 3.50)	-0.05 (-1.50, 1.40)
L39		n=3	n=3	------
	Total Material	3.25 (1.84, 4.66)	5.94 (4.53, 7.34)	2.69 (0.70, 4.68)*
	Organic Material	1.91 (1.14, 2.69)	3.58 (2.80, 4.36)	1.67 (0.57, 2.77)*
	Inorganic Material	1.34 (0.61, 2.07)	2.36 (1.62, 3.09)	1.02 (-0.02, 2.05)
		Mean value Sector 31	Mean value Sector 35	Mean difference between sectors
L26		n=3	n=3	------
	Total Material	5.65 (3.57, 7.73)	5.35 (3.27, 7.43)	-0.30 (-3.24, 2.64)
	Organic Material	3.28 (2.56, 4.00)	3.61 (2.89, 4.32)	0.33 (-0.68, 1.34)
	Inorganic Material	2.38 (1.00, 3.76)	1.74 (0.36, 3.12)	-0.64 (-2.59, 1.32)

Note: *Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.*

3.3.2 Algal Chlorophyll a

There were no consistent responses regarding changes in algal chlorophyll a

(chl a), used to index algal biomass, between logged and unlogged sites on

L42 and L39 (Table 4). Total chl *a* was significantly lower in L42 at sites that had experienced shoreline logging than at sites with intact shoreline forest. Total chl *a* was lower after shoreline logging at the experimental sectors (18 and 22) on L42, but the difference was not significant. However, in L39, the study site that experienced shoreline logging had significantly higher total chl *a* than the unlogged site. HPLC measurements, which measure living chl *a* concentration, were significantly higher in L39 at the logged study site than the unlogged site, but there were no significant differences in L42 between logged and unlogged sites.

Table 4. June to September epilithic total chlorophyll *a* (including chl *a* degradation products), HPLC chl *a* (only chl *a*) for L42 (1997 and 1998), L39 (1997) and L26 (1997), and for L42, sectors 18 and 22 before shoreline cutting (1997) and after shoreline cutting (1998). All concentrations are reported in μg cm^{-2}. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. Asterisks identify differences where the 95% confidence interval does not span zero.

Lake	Index	Mean pre-logging value	Mean post-logging value	Mean difference after logging
L42		n=6	n=12	------
	Chl *a*	3.03 (2.54, 3.52)	2.25 (1.90, 2.60)	-0.78 (-1.38, -0.18)*
	HPLC	1.56 (-1.17, 4.28)	0.55 (-1.38, 2.48)	-1.01 (-4.34, 2.33)
L42, s18&22		n=6	n=6	------
	Chl *a*	3.03 (2.42, 3.64)	2.25 (1.64, 2.86)	-0.78 (-1.64, 0.09)
	HPLC	1.56 (1.11, 2.01)	1.76 (1.31, 2.21)	0.20 (-0.43, 0.84)
L39		n=3	n=3	------
	Chl *a*	0.58 (0.28, 0.87)	1.09 (0.79, 1.38)	0.51 (0.09, 0.92)*
	HPLC	0.40 (0.20, 0.60)	0.75 (0.54, 0.95)	0.35 (0.06, 0.63)*

		Mean Value Sector 31	Mean Value Sector 35	Mean difference between sectors
L26		n=3	n=3	------
	Chl *a*	1.02 (0.63, 1.42)	1.10 (0.71, 1.49)	0.07 (-0.48, 0.63)
	HPLC	0.59 (0.34, 0.85)	0.71 (0.45, 0.96)	0.12 (-0.25, 0.48)

Note: *Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.*

3.3.3 Nutrients

There were no consistent responses regarding changes in particulate carbon (C), nitrogen (N), and phosphorus (P) between logged and unlogged sites on L42 and L39 (Table 5). Both C and P were significantly higher at the logged study site on L39 than at the unlogged site, but there was no significant difference between sites for N. There were no significant differences in C, N, or P between logged and unlogged sites on L42, or before and after shoreline logging at sectors 18 and 22 on L42, or between sites on L26.

N:P ratios were lower at logged sites on L42 and after shoreline logging at sectors 18 and 22 in L42 (Table 5). However, N:P ratios were higher at logged sites than at unlogged sites in L39. There was no difference in N:P ratios between sites in L26.

Table 5. June to September epilithic carbon (C), nitrogen (N) and phosphorus (P) for L42 (1997 and 1998), L39 (1997) and L26 (1997), and for L42, sectors 18 and 22 before shoreline logging (1997) and after shoreline logging (1998). Carbon and nitrogen are recorded in mg cm^{-2}, and phosphorus is recorded in μg cm^{-2}. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. Asterisks identify differences where the 95% confidence interval does not span zero.

Lake	Index	Mean pre-logging value	Mean post-logging value	Mean difference after logging
L42		n=6	n=12	------
	C	2.57 (2.13, 3.01)	2.16 (1.85, 2.47)	-0.41 (-0.95, 0.13)
	N	0.21 (0.17, 0.25)	0.17 (0.14, 0.19)	-0.05 (-0.10, 0.001)
	P	6.29 (2.88, 9.71)	6.04 (3.63, 8.46)	-0.25 (-4.43, 3.93)
	N:P	33:1	28:1	
L42, s18&22		n=6	n=6	------
	C	2.57 (2.02, 3.11)	2.24 (1.70, 2.78)	-0.33 (-1.09, 0.44)
	N	0.21 (0.17, 0.26)	0.18 (0.13, 0.22)	-0.04 (-0.11, 0.03)
	P	6.29 (4.26, 8.32)	7.75 (5.72, 9.78)	1.46 (-1.41, 4.33)
	N:P	33:1	23:1	
L39		n=3	n=3	------
	C	0.76 (0.28, 1.23)	1.66 (1.18, 2.13)	0.90 (0.23, 1.57)*
	N	0.04 (-0.02, 0.10)	0.12 (0.06, 0.18)	0.08 (-0.002, 0.17)
	P	1.75 (0.64, 2.86)	3.58 (2.47, 4.69)	1.83 (0.26, 3.40)*
	N:P	23:1	34:1	

Lake	Index	Mean Value Sector 31	Mean Value Sector 35	Mean difference between sectors
L26		n=3	n=3	------
	C	1.34 (1.12, 1.56)	1.40 (1.19, 1.62)	0.07 (-0.24, 0.38)
	N	0.11 (0.09, 0.14)	0.11 (0.09, 0.13)	-0.001 (-0.04, 0.03)
	P	3.00 (1.87, 4.13)	3.00 (1.87, 4.13)	0.00 (-1.60, 1.60)
	N:P	37:1	37:1	

Note: Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.

3.4 TAXONOMIC COMPOSITION OF EPILITHIC ALGAE

3.4.1 Overview

All three lakes had a similar epilithic algal composition by class (Figure 5).

In all three lakes, the three dominant algal classes were Chlorophyceae,

Cyanophyceae and Bacillariophyceae in descending order. Mean percentages

of the total composition for L42, L39 and L26 were 52, 33 and 14; 40, 32 and

26; and 44, 28 and 27 respectively (see Figure 5 for mean cover values).

Chrysophyceae, Xanthophyceae and Dinophyceae account for the remaining

percentage composition of 1.5, 1.6 and 1.1 respectively in all three lakes (see

Figure 5 for mean cover values).
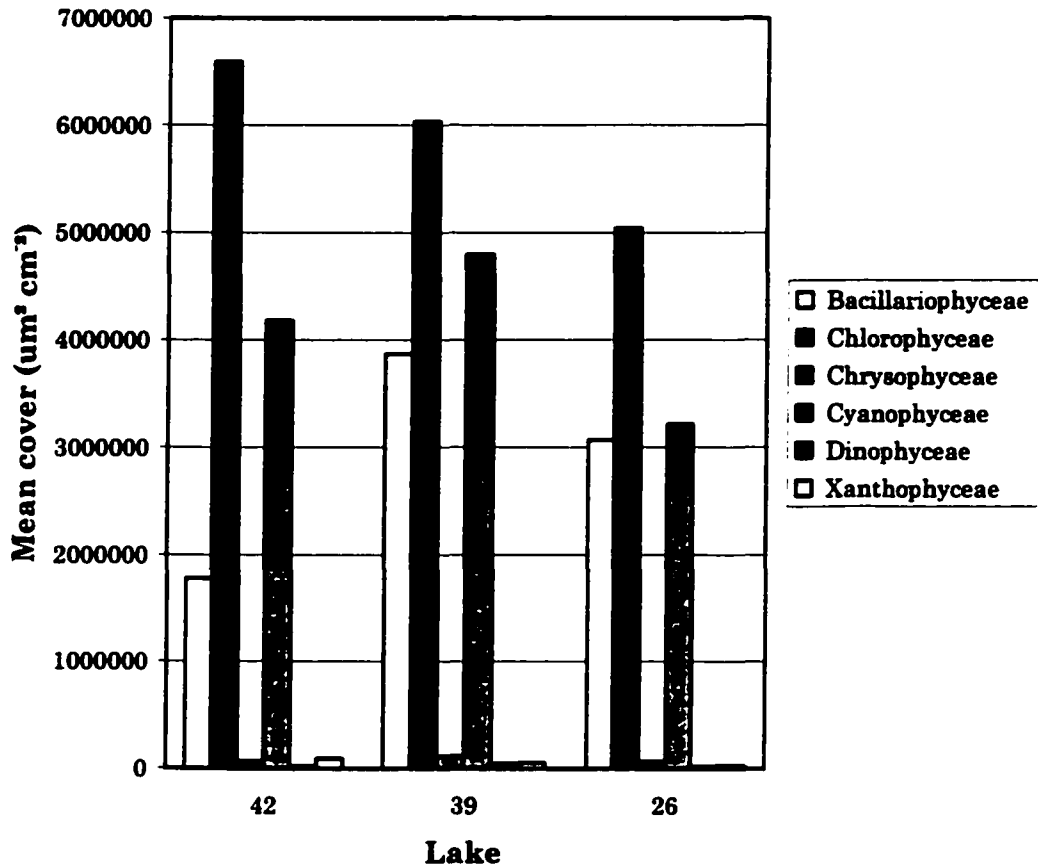
Figure 5. Seasonal mean cover values (µm² cm⁻²) of epilithic algal taxonomic classes for L42, L39, and L26, 1997 and 1998, all depths and sectors pooled.

3.4.2 Treatment Effects

L42 and L39 showed different taxonomic responses to shoreline logging

(Tables 6, 7 and 8), and L26 showed no taxonomic difference at sites with

contrasting shoreline structure (no shoreline vegetation compared to

shoreline forest, Table 9). In L42, there were no significant differences in

densities of epilithic algae between sites with a logged shoreline and sites

with an unlogged shoreline (Table 6).

Table 6. Mean differences in epilithic algae according to taxonomic class between sites with intact shoreline forest and sites where shoreline logging has occurred in L42. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. All numbers in table should be multiplied by 10^6. Asterisks identify differences where the 95% confidence interval does not span zero.

Lake	Class	Mean unlogged (μm^2 cm^{-2}) n=3	Mean logged (μm^2 cm^{-2}) n=9	Mean difference at logged sites (μm^2 cm^{-2})
L42	Chlorophyceae	10.0 (7.0, 13.1)	13.0 (11.4, 14.9)	3.1 (-0.4, 6.6)
	Cyanophyceae	6.9 (4.2, 9.5)	13.0 (11.4, 14.9)	3.1 (-0.4, 6.6)
	Bacillariophyceae	3.3 (2.2, 4.4)	3.4 (2.7, 4.0)	0.07 (-1.2, 1.3)
	Xanthophyceae	0.1 (-0.05, 0.3)	0.2 (0.1, 0.3)	0.08 (-0.1, 0.3)
	Chrysophyceae	0.1 (-0.07, 0.3)	0.1 (0.04, 0.2)	0.03 (-0.2, 0.2)
	Dinophyceae	0.04 (-0.001, 0.09)	0.04 (0.02, 0.07)	-0.0001 (-0.05, 0.05)

Note: Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.

In L42 at sectors 18 and 22, there were no significant differences in densities of epilithic algae before or after shoreline logging (Table 7).

Table 7. Mean differences in epilithic algae according to taxonomic class before and after shoreline logging at sectors 18 and 22 in L42. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. All numbers in table should be multiplied by 10^6. Asterisks identify significant differences where the 95% confidence interval does not span zero.

Lake	Class	Mean pre-logging (μm^2 cm^{-2}) 1997 n=3	Mean post-logging (μm^2 cm^{-2}) 1998 n=3	Mean difference after logging (μm^2 cm^{-2})
L42, s18&22	Chlorophyceae	10.0 (5.6, 14.5)	11.0 (6.9, 15.7)	1.3 (-5.0, 7.5)
	Cyanophyceae	6.9 (2.4, 11.3)	8.3 (3.8, 12.7)	1.4 (-4.9, 7.7)
	Bacillariophyceae	3.3 (1.8, 4.8)	2.8 (1.3, 4.2)	-0.5 (-2.6, 1.6)
	Xanthophyceae	0.1 (0.007, 0.2)	0.09 (-0.02, 0.2)	-0.02 (-0.2, 0.1)
	Chrysophyceae	0.1 (-0.07, 0.3)	0.08 (-0.09, 0.3)	-0.02 (-0.3, 0.2)
	Dinophyceae	0.04 (-0.02, 0.1)	0.03 (-0.03, 0.09)	-0.02 (-0.1, 0.07)

Note: *Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.*

In L39, Bacillariophyceae, Chlorophyceae, Dinophyceae and Xanthophyceae were all significantly higher at the logged shoreline than at the unlogged shoreline (Table 8). Cyanophyceae were significantly lower at the site with a logged shoreline than at the site with an unlogged shoreline. There was no significant difference in Chrysophyceae between sites with a logged and an unlogged shoreline.

Table 8. Mean differences in epilithic algae according to taxonomic class between the site with intact shoreline forest and the site with logged shoreline forest in L39. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. All numbers in table should be multiplied by 10^6. Asterisks identify significant differences where the 95% confidence interval does not span zero.

Lake	Class	Mean unlogged (μm^2 cm^{-2}) n=3	Mean logged (μm^2 cm^{-2}) n=3	Mean difference at logged sites (μm^2 cm^{-2})
L39	Chlorophyceae	4.9 (3.7, 6.1)	7.2 (5.9, 8.4)	2.3 (0.5, 4.0)*
	Cyanophyceae	7.3 (5.7, 8.9)	2.3 (0.7, 3.8)	-0.5 (-7.2, -2.8)*
	Bacillariophyceae	3.0 (1.9, 4.1)	4.8 (3.7, 5.9)	1.8 (0.2, 3.3)*
	Chrysophyceae	0.08 (-0.04, 0.2)	0.1 (0.02, 0.3)	0.06 (-0.1, 0.2)
	Xanthophyceae	0.03 (-0.05, 0.1)	0.08 (0.001, 0.2)	0.05 (-0.06, 0.2)*
	Dinophyceae	0.01 (-0.03, 0.06)	0.09 (0.04, 0.1)	0.07 (0.009, 0.1)*

Note: Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.

In L26, there were no significant differences in densities of epilithic algae

between sectors 31 and 35 (Table 9).

Table 9. Mean areal values and differences in epilithic algae classes between sectors 31 and 35 in L26. Seasonal means and differences were calculated by repeated-measures ANOVA. 95% confidence limits are shown in brackets. All numbers in table should be multiplied by 10^6. Asterisks identify significant differences where the 95% confidence interval does not span zero.

Lake	Class	Mean Sector 31 (μm^2 cm^{-2}) n=3	Mean Sector 35 (μm^2 cm^{-2}) n=3	Mean difference between sectors (μm^2 cm^{-2})
26	Chlorophyceae	4.9 (-0.8, 10.5)	3.1 (-2.6, 8.7)	-1.8 (-9.8, 6.1)
	Cyanophyceae	5.0 (-0.3, 10.2)	2.6 (-2.7, 7.8)	-2.4 (-9.8, 5.0)
	Bacillariophyceae	1.9 (0.3, 3.5)	1.8 (0.1, 3.4)	-0.1 (-2.4, 2.1)
	Chrysophyceae	0 (-0.2, 0.2)	0.2 (-0.07, 0.4)	0.2 (-0.2, 0.5)
	Xanthophyceae	0.06 (-0.01, 0.1)	1.7E-18 (-0.07, 0.07)	-0.06 (-0.2, 0.04)
	Dinophyceae	0.01 (-0.002, 0.03)	3.9E-19 (-0.02, 0.02)	-0.01 (-0.04, 0.008)

Note: *Bonferroni confidence interval adjustment used for post-hoc multiple comparisons.*

4 DISCUSSION

4.1 PHYSICAL AND CHEMICAL IMPACTS OF SHORELINE LOGGING

4.1.1 Light Exposure and Water Temperature

A comparative study of lakes close to the CLEW study lakes found that
shoreline logging was associated with increased littoral light exposure only
on a southern shoreline (Steedman *et al.* 1998). The theoretical light model
developed to support an experimental study of littoral water temperature
change following shoreline logging on L42 suggests up to 15% increases in
light up to 10 m from shore depending on aspect and surrounding topography
(Steedman *et al.* 2001). The southwestern shoreline of L42, where sectors 18
and 22 are located, was logged two years after the main harvesting effort, so
there was likely an increase in littoral light exposure of between 5 and 15%
(depending on distance from shore) after shoreline logging. The
southwestern shore of L39, where sector 23 is located, was logged with the
original harvest. In addition, the relief is fairly low, so there was probably a
similar increase in littoral light exposure on that section of shoreline
following logging.

While some studies have found that green filamentous algae are favoured by
higher light environments (e.g. Shortreed and Stockner 1983), it has also
been found that attached algal biovolume does not respond to shade

treatments (Marks and Lowe 1993). In an experimental study, high-nutrient, high-light substrates failed to produce increased growth in green filamentous algae (Marks and Lowe 1993). Also, shading did not result in decreased growth of attached algae (Marks and Lowe 1993). It is possible that shade treatments were still within the range in which algal photosynthesis was light saturated, which, in benthic algal communities in streams, occurs at approximately 20% of full sunlight (Gregory et al. 1987). Similarly in the case of L42 and L39, it is possible that even with an intact shoreline canopy, epilithic algal photosynthesis in L42 was already light saturated (Turner et al. 1983), thus algal density did not increase following shoreline logging. In addition, removal of shoreline forest canopy may have led to photoinhibition of some algal types, which could explain the lower chl a (only when considering all study sites in L42) concentrations at unlogged sites in L42. Many studies, experimental and comparative, have found that there are interacting effects between nutrients and light that may affect algal biomass (e.g. Hansson 1992; Marks and Lowe 1993; Hill et al. 1995). Another experimental study found that while light is likely to be the primary abiotic constraint on photosynthesis in most shaded streams with nutrients being important only secondarily, the attached algal response may be in physiological or morphological changes rather than in biomass accrual (Hill et al. 1995). If the attached algal response is evident in biomass accrual, the response may differ depending on the age of the attached algal community.

Shoreline logging, which led to loss of shade in littoral areas, also led to transient temperature increases and an increase in diurnal temperature variation in littoral waters in L42 (Steedman *et al.* 2001), and probably also in L39. Clearcut shorelines were associated with increases of 1-2°C in daily maximum water temperature, and increases of 0.3-0.6°C in average diurnal temperature range, compared with undisturbed shorelines or shorelines with 30 m forested buffer strips (Steedman *et al.* 1998; Steedman *et al.* 2001). Littoral temperature increases were significant only during daytime hours (usually in late afternoon) and did not produce detectable changes in long-term average temperature (Steedman *et al.* 2001).

The largest temperature impacts following shoreline logging would be expected on southern shores that had little shading associated with topographic relief (Steedman *et al.* 1998; Steedman *et al.* 2001). Nonetheless, temperature effects observed in lakes for both studies (Steedman *et al.* 1998; Steedman *et al.* 2001), were relatively small compared with those reported for streams with logged shorelines, where increases of up to 10°C in maximum temperature of cold-water streams have been reported (Belt *et al.* 1992). The thermal effects of shoreline logging can be expected to last at least 20-30 years, the minimum time required to re-establish a mature forest canopy in northwestern Ontario (Alban 1982 and Morris *et al.* 1997, cited in Steedman

et al. 1998). Nonetheless, it remains to be demonstrated how such slight heating of littoral waters (i.e. <2°C) and such small changes to diurnal temperature variation will affect biotic and abiotic variables in lakes.

Little is known about the ecological effects of small (i.e. <2°C) temperature changes in thermally dynamic habitats such as lake littoral zones, but changes as small as 2-4°C have been shown to affect lake thermal structure and physical properties (Schindler *et al.* 1990), detrital decomposition, primary production, invertebrate growth and fish behaviour and production (Steedman *et al.* 2001). Water temperature may only be a limiting factor to algal growth and accrual in the case of very low water temperatures (Moore 1980; Hill *et al.* 1995) and may only be stimulatory with greater temperature increases than those seen in L42 and what probably occurred in L39.

Littoral zone biota inhabiting the study lakes must be able to tolerate water temperatures that vary annually from about 1 to 28°C, with daily variations of 0-3°C, light conditions from full shade to full sunlight, and frequent onshore winds. Further, the environmental variability resulting from partial catchment and shoreline deforestation by wildfire probably occurred historically at intervals of not more than 100 years in this region of the boreal forest (Li 2000). It is, therefore, not likely that the changes observed thus far in maximum and average temperature associated with temporary shoreline

deforestation will have significant life history implications for littoral epilithic algae in these study lakes.

4.1.2 Wind Exposure and Water Turbulence

4.1.2.1 Wind Exposure

Midlake wind velocity increased only slightly after extensive shoreline logging on L39 and L42 (Steedman and Kushneriuk 2000). Average wind velocity varied by less than 15% before and after logging, with highest average velocities in 1997, rather than after maximum logging in 1998 (Steedman and Kushneriuk 2000). Although midlake wind speeds on L39 and L42 did not increase much after watershed logging and not at all after shoreline logging, it might be expected that average wind velocities close to lee shores increased more after shoreline logging. Shoreline forest should have an effect on wind velocity analogous to that of agricultural shelterbelts. Wind velocity may be reduced in the lee of forested shelterbelts by 40-80% within five times tree height and by 20-60% between five and 10 times tree height, depending on forest type (Geiger 1965, cited in Steedman and Kushneriuk 2000). However, on all but the smallest lakes, shoreline wind exposure will be influenced more by upland watershed features, which determine upwind fetch, and predominant regional wind direction characteristics, than by local shoreline features. Only small lakes in relatively flat terrain, with fetch less than a few hundred metres, should

experience significantly increased wind exposure after shoreline deforestation (Rask *et al.* 1993, Scully *et al.* 2000).

Sector 22 on L42, where littoral wind velocity was measured, is located on the southwestern shore, which was generally a lee shore. The increase in average littoral wind velocities following shoreline logging, particularly from the southwest and west, may have been associated with shoreline logging on the southwest shoreline. However, the change in predominant wind direction from west during the summer of 1997 to south and southwest in 1998 after shoreline logging was probably a phenomenon related to regional wind characteristics rather than removal of the trees.

4.1.2.2 Water Turbulence

Water turbulence is an important feature of the littoral zone, which may increase following watershed logging, particularly if combined with shoreline logging, due to increased exposure of the littoral zone to wind. Excessive turbulence of shallow zones in lakes is generally detrimental to attached algae due to erosion. However, increased water movement that is non-erosional leads to thinner boundary layers overlying epilithic algal films, potentially increasing rates of nutrient uptake (Turner *et al.* 1991). When water movement is sufficient to erode the algal film, as can occur in streams and the upper littoral zones of lakes, both the degree of diffusive resistance to

nutrients and density-dependent effects will be reduced (Turner 1993).

Another study found an inverse relationship between attached algal biomass

and current velocity even after nutrient enrichment (Ghosh and Gaur 1994).

So, increases in turbulence can be either beneficial or detrimental to growth

and accumulation of algal biomass depending on the amount of the increases

and possibly the timing of turbulence events.

There are two possible reasons for the different turbulence results between

L42 and L39 at the top of the water column, about 2 m offshore. The first is

that the north end of both lakes appeared to be more exposed to wind in

general (personal observation). However, by 1997 the shoreline and

watershed had been logged at the north end of L42 whereas the shoreline and

watershed remained unlogged at the north end of L39. Secondly, the south

end of L39 is generally more exposed to wind than the south end of L42

because of watershed topography, which is flatter in the L39 watershed

thereby increasing fetch. These differences in site character and treatment

probably left the south and north ends of L39 more similarly exposed, and

probably accentuated the greater exposure of the north end of L42 in relation

to the south end. In sum, the comparative study between treatment sites

suggests, but does not demonstrate, an increase in littoral water turbulence

following shoreline logging, at least at the water surface offshore. Since

water turbulence was not monitored in 1998, I did not determine whether

there was a change in water turbulence following shoreline logging at sectors 18 and 22 in L42.

The results potentially more relevant to epilithic algae were those from the shoreline 0.5 m depth and the offshore 1.5 m depth. While the shoreline findings do not demonstrate any differences in water turbulence between logged and unlogged sites, despite the increased littoral wind velocities, the offshore 1.5 m depth findings suggest opposite effects between logged and unlogged treatment sites in L42 and L39. The offshore 1.5 m depth findings are probably more indicative of inter-site differences within L42 and L39 than of treatment effects. Turbulence could be expected to be higher and more variable at the water surface than at the lake bottom since water at the surface would be more responsive to gusts, whereas deeper water would be more integrative of longer-term wind conditions. Thus, the generally more exposed northwestern shore of L42 (sectors 28 and 30) may have contributed to the higher water turbulence at the 1.5 m depth there than at the southwestern shore (sectors 18 and 22). With the steeper littoral slope at sector 38 on L39, water turbulence may be attenuated compared to the shallower slope at sector 23 on L39.

No direct relationship was found between littoral wind velocity and gypsum cylinder diameter loss in L42 at Sector 22. Wind velocity was integrated over

both 24-hour and 48-hour periods and wind directions were selected with the longest fetch. Wind data were compared both to all of the gypsum loss data pooled or analyzed separately by cylinder position (i.e. shoreline, offshore surface or offshore bottom). A conclusion from this lack of relationship might be that average littoral water turbulence, as indexed by gypsum cylinder erosion, is not strongly influenced by littoral wind velocity. In addition, evaluating a relationship between littoral wind velocity and littoral water turbulence may be below the resolution of the methods used and the data collected.

Since there does not appear to be a relationship between wind velocity and water turbulence in the littoral zone on L42 and since there were no real differences in water turbulence between logged and unlogged sites in 1997, except at the top of the water column 2 m offshore, there was probably not an increase in water turbulence that would be relevant to growth and accumulation of epilithic algae due to shoreline logging on L42, sectors 18 and 22 in 1998 despite the higher littoral wind velocities in that year compared to 1997. However, since turbulence was not measured in 1998, there is no way of verifying that supposition.

4.1.3 Nutrients

These lakes did not show much change in water quality in the first three

years after experimental logging. Impacts on biomass and nutrients may take more than one or two years to appear because of slow lake volume replacement by post-logging runoff (Steedman 2000). Preliminary analyses did show post-logging increases in runoff and export of nutrients, cations and anions immediately after the 1998 clearcuts on subcatchments of L26 and L39 (C. Allan, personal communication in Steedman 2000). Post-logging catchment export associated with the larger clearcuts of 1996 probably increased in a similar way.

The nutrient changes found in L39 indicated slight phosphorus (P) enrichment, which, even though P is a limiting nutrient in these lakes, was probably insufficient to induce changes in epilithic algae. Under oligotrophic to mesotrophic conditions, P controls phytoplankton biomass and productivity (various references in Welch et al. 1989). Phosphorus limits biomass of attached algae at N:P ratios above 17 by mass (Chessman et al. 1992). At ratios below 7 by mass, nitrogen was limiting (Chessman et al. 1992). Nitrogen:phosphorus ratios in all three study lakes were always well above 17:1 (Table 5). However, N becomes a colimiter with phosphorus as the N:P ratio in nutrient supply decreases below about 25:1 by weight (Smith 1982 cited in Welch et al. 1989). Ratios were a bit lower than 25:1 after shoreline logging on L42 (sectors 18 and 22) and at the unlogged site on L39, thus under some conditions both nutrients were limiting to the growth of algal

biomass. Otherwise, ratios exceeded 25:1, in which cases P was the limiting nutrient.

The C enrichment seen at the logged treatment site in L39 was slight and may only have been a temporary response from fine logging debris and particulate export immediately following logging. After this immediate post-logging pulse, there may be a lower C level due to decreased allochthonous input (France *et al.* 1996), nonetheless long-term control is exerted by N and P (Welch *et al.* 1989).

These results of none to slight nutrient enrichment in L39 and L42 following shoreline and catchment logging are consistent with those of Schindler *et al.* (1980). That study found that catchment deforestation by wildfire was not likely to cause important water quality changes in boreal Shield lakes unless water renewal time was short relative to the time required for catchment revegetation (Schindler *et al.* 1980). While it may take at least 20 years to reestablish forest canopy, revegetation begins as soon as the first summer following logging. Water renewal times in L26, L39 and L42 are 13.1, 8.2 and 10.5 years respectively (Steedman 2000). On the other hand, a comparative study of boreal Shield lakes impacted by harvesting or wildfire found that impacts were directly proportional to the area of watershed harvested or burnt divided by the lake's volume or area (Carignan *et al.* 2000). Of

particular concern for lake water quality were increases in dissolved organic

carbon, light attenuation coefficient and total phosphorus, which appeared to

evolve on a longer time scale and which could affect other key limnological

properties (Carignan et al. 2000). In the case of these study lakes with

relatively small catchment to lake surface area ratios, and with relatively

large volumes and long water renewal times, impacts of catchment

disturbance are likely to be very small. Alternatively, impacts may evolve on

longer time-scales than a few years.

4.2 EPILITHIC ALGAE

All lakes had similar algal assemblages, although L42 had proportionately

fewer Bacillariophyceae and Cyanophyceae than L39 and L26. In general, all

three of the study lakes were extremely dilute and unproductive. However,

L42 was slightly higher in particulate nutrients C and N, and much higher in

P than the other two lakes, which may explain the higher density of

Chlorophyceae in L42 (Cattaneo 1987).

Pre-logging mean silica (Si) contents in L26, L39 and L42 were 1.17, 0.52 and

0.03 mg L respectively (Steedman 2000). Post-logging means were higher for

L42 but not for either L26 or L39. A concentration of 0.4 mg L of Si has been

suggested to limit planktonic diatom growth in offshore waters of Lake

Michigan (Parker et al. 1977). These figures suggest that L26 and L39

contained sufficient Si to support diatom growth, but that L42 was severely

limited, which may explain the low populations of Bacillariophyceae and

Chrysophyceae in that lake. Despite the low levels of particulate nutrients in

L26, the high levels of Si would help support the density of Bacillariophyceae

and Chrysophyceae. Bacillariophyceae were more numerous in L39 than in

either of the other two lakes, probably due to the sufficient levels of Si and

the higher levels of particulate nutrients than in L26.

Cyanophyceae were abundant in all three lakes. Attached Cyanophyceae are

often important in oligotrophic lakes, which is sometimes linked to their

ability to fix N (Reuter *et al.* 1983 cited in Cattaneo 1987). Classes

Dinophyceae and Xanthophyceae were found in low densities in all three of

the study lakes.

4.2.1 Epilithic Algal Community Impacts

This study provided equivocal evidence for alteration of attached algal

diversity after shoreline logging. There were no differences in algal

taxonomic class densities found between logged and unlogged sites in L42,

whether considering the whole lake or just the experimental sectors 18 and

22 before and after logging. In L39, however, areal densities of

Bacillariophyceae, Chlorophyceae, Dinophyceae and Xanthophyceae were

higher at the logged site than at the unlogged site and the density of

Cyanophyceae was lower at the logged site. There were no differences between sites in L26. Total chl a and hplc chl a concentrations echoed the algal class analyses except in L42 where total chl a was lower at sites with a logged shoreline. This inconsistency in L42 may be explained by the fact that total chl a includes live algal pigment as well as degradation or non-living products, whereas only live cells were used in the taxonomic analysis.

The different algal densities found at the logged study site in L39 compared to the unlogged study site were reflected in higher biomass of organic material and higher chl a (both total and hplc) concentrations at the logged study site. There were also higher concentrations of both carbon (C) and phosphorus (P) at the logged study site in L39. These findings are in contrast to the findings from L42, including at the experimental sites on L42 (sectors 18 and 22), where pre-logging and post-logging data were collected, which showed no differences between logged and unlogged sites in algal class densities, nutrients or biomass of material (organic or inorganic). Because of the lack of differences in L42 between logged and unlogged sites, the differences found in L39 may be at least partly attributed to inter-site differences rather than to treatment effects. These results are in contrast to the findings of a comparative study on L42, L39 and L26, which found a post-logging increase in inorganic littoral sedimentation in L42 in 1997, probably from aeolian deposition from upland roads, but not in L39 or L26 (Steedman

and France 2000).

These findings are in contrast to those of others studying impacts of logging on benthic algae. One post-disturbance comparative study showed higher attached algal biomass (indexed by chl a concentration) in littoral areas of lakes with logged watersheds (Planas et al. 2000). However, these increases in algal biomass were associated with shorter water renewal times and larger post-logging nutrient increases than in the Coldwater study lakes. In an experimental study of forest clearcutting and soil scarification conducted in three Finnish lakes, it was found that chl a increased in the experimental lakes only after both logging and soil scarification (Rask et al. 1998). Changes in the water chemistry showed that the forestry treatments, especially the soil scarification, increased both the organic and the inorganic load from the catchment to the lake (Rask et al 1998). The increase in attached algal chl a and the bloom of planktonic cyanophytes in one of the lakes were indicative of a slight eutrophication (Rask et al 1998). In streams, it has been found that filamentous green algae are favoured over diatoms after logging (Hansmann and Phinney 1973; Shortreed and Stockner 1983; Noel et al. 1986). Anecdotal observations of increased growth of attached diatoms on passive fishing gear in the Finnish lakes subject to catchment forestry support the view that in lakes diatoms are commonly favoured by catchment forestry (S.-L. Markkanen personal communication in Rask et al.

1998). In the Finnish lake studies, the limnological responses to catchment forestry were generally found to be modest, probably partly due to the ca. 50 m wide protective zones left along the shorelines of the lakes (Rask *et al.* 1998).

The uncertainty in these findings of little evidence for post shoreline-logging impacts on the epilithic algal community of the Coldwater study lakes, revolves at least partly around the lack of large changes in the physical and chemical environment after shoreline logging. There was only a small increase in light exposure and only very small and transient increases in littoral water temperature and in diurnal temperature variation found in L42. Fetch and wind velocities from the west and the southwest increased somewhat, but there appeared to be no associated increase in water turbulence. There were also no apparent changes associated with shoreline logging in environmental factors like particulate nutrients or in biomass of material, organic or inorganic, or in concentration of chl *a* (except in total chl *a* at unlogged sites in L42).

What complicates the evidence for little apparent change in the physical environment after shoreline logging, however, are findings from L39. Data from L39 showed differences in almost all measured variables between the logged and unlogged study sites. However, the L39 study was a comparative

one while the L42 study included an experimental component. Thus, inter-site variability in the case of the L39 findings must be considered. And, in fact, the two sites on L39 were very different. Watershed, shoreline and littoral slopes were much shallower at sector 23 than at sector 38, in L39. Steeper slopes tend to promote greater sediment transportation whereas shallower slopes favour greater sediment accumulation (Blais and Kalff 1995). This difference may lead to higher biomasses of material and nutrients, as well as greater accrual of algal biomass at the more shallowly-sloped study site than at the more steeply-sloped study site. Further, the site at sector 23, L39, was more exposed to light, being on the southwestern shoreline and having experienced logging of shoreline forest, than the site at sector 38, which was on the northwestern shoreline and retained shoreline forest. More light tends, in general, to promote algal growth, particularly of green filamentous types (although there are other factors involved as discussed above). Finally, the north end of L39 was generally more exposed to wind, and water turbulence tended to be higher there at the water surface offshore than at sector 23. With the steeper slope, however, despite water turbulence at 1.5 m depth being similar between the two sites, the turbulence may have been an erosive force at sector 38 due to the steeper slope, further preventing accumulation of material at sector 38. The greater exposure of sector 38 to wind than sector 23 may have been partly evened out by the shoreline forest logging at sector 23, which, combined with watershed logging

there and the relatively flat terrain, effectively increased fetch and therefore possibly exposure of sector 23 to wind.

Inter-site differences in L42 were smaller, but also, because two sites were sampled within each treatment regime rather than just one, differences could be compensated for somewhat. For example, while littoral slope at sector 28 was steeper than at the other sites, sector 28 was paired with sector 30, which had a very shallow slope. Littoral slope at sectors 18 and 22 was very similar. Further, shoreline forest was logged at sectors 28 and 30 on the northwestern shoreline of L42, effectively increasing light exposure. Water turbulence was only slightly higher at the northern sectors, however, compensatory factors such as slightly higher light levels after logging, may have contributed to evening out conditions between the sets of treatment sites. Most importantly, there were no differences in biomass of epilithic material (organic or inorganic), nutrients, chl a, or in taxonomic classes before and after shoreline logging at sectors 18 and 22 on L42. From these results, logging of shoreline forest did not appear to produce measurable impacts on epilithic algae or their environment in L42.

4.2.2 Study Evaluation

Epilithic algae are, for many reasons, a community used for studying impacts of experimental manipulations of water or catchment. They have a short

generation period, they are numerous and diverse, and they are easy to find and relatively easy to sample. However, they also appear to have fairly broad ranges of tolerance, they appear to be very adaptable, and taxonomic classification is difficult. Because attached algae are attached to a substratum, they are not limited to obtaining nutrients from the surrounding water. Rather, they appear to be able to switch to obtaining nutrients from the material around them and sometimes from the substratum itself. Further, the physical morphology of the community is such that algal organisms are able to switch from being light-requiring to being more nutrient-requiring. As the season progresses, the mat becomes denser and thicker and light becomes less available due to light attenuation through the water column and through the mat itself. In addition, cellular and colonial growth forms can change morphologically to adapt to the maturing attached algal mat, thereby allowing organisms to continue to compete effectively for nutrients and light.

Algal taxonomic classification in general is difficult, but taxonomic classification analysis for attached algae is particularly challenging because of the difficulties involved in getting a pure sample of algae. The epilithic environment includes sediment, pollen, seeds, plant debris, fungi and whatever else has blown in or settled out there in addition to algae. There appears to be no satisfactory way to clean samples without destroying algal

cells and breaking up colonies so the analyst generally has to dilute the sample and use some way of breaking up the macroscopic clumps, suspend it, extract a very small amount and place it in some kind of counting chamber. One is still faced, however, with picking out the algae from impenetrable microscopic clumps of material and among a lot of other debris. There are methods to further break up these microscopic clumps, but none of the necessary equipment was available. In addition, there is some difference of opinion among taxonomic experts on identifying characteristics, taxonomic assignation of algal cells and on names within the finer taxonomic designations. Further, discrepancies between biovolume and cell density reflect another problem in analyzing attached algal communities in that large changes in small taxa are often not revealed in a biovolume analysis (Marks and Lowe 1993) or in an areal density analysis as was done in this study. Responses by epilithic algal communities resulting from environmental change may not be reflected in physical aspects such as biovolume, density, or taxonomy, but in physiological characteristics such as carbon uptake or photosynthesis-irradiance relationships (Hill *et al.* 1995).

Given the combination of a highly adaptable and mutable community and analytical difficulties, impacts on the epilithic algal community resulting from experimental manipulations may be very difficult to detect. Since we are dealing here with a catchment manipulation, aquatic effects may be

damped out and diluted so that only the most extreme impacts would be detected. This was anticipated when the catchment experiment was planned, so the manipulation was extreme with very high percentages of watersheds and shorelines logged. Nonetheless, impacts detected on the physical environment have been negligible to nonexistent for a number of reasons, among which are the long water renewal times of these small lakes and the low catchment area to lake volume ratios (Steedman 2000).

While chemical and physical impacts are often well understood and characterized, impacts on biota are relatively poorly understood partly because of ecological reasons discussed above, but largely because of the analytical challenges. However, ultimately what we really care about are the biota and the reason that we measure chemical and physical conditions is to decide whether there may be impacts on biota. So eventually we have to try to measure the biotic response. After all, biota may be more sensitive detectors than are our analytical instruments so if we are unable to detect physical changes, but we can detect biotic changes, then we can extrapolate to physical changes and decide that either our sampling methodology or our detection was inadequate. On the other hand, if we also detect no biotic changes, evidence as to impacts is less conclusive since then either our biotic sampling or our analysis could still be inadequate in some way.

With this study, the time frame (two sampling seasons) may have been too short and the sampling effort not intensive enough to understand the natural variability of the epilithic algal community in the Coldwater Lakes. There may have been insufficient pre-disturbance sampling time to be able to identify post-disturbance changes and possibly not enough of the lake bottom was sampled to overcome the huge natural variability inherent in the community. Further, in this study I was evaluating effects of shoreline logging after watershed logging had already been done. It is very possible that any impacts from shoreline logging were overwhelmed by impacts from watershed logging. Sampling before any logging had been performed in the watersheds of the study lakes may have been necessary to be able to assess impacts of shoreline logging following watershed logging.

4.3 CONCLUSIONS

Overall there is little evidence from this study demonstrating impacts from shoreline logging on epilithic algae or their environment. The clearest changes after shoreline logging were higher littoral wind velocities and slightly increased light exposure. The post-logging increase in littoral wind velocities did not appear to result in higher littoral water turbulence. Increased light exposure apparently did not lead to higher algal chl a or algal density, except perhaps in L39. The ambivalence of these findings, with regards to the inconsistent results between L42 and L39, does suggest the

potential for impacts given certain conditions. Inter-site differences between treatment sites in L39 may have been accentuated by shoreline logging. That does not appear to have been the case in L42 since perhaps the sites were more similar to begin with and there were no measured changes in the before and after study at sectors 18 and 22. Despite the different shoreline conditions in L26, no inter-site differences were demonstrated.

Shoreline logging in L42 and in L39 generally left a fringe of shrubby vegetation along the shoreline, which may have helped maintain the stability of shoreline sediments (personal observation). Also, while there was some post-logging slash lying in littoral areas, occurrences were minimal and isolated. It appears from this study that shoreline logging, performed carefully to minimize direct littoral disturbance, did not impact the epilithic algal community at the level investigated here. These results should not be generalized, however, without further, more intensive experimental study than could be performed here. A study with greater replication and improved site matching would have greater detection power than this preliminary study. In addition, comparative pre- and post-disturbance investigation of lakes in northwestern Ontario would be beneficial.

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APPENDIX

APPENDIX A

Study site profiles for L42, L39 and L26.

Study site profiles for L42

Depth (m) vs Distance from shore (m)

Legend: Sector 18, 22, 28, 30

Study site profiles for L39

Depth (m) vs Distance from shore (m)

Legend: Sector 23, 38

Study site profiles for L26

Depth (m) vs Distance from shore (m)

Legend: Sector 31, 35