ENVIRONMENTAL IMPACTS OF WINTER ROAD MANAGEMENT AT THE CASCADE LAKES AND CHAPEL POND

THE FINAL REPORT

Route 73 at Upper Cascade Lake, looking west. Photo by Kerop Janoyan.

ENVIRONMENTAL IMPACTS OF WINTER ROAD MANAGEMENT AT THE CASCADE LAKES AND CHAPEL POND

TOM A. LANGEN, MICHAEL TWISS, THOMAS YOUNG, KEROP JANOYAN,
J. CURTIS STAGER, JOSEPH OSSO JR., HANNA PRUTZMAN, BEN GREEN

FINAL VERSION
SUBMITTED JUNE 6, 2006

CLARKSON CENTER FOR THE ENVIRONMENT REPORT #1

FUNDING PROVIDED BY THE
NEW YORK STATE DEPARTMENT OF TRANSPORTATION

TABLE of CONTENTS

PROJECT PERSONNEL

Principal Investigators

Tom Langen, Dept. of Biology, Clarkson University

Michael Twiss, Dept. of Biology, Clarkson University

Thomas Young, Dept. of Civil & Environmental Engineering, Clarkson University

Kerop Janoyan, Dept. of Civil & Environmental Engineering, Clarkson University

J. Curtis Stager, Natural Resources Division, Paul Smith's College

Graduate Student Investigators

Joseph Osso, Jr., M.S. candidate, Environmental Science & Engineering

Hanna Prutzman, M.S. candidate, Environmental Science & Engineering

Ben Green, M.S. candidate, Civil & Environmental Engineering

Graduate Student Research Assistants

Timothy Sharac, M.S. candidate, Environmental Science & Engineering

Tiffany Basara, M.S. candidate, Environmental Science & Engineering

Andrew Casper, Ph.D. candidate, University of Laval

Undergraduate Research Assistants

Amanda McKenna, Environmental Engineering Major, Michigan Technological University (Clarkson REU Site Program in Environmental Science & Engineering, Summer 2003)

Sonia Mae Johns, Biology Major, Clarkson University (Fall 2003)

Matthew Williams, Biology Major, Clarkson University (Honors Program, Summer 2004)

Nicole Reed, Environmental Engineering Major, Pennsylvania State University. (Clarkson REU Site Program in Environmental Science & Engineering, Summer 2004)

Lindsay Hoffman, Civil & Environmental Engineering Program, Clarkson University (Honors Program, Summer 2004)

Anirban Ghosh, Environmental Science & Policy Major, Clarkson University (Honors Program, Summer & Fall 2004)

Neal Liddle, Biology Major, Clarkson University (Summer 2004)

Brian Roden, Environmental Science & Policy Major, Clarkson University (Summer 2004)

Adam Rose (Honors Program, Spring 2005)

TECHNICAL WORKING GROUP MEMBERS

TWG Chair / Project Supervisor

Kyle Williams, Associate Environmental Specialist, NYSDOT Environmental Analysis Bureau

Other TWG Members

New York State Department of Transportation

Jeanne Hewitt, Associate Environmental Specialist, NYSDOT Environmental Analysis Bureau
Mauricio Roma, Senior Environmental Specialist, NYSDOT Environmental Analysis Bureau
Robert Selover, Associate Director, NYSDOT Transportation Maintenance Division
Michael Lashmet, NYSDOT Transportation Maintenance Division
Chuck Briggs, NYSDOT Transportation Maintenance Division
Peter Howard, Maintenance Environmental Coordinator, NYSDOT Region 1
Ray Oram, Maintenance Environmental Coordinator, NYSDOT Region 2
Dennis Pawlicki, Maintenance Environmental Coordinator, NYSDOT Region 7
Richard Ambuske, Regional Landscape/ Environmental Manager, NYSDOT Region 1 Design
John Falge, Regional Environmental Coordinator, NYSDOT Region 7
Mark Bonfey, Assistant Resident Engineer, NYSDOT Essex County Residency

New York State Department of Environmental Conservation

Richard Preall, Senior Aquatic Biologist, NYSDEC Region 5 Fisheries

Adirondack Park Agency

Daniel Spada, Supervisor, Natural Resources Analysis, NYS Adirondack Park Agency
Edward Snizek. Associate Project Analyst, Fresh Water Resources, NYS Adirondack Park Agency

PROJECT OVERVIEW

This document is a final report of the project *Winter Impacts along Cascade Lakes in the Adirondacks (C-03-06),* funded by the New York State Department of Transportation (NYSDOT). The Technical Working Group over-seeing this project included representatives from several NYSDOT units, the New York State Department of Environmental Conservation, and the Adirondack Park Agency. This research project was conducted at two sites along New York State Route 73 in Essex County, New York State: Upper and Lower Cascade Lakes, and Chapel Pond. It was initiated July 2003, and final fieldwork was completed in June 2005.

The general goal of this project was to assess the magnitude of environmental problems associated with winter road management (i.e. chemical deicer and abrasive applications) at these sites, and to make appropriate recommendations on how to ameliorate environmental impacts of winter road management (**Note**: Throughout the report, we will use the standard term *chemical diecer*, recognizing that materials may be used in pre-storm *anti-icing* (as is the normal practice at the Cascade Lakes), or for true *deicing* after precipitation has accumulated onto the road surface).

The '*Cascade Lakes*' project had the following five specific objectives:

(1) Review the literature on sodium chloride and other chemical deicers, particularly in regards to environmental impact. Also review the literature on other winter event management technologies that may reduce the need for chemical deicers. The reports on these components of the project are found in Chapter 1 and Chapter 2.

(2) Using the results of our original research, review of the published literature, and other sources of data, evaluate the evidence that winter road maintenance has resulted in the dieback of paper birch trees (*Betula papyrifera*) and other vegetation at Upper and Lower Cascade Lakes. Our original research included characterizing the physical and chemical structure of soil and birch tree stand structure both adjacent and distant from NYS Route 73 at the Cascade Lakes and Chapel Pond. Information relevant to this objective is found in Chapter 1, Chapter 3, and Chapter 4.

(3) Using the results of our original research and review of the published literature and other sources of data, evaluate the evidence that winter road maintenance has degraded water quality and stressed natural aquatic communities within the Cascade Lakes and Chapel Pond. Our original research included two years of monthly water quality sampling at the Cascade Lakes and Chapel Pond, and a two hundred year reconstruction of water quality at each lake based on sediment cores. The reports on these components of the project are found in Chapter 1, Chapter 3, and Chapter 5.

(4) Using the results of original research by our research team and data from past research at the lakes (presented in Chapter 3 and Chapter 5), produce a mass-balance computer model of the fate and transport of chloride through the Cascade Lakes watershed, and use it to predict future trends in water quality of the Cascade Lakes due to use of deicing salt on NYS Route 73. The model was also to be used to predict the likely consequences of changing deicer application practices. The report on this component of the project is found in Chapter 6.

(5) Synthesize the findings under Objectives 1-4, and use them to make policy recommendations on future winter maintenance practices on NYS Route 73, and elsewhere within the Adirondack Park region. The report on this component of the project is found in Chapter 7.

SUMMARY FINDINGS

We have concluded the following, based on our research and review of the literature. These summarized findings are expanded upon in Chapter 7.

(1) The average annual quantity of sodium chloride deicer deposited within the watershed of Upper Cascade Lake (UC) during the last 5 years = 152 t (t = metric tons), at Lower Cascade Lake (LC, exclusive of what may flow from Upper Cascade Lake) = 205 t, and at Chapel Pond (CP) = 71 t. Sand was applied at the following annual amounts: UC = 0.8 t, LC = 1.1 t, CP = 0.07 t. Relatively insignificant quantities of calcium chloride were also applied during the period. Annual application rates are presently very different from the early 1980s, when sand was more heavily used: UC salt = 75 t, sand = 400 t; LC salt = 125 t, sand = 550 t.

(2) At NYS Route 73 near the Cascade Lakes, application rates of road salt per run are lower than application rates on highways in most states outside of New York. The annual salt loadings are extraordinarily high due to the high cumulative number of runs per winter, necessitated by the large number of winter storm events over the long winter season, poor drainage and heavy runoff from surrounding steep mountain slopes, and strong sustained winds that blow snow onto the road and deicing salt and sand off of the road surface.

(3) Roadside soil has become physically and chemically altered due to the direct and indirect consequences of applying sodium chloride and sand during winter road maintenance. Soil organic matter is low, soil particle size is dominated by sands, sodium is high and nutrient cation concentrations are low. As a consequence, the soil is droughty, easily erodible, and nutrient-poor.

(4) Paper birch (*Betula papyrifera)* and other vegetation have significantly declined in abundance along State Highway 73 in the Cascade Lakes region since the early 1980’s, when there was already evidence of a decline in stand health. The decrease in adult birch tree abundance and health is in part attributable to succession in a

naturally senescent birch stand, and is occurring throughout the region. Roadside birch mortality appears to have been accelerated, however, due to increasingly infertile roadside soil, salt exposure at the roots, aerial deposition of road salt, and erosion and related damage caused by natural and road-associated disturbance of the soil.

(5) Most important as a cause of the long-term decline in vegetation cover near the road is an absence of recruitment of young trees and other plants to replace those that die. Poor recruitment is likely to be a consequence of the extensive changes to the physical and chemical properties of roadside soil caused by heavy applications of chemical deicers and sand, and damage to above-ground plant parts caused by exposure to road salt from road splash and other forms of aerial deposition.

(6) Chloride levels within soils adjacent to State Highway 73 are generally low, indicating that chloride is rapidly transported away via surface and ground water flow. Upper and Lower Cascade Lakes now have chloride levels 100 to 150 times higher than expected for a comparable Adirondack lake. Within the last five years, there has been a 250% increase in chloride concentrations within the Cascade Lakes, which has been caused by the recent dramatic increase in road salt applications. The concentration of chloride in Chapel Pond is slightly elevated, about twice as high as the average for Adirondack Lakes. Chloride loadings to the lakes peak in summer, caused by percolation through the ground and eventual groundwater discharge into the lakes.

(7) A strong concentration gradient of chloride occurs in Upper and Lower Cascade Lakes, with as much as a 57% difference in concentration between surface water (epilimnion) and bottom water (hypolimnion). Although the chloride concentrations and magnitude of the concentration gradients are within the range that results in a permanent thermal stratification on some lakes (meromixis), Upper and Lower Cascade Lakes remain dimictic (i.e. complete turnover occurs twice a year, caused by thermal mixing). Lower Cascade Lake turns-over earlier than Upper Cascade Lake,

indicating that there is little resistance to thermal mixing at present in this more heavily chloride-contaminated lake.

(8) In the late summer, Lower Cascade Lake has dissolved oxygen concentrations at depths below the thermocline (hypolimnion) that are low enough to cause severe physiological stress to lake trout (*Salvelinus namaycush*), and probably therefore to round whitefish (*Prosopium cylindraceum*), which have similar physiological requirements to the trout. During this same period, temperatures above the thermocline (epilimnion) are high enough to also cause stress to these fish. Upper Cascade Lake and Chapel Pond remain cool and oxygenated enough in the hypolimnion to support these fish throughout the year. Other researchers have shown that whitefish in Lower Cascade Lake are stunted, and thin for their body length, relative to Upper Cascade Lake and other populations in the Adirondacks. At Lower Cascade Lake, low oxygen in the hypolimnion in late summer is due to the lake's morphology, and the morphology of the surrounding watershed, which cause large quantities of organic matter (e.g. leaves and woody debris from the surrounding steep mountain slopes, and dead roadside vegetation) to be transported into its small deep hole, and therefore resulting in high biochemical oxygen demand at depth. Fortunately, Lower Cascade Lake remains dimictic, but there is a possibility that the strong chloride concentration gradient causes resistance to thermal mixing, and therefore prolongs stressful conditions by delaying fall turnover. Lower Cascade Lake turns-over earlier than Upper Cascade Lake, indicating that there is little resistance to thermal mixing at present.

(9) Salt loadings into the Cascade Lakes and Chapel Pond occur throughout the year, thus aquatic organisms' exposure to salt contamination is chronic and relatively mild rather than severe and short duration. The published research on the toxicology of exposure to chronic exposure to elevated salt concentrations indicate that concentrations of the magnitude currently found in Upper and Lower Cascade Lakes are not likely to negatively impact the health of aquatic vertebrates, benthic macroinvertebrates, zooplankton, or aquatic vascular plants. The levels are high

enough to potentially alter phytoplankton and periphyton communities by eliminating the most sensitive species. We completed experiments on the impact of chronically elevated concentrations of chloride on phytoplankton and periphyton from the Cascade Lakes, using concentrations higher than currently recorded but plausible in the near future. These results indicate that changes in the composition of Cascade Lake algal communities are likely to occur as a consequence of continued increases in chloride contamination, but lake productivity may not change.

(10) Benthic macroinvertebrates differ among the three lakes. Chapel Pond has the lowest abundance, but highest taxonomic diversity (as indicated by Simpson's diversity index), and appears to typify an oligotrophic (low-nutrient) lake with fine water quality. Upper Cascade Lake has the highest abundance of macroinvertebrates, and a similar taxonomic composition to Chapel Pond. Lower Cascade Lake is the most aberrant, with a taxonomic composition that indicates high organic inputs and somewhat degraded water quality relative to the other lakes. This is not likely to be due directly to chloride contamination, but rather to the factors described under Finding 8.

(11) At Upper and Lower Cascade Lakes, major changes in lake water quality are associated with sedimentation caused by construction of NYS Route 73 in the first half of the 20th century, and heavy use of sand abrasives in winter, which has resulted in an elevated rate of sediment deposition and sediment composition that is lower in organic content and higher in sand and silt. The abundances of chloride-tolerant epilimnetic diatom species have increased in the last half-century, associated with increased chloride loading from deicing road salt. At the Cascade Lakes, the diatom species composition in sediment cores indicate chloride concentrations similar to contemporaneous water sample measurements, but the Chapel Pond sediment-core diatom samples indicate a higher level of chloride contamination than contemporaneous water samples.

(12) We have remapped in three dimensions Upper Cascade Lake and Lower Cascade Lake, aided by the application ArcGIS. Upper Cascade Lake is 14% larger in volume than calculated by the Adirondack Lake Survey Corporation, Lower Cascade Lake is 73% larger. Upper Cascade Lake contains 80,000 – 130,000 kg chloride, and Lower Cascade Lake contains 50,000 – 75,000 kg chloride. Seasonal changes in chloride concentrations in the lakes appear to be gradual, peaking in summer, suggesting that there is no 'shock' elevation of concentrations associated with seasonal events (e.g. snow melt), and that sizeable input into the lakes are via groundwater discharge. Based on the mass balance model of chloride transport through the Cascade Lakes, simulated over a period of 20 years, chloride concentrations are predicted to rise over the next five years in the Cascade Lakes, with the biggest increases in the Lower Cascade hypolimnion (a 40% increase). Under present salt loadings, peak chloride concentrations in the Lower Cascade Lake hypolimnion are predicted to approach the USEPA recommended maximum limits for chronic exposure to aquatic life. Lower Cascade Lake also remains at risk of becoming meromictic. Doubling the annual salt loading will double the lakes' concentrations of chloride, halving the salt loading will halve the concentration of chloride in each lake (as was empirically observed in the early 1990s). Changes in salt loadings result in a new equilibrium concentration of chloride within about seven years.

(13) Some alternative chemical deicers (liquid magnesium chloride, magnesium chloride with organic anti-corrosive) have been applied on NYS Route 73 at the Cascade Lakes, and found to be unsatisfactory in terms of effectiveness. Acetate based chemical deicers may lessen some negative environmental impacts associated with road salt, but the added biochemical oxygen demand to the lakes may put Lower Cascade at further risk. Acetate would be expected to have already degraded before the late summer period of maximum hypolimnetic oxygen stress, however, so it may be worthwhile to conduct a carefully monitored experimental application of an acetate-based chemical deicer such as calcium magnesium acetate.

(14) Winter road management technologies may reduce the need to apply the current quantities of chemical deicer. In the short run, reducing motorists speed, better weather monitoring, and technologies to reduce melt-water runoff over the road surface may be possible to implement. In the long term, road reconstruction using surface materials that reduce ice bonding, and barriers that prevent winter precipitation from being deposited or blowing onto the road surface may be worth considering.

CHAPTER 1

THE EFFECTS OF CHEMICAL ROAD DEICERS ON SURROUNDING FORESTS AND LAKES – A REVIEW

Salt-contaminated snow pile at a pulloff along Route 73 at Lower Cascade Lake. Photo by Michael Twiss.

Chapter 1. **THE EFFECTS OF CHEMICAL ROAD DEICERS ON SURROUNDING BOREAL FORESTS AND LAKES.**

Report prepared by Tom A. Langen, with contributions by Hanna Prutzman.

GENERAL INTRODUCTION

In the US, Canada, and Europe, the quantity of sodium chloride and other chemical deicers applied to roads has increased dramatically over the last 50 years, including substantial increases in use over the last 15 years (Environment Canada and Health Canada 1999, Thunqvist 2004, Jackson and Jobbagy 2005). Presently, about 18 million tonnes of salt are applied to roads in the US, which is approximately 8-fold greater than the background rate of natural deposition from the atmosphere (Jackson and Jobbagy 2005). Averaged over New York State, road applications of NaCl result in an average annual salt loading of 200 kg/ha, as compared to 0.9 kg/ha from natural sources of atmospheric deposition (Jackson and Jobbagy 2005).

Of course, chemical deicers are not evenly dispersed in the environment; contamination is greatest near roads. Annual loadings of chemical deicer to the surrounding environment can be quite high; annual applications of sodium chloride, in tonnes per km two-lane highway: Sweden = 12 – 14 (Thunqvist 2004), Maine = 12 (Mason et al 1999), New York = 18.6 (Albright 2005), Michigan = 19 (Gales and VanderMeulen 1992), Massachusetts = 22.6 (Mattson and Godfrey 1994), Ontario, Quebec = 20 – 35 (Environment Canada and Health Canada 1999), southeastern Ontario = 27 – 75 (Environment Canada and Health Canada 1999). Single pass loadings in the Northeastern US and Eastern Canada range from 130 – 170 kg per two-lane highway km (Environment Canada and Health Canada 1999).

Negative impacts of chemical deicers on the environment have been noted since the 1940s (reviewed in Haynes et al. 1970, 1976). Long term data on water quality indicate that there has been a dramatic increase in sodium and chloride concentrations in northeastern US rivers and streams over the last 50 years, and this increase is attributable to the use of sodium chloride as a chemical deicer (Smith et al. 1987, Jones et al. 1992, Godwin et al. 2003, Kaushal et al. 2005). These increases occur in urban, suburban, and rural landscapes (Kaushal et al. 2005). Similar increases in chloride concentrations have been documented in the Laurentian Great Lakes (Jones et al. 1992, Moll et al. 1992). Reconstructed historical changes in water quality, as indicated by diatom remains and other tracers in sediment cores, show that the water quality and biological communities in many lakes located near roads in the northeastern US have been altered by sodium chloride and other chemicals associated with winter chemical deicing within the last five decades (Siver et al. 1996, Dixit et al. 1999, Ramstack et al. 2004). Extrapolating from trends reported in Kaushal et al. (2005), Jackson and Jobbagy (2005) predict that many rural streams within the next century will reach chloride concentrations that exceed the USEPA guidelines for potable water (250 ppm), which coincides with a concentration for which chronic exposure is toxic for the most sensitive aquatic organisms (United States Environmental Protection Agency 1988, Evans and Frick 2001).

In this chapter, we will review the environmental impact of chemical deicers, with an emphasis on sodium chloride, the most widely used form. The review includes a general overview of sources, transport, and fate of deicing road salt (Section 1-1); the impacts of deicing salt on vegetation, including changes to soil properties that affect plant health (Section 1-2), the impacts of deicing salt on water quality and aquatic organisms (Section 1-3), and the known environmental impacts of chemical deicer alternatives and additives to sodium chloride (Section 1-4). Excellent comprehensive general reviews of the environmental impact of sodium chloride and other chemical deicers include Haynes et al. (1970, 1976), Transportation Research Board (1991), D'Itri (1992), Blomqvist (1998), Environment Canada and Health Canada (1999), Buckler and Granato (1999), Fischel (2001), Hauser and Wunderlich (2004) and Ramakrishna and Viraraghavan (2005).

Section 1-1. **SOURCES, FATE & TRANSPORT**

Natural sources of sodium and chloride in the environment include geological sources (e.g. mineral sodium chloride deposits), and aerial wet deposition originating from ocean evaporation. Anthropogenic sources besides chemical deicer applications include wastewater treatment effluent, effluent from chemical and petrochemical industrial processes, smokestack emissions from fossil fuel combustion at power plants, landfill leachate, chemical fertilizers, animal manure, and salt storage sites (Wagner and Steele 1989, Mayer et al. 1999, Thunqvist 2004, Jackson and Jobbagy 2005). Other sources of sodium and chloride loading to the environment are generally small or localized in comparison to chemical deicers (Mayer et al. 1999, Thunqvist 2004, Jackson and Jobbagy 2005). In boreal shield environments, the background concentrations of sodium and chloride in soil and water are very low (Mayer et al. 1999, Environment Canada and Health Canada 1999).

When chemical deicer is applied to a road, the substance is transported to the surrounding environment via several mechanisms. (1) When a solid is applied (e.g. rock salt), some bounces off the road and lands on the surrounding soil and vegetation, referred to as ‘scatter’. (2) Liquid deicer and solid deicer dissolved in melt water flows off the road to the adjacent soil or water. (3) Melt water and slush contaminated with deicer are splashed off the road by vehicle tires. (4) Snow and slush contaminated with deicer is plowed off of the road and discarded along the road or at a ‘snow dump’. (5) Aerosolized road splash is transported as a fine spray by wind currents. (6) Fine particles of solid chemical deicer are transported by wind currents from the road surface to the surrounding environment (Blomqvist 1998, Environment Canada and Health Canada 1999). The relative importance of each of these mechanisms varies depending on (1) the type of chemical deicer applied and the method of application; (2) the design of the road, e.g. its surface characteristics; (3) patterns of vehicle usage, e.g. volume and timing of vehicle traffic, speed; (4) characteristics of surrounding soil and vegetation; (5) local hydrology, including natural hydrology and road drainage structures; (6) local topography; and (7) microclimate (Blomqvist 1998).

Snow Banks

Chemical deicer can accumulate to high concentrations in contaminated snow, particularly where snow cover persists for a long duration of time. Snow and snowmelt usually is contaminated with heavy metals, organic compounds and particulates (Evans and Frick 2001). Contaminants become concentrated at the base of the snowpack by a process called *preferential elution* (Oberts 1991). Contaminated snow is typically localized to within 5 m of the road (Buttle and Labadia 1999).

During snowmelt a large pulse of chemical deicer can enter soil and waters. Highest loadings occur during rain on snow events at the end of winter. Contaminated melt water may infiltrate soil and groundwater, or may flow over the soil surface to surface-waters; the dynamics of snowmelt discharge, contaminant composition, and fate of runoff depends on a variety of factors (Oberts 1991). Roadsalt can facilitate the mobility of metals and other contaminants (reviewed in Buckler and Granato 1999).

Along one Ontario highway, the snow pack had concentrations of up to 6500 ppm Na and 9900 ppm Cl. Upon snowmelt, 3 – 5 kg of sodium chloride per m highway was released in a brief pulse (Labadia and Buttle 1996, Buttle and Labadia 1999). Concentrations in roadside snow are quite variable (<100 – 10000 ppm, typical value = 4000 ppm), and as high as 18000 ppm (Environment Canada and Health Canada 1999, Evans and Frick 2001).

Aerial Transport and Deposition

Aerial transport occurs from vehicle tire passage stirring up road brine into a salty spray, and by dry salt residue being stirred up by air currents from passing vehicles or wind. Once aerosolized, salt may be transported by wind until deposited onto the surface of plants or soil. The distance of transport has been directly measured by collecting deposition in containers and sampling surfaces (e.g. absorbent filter paper disks) or indirectly by measuring concentrations in snow, soil, or plant parts (Foster and Maun 1978, Blomqvist 1998, Cain et al. 2001). Visible salt damage to plants has also been used to infer transport distance (Kelsey and Hootman 1992, Cain et al. 2001)

In one study, 20 – 63% of deicing salt applied to a road was transported by air to the surrounding environment, most depositing 2 – 40 m from the road. In that study, 90% was deposited within 20 m of the road (Blomqvist and Johansson 1999). In general, the effect zone (maximum distance at which plant damage occurs from aerial deposition) is 40 m – 100 m from the road, but damaging levels of salt in plant tissues receiving aerially deposited sodium chloride can occur as far as 200 m distance (Cain et al. 2001). Other studies have documented transport several hundred meters from a major highway (Kelsey and Hootman 1992, Cain et al. 2001, Blomqvist 2003), but in all studies the greatest quantities of contaminant are deposited a short distance (within 30 m) of the road (Blomqvist 1998). Vertically, spray sufficient to damage sensitive tree species may reach heights of 15 m or more (Kelsey and Hootman 1992). The distance transported depends on the topography of the site, prevailing wind direction, the amount of roadside vegetation, and traffic speed and volume (Cain et al. 2001). Vehicle velocity significantly affects the distance to which chemical deicer is transported, and reductions in vehicle speeds can measurably reduce the area of surrounding environment contaminated by chemical deicers (McBean and Al-Nassri 1987). Salt-resistant plants may serve as an effective barrier to inhibit aerial transport (Kelsey and Hootman 1992, Forman et al. 2003).

Soil and Vegetation

The terrestrial environment can be altered dramatically by chemical deicers. Sodium chloride, the most widely used chemical deicer, alters soil chemistry, soil physical structure, and soil biota. Plants uptake sodium, chloride, and other constituents of chemical deicers, and thus serve as a 'sink' for these substances, but plants may also reduce growth or die as a consequence of exposure to toxic levels of chemical deicer, resulting in further soil degradation. Thorough reviews of the impact of deicing road salt on soils, and the consequences of the alterations to plants, are found in Morin et al. (2000), Transportation Research Board (1991), Butler and Addison (2001) and Cain et al. (2001). The results of these four reviews are summarized in Environment Canada and Health Canada (1999).

Sodium and chloride move differentially through soils, since sodium is attracted to and absorbed by the predominantly negatively charged soil surfaces, whereas the chloride is not absorbed (Scott 1980). Soil sodium concentrations tend to increase across years (Hutchinson 1970, Hofstra and Smith 1984). Since the chloride anion is not retained in the soil, it may infiltrate into the groundwater or run off over the surface of frozen soils (Labadia and Buttle 1996, Norrström and Bergstedt 2001). Levels of sodium and chloride are highest in early spring, due to the melt period, and decrease over time as the ions gradually infiltrate deeper into the soil profile (Hanes et al. 1976, Scott 1980, Hofstra and Smith 1984). Eventually, these substances enter the groundwater and are carried with groundwater flow. The chloride usually flushes out of soils quickly, and is readily transported by groundwater or surface flow, while the sodium competes with the other base cations (magnesium, calcium, potassium, and protons) for exchange sites on soil particles (Mason et al. 1999). The surface soil horizon is assumed to have the greatest potential to retain deicing salts, since it usually has greater organic matter content, a finer texture, and larger cation and anion exchange capacity than sand and gravel found at depth (Labadia and Buttle 1996).

Accumulation of sodium causes the leaching of base cations. In soil systems, calcium and magnesium are more strongly attracted to the cation exchange sites of clay particles and organic matter in the soil than is sodium since they have a higher valence than sodium (i.e. they are divalent cations, whereas sodium is monovalent). However, if the sodium concentration in the soil is high enough, the sodium can out-compete calcium, magnesium, and (monovalent) potassium for cation exchange sites (Howard and Beck 1993, Norrström and Bergstedt 2001, Löfgren 2001; Bryson and Barker 2002). If calcium, magnesium and potassium are not bound to soil particles they can be flushed out by soil water, e.g. transport via groundwater flow (Mason et al. 1999). In principal, the same process can occur with some other chemical deicers (e.g. magnesium chloride, calcium magnesium acetate), in which case the magnesium or calcium cations from the chemical deicer result in the loss of other soil cations (Amrhein et al. 1992).

There is also a strong relationship between trace metal mobilization (e.g. cadmium, chromium, copper, iron, lead, nickel, zinc) and sodium chloride (Ahmrein et al. 1992, Granato et al. 1993; Norrström and Jacks 1998, Buckler and Granato 1999, Bäckström et al. 2004). By reducing cross-linking/bridging complexation between adjacent organic compounds or organic matterand mineral surfaces, sodium tends to destroy soil structure and increase organic matter mobility, thereby increasing the chance of metal mobilization since trace metals are primarily complexed to organic compounds in soils (Amrhein et al. 1992). Metal mobilization occurs through different mechanisms such as ion exchange, chloride complex formation and possibly colloid dispersion (Bäckström et al. 2004). Ion exchange can increase the mobility of H+ and trace metals such as Zn and Cd (Löfgren 2001). The heavy metals that are mobilized can be highly toxic to plants, soil biota, and aquatic organisms. Chloride forms complexes with toxic heavy metals (e.g. cadmium, zinc, copper, aluminum, mercury, and lead), making them more water soluble, and as a consequence more bioavailable to plants (Bäckström et al. 2004). Roadside environments typically have elevated concentrations of heavy metals originating from fuel and lubricants, worn vehicle parts, and road construction materials (Forman and Alexander 1998, Forman et al. 2003, Trombulak and Frissell 2000). Chloride-containing chemical deicers (sodium, magnesium, or calcium chlorides) may act synergistically with these metals to expose plants to toxic levels of metals via root uptake.

Excessive exchangeable Na, a consequence of heavy applications of road salt, causes breakdown of soil aggregates, soil swelling, decreased pore size and reduced permeability of the soil to air and water (Amrhein et al. 1992, Jones et al. 1998, Morin 2000). Silt and clay particles and soil organic matter are lost. This reduces the capacity of the soil to retain water and nutrients. Soil infiltration rates are reduced and surface runoff is increased because of the greater impermeability of the soil due to salt crusting and breakdown of soil particles. Because of the breakdown in soil structure, the soil becomes more prone to erosion. High sediment load from salt-contaminated soils also increases transport of trace metals (Amrhein et al. 1992). Thus, the soil is more unstable, infertile and drought-prone. This, in turn, reduces vegetation cover (Blomqvist 1998, Cain et al. 2001) with the resulting increase in transport of soil, toxic metals, and other contaminants to water bodies.

Sand is used in conjunction with chemical deicers, and acts synergistically with them to reduce soil fertility to plants. Unlike sodium chloride, which either infiltrates into the soil or is transported away with water flow, sand accumulates near the site of deposition. Accumulations of sand can alter the bulk density and porosity of the roadside environments. Sand deposition reduces the proportion of soil organic matter in a soil, which in turn lowers cation exchange capacity and therefore the ability of soil to retain nutrients. Sand does not retain water as well as silts, clay and organic matter; therefore, water can flow through the sand carrying essential cations with it.

There is a positive feedback between degradation in soil fertility and vegetation loss due to chemical deicers. The direct impacts of deicing road salt on plant health and the indirect effects due to changes in soil fertility can result in a loss of vegetative ground cover. Vegetation loss results in less soil organic matter, which in turn reduces nutrient retention in the soil. Vegetation loss also results in higher erosion rates, causing loss of soil fertility due to changes in physical and chemical structure, and increased sedimentation of adjacent surface waters (Environment Canada and Health Canada 1999, Cain et al. 2001).

Ground Water

Water contaminated with chemical deicer enters groundwater after infiltration through soil, or less commonly through infiltration from a contaminated surface water body (e.g. contaminated pond or stream). Important sources include road melt-water runoff, melting of a deicer-contaminated snow pack, or rainwater percolating through contaminated soil (Labadia and Buttle 1996). The rate of infiltration depends on soil structure, weather, and bedrock geology. Both soil surface freezing and decreases in soil permeability caused by sodium contamination decrease infiltration and increase surface water flow (Rosenberry et al. 1999). Frequent rainfall increases the rate of movement through soil to groundwater (Howard and Beck 1993). Most groundwater recharge in the northeastern US and southeastern Canada occurs in late winter and early spring from snow melt and spring rains, and therefore is likely to carry a high burden of dissolved chemical deicer (Environment Canada and Health Canada 1999). Bedrock geology affects the direction of flow, and can lead to considerable storage of contaminant in rock fissures (Pollock 1992). Depending on the study, anywhere from 10% - 55% of deicing salts applied to roads ends up in the groundwater. Major factors affecting how readily deicing salts enter groundwater include soil permeability, vegetation cover, topography, and roadside drainage techniques (Jones et al. 1992, Blomqvist 1998).

Not surprisingly, chemical deicer concentrations in groundwater are highest near the source road. Concentrations of sodium and chloride in groundwater may continue to rise for years, and the plume of contaminated groundwater may move slowly from the source. Due to the much greater hydraulic residence time of groundwater versus surface waters, there is likely to be a considerable lag (up to decades) between changes in chemical deicer application practices and re-equilibration of groundwater concentrations of chemical deicer (Pollock 1990, Labadia and Buttle 1996, Environment Canada and Health Canada 1999).

Both field measurements and mass-balance modeling indicate groundwater chloride concentrations of 300 – 450 ppm in southern Ontario, where annual applications are 18 – 36 tonnes sodium chloride per km two-lane highway (Environment Canada and Health Canada 1999). Ground water chloride concentrations are much less variable throughout the year than surface waters (Williams 1999). Elsewhere in Ontario, springs in rural regions located near roads have chloride concentrations of 100 ppm (background less than 3 ppm), with springs located in urban areas attaining concentrations over 1000 ppm (Williams 1999). In one extensive survey in Ontario, where background concentrations of chloride in groundwater is less than 20 ppm, chloride levels ranged up to 700 ppm in wells, 2,800 ppm in springs, and 13,700 in soil pore water; elevated chloride levels were attributable to deicing road salt (Howard and Beck 1993). Wells located within 100 m of highways are frequently contaminated at levels that exceed recommended concentrations for potable water (Hutchinson 1970, Pollock 1992, Howard and Beck 1993, Bäckström et al. 2004, Ramakrishna and Viraraghavan 2005). Chloride concentrations have been rising in municipal production wells in southern Ontario since the 1960s, and have increased five-fold over a 25 year period. Because of the effects of deicing road salt on soil physical structure and chemistry, groundwater is predicted to have elevated concentrations of some cations (e.g. magnesium, calcium, potassium) and metals, but there is little evidence of such groundwater chemistry changes in the few studies that have examined this issue (Bowser 1992). Groundwater concentrations of sodium and chloride are high enough to impact sensitive aquatic organisms that inhabit groundwater and springs (Williams 1999, Evans and Frick 2001).

Surface Waters

Surface waters receive loadings of chemical deicer from direct input (e.g. by dumping contaminated snow into the water or onto surface ice), aerial deposition, surface runoff, input from contaminated surface water sources (e.g. contaminated inlet streams), and via groundwater discharge. Surface runoff, input from contaminated surface water sources, and groundwater discharge are the principal ways that most surface waters receive chemical deicer contamination (Ramakrishna and Viraraghavan 2005). The surface water bodies most sensitive to road salt impacts include wetlands; small ponds or lakes with long water residence times, located in urban areas or having a relatively large proportion of the perimeter adjacent to road; and streams draining large urban areas or rural highways (Environment Canada and Health Canada 1999). Chloride mass-balance models have been successful at predicting and explaining temporal and spatial patterns of salt contamination of surface waters by chemical deicer and other sources (Bowser 1992, Rosenberry et al. 1999, Rimmer et al. 2005). Landscape models using GIS have been successful at predicting surface water contamination levels using predictors such as proximity to roads, chemical deicer application rates, land use, and characteristics of water bodies (Mattson and Godfrey 1994, Rhodes et al. 2001, Thunqvist 2004).

There are important differences between loadings caused by surface sources (surface runoff, direct dumping, and aerial transport) and groundwater discharge. Surface runoff will be comprised primarily of dilute but otherwise unmodified chemical deicer. There may be minor concentrations of other contaminants found in road runoff (e.g. Legret and Pagotto 1999, Oberts 1991), and other contaminants originating from the chemical deicers themselves (Environment Canada and Health Canada 1999, Ramakrishna and Viraraghavan 2005). When sodium and chloride are applied as deicers, the molar ratio of the two ions will be 1:1 if surface sources are the principal loading sources (Thunqvist 2003, Jackson and Jobbagy 2005). Loadings will be high during the winter period of application, and spike at snow melt (Oberts 1991). Concentrations will drop over the summer and fall as chemical deicer is diluted and transported away, and loadings are stopped. When surface loadings are the primary source of chemical deicer in surface water, changes in winter management (e.g. changes in application rate or type of chemical deicer) will result in a rapid response in the receiving surface waters (Scott 1981, Oberts 1991). Concentrations in runoff depends on the area of road drained, topography and drainage patterns, amount of chemical deicer applied before a thaw period, rate of rise and duration of temperatures above freezing, and precipitation (Scott 1981, Oberts 1991). Runoff can have very high concentrations of chloride (often 1,000 – 6,000 ppm, maximum = 25100 ppm), and can vary seasonally by 2 orders of magnitude at a location (Evans and Frick 2001).

When the loading source is groundwater discharge, the chemical profile may become modified due to the interaction of chemical deicer with soil. When sodium and chloride are applied as deicers, because sodium is selectively retained in soil, the molar ratio of the two ions in water is altered from 1:1, with chloride having the higher concentration (Bowser 1992, Rhodes et al. 2001, Jackson and Jobbagy 2005). Other cations (magnesium, potassium, calcium) may also become enriched in surface waters receiving deicer contaminated groundwater flow, because of soil losses of these cations due to competition with sodium (Bowser 1992, Howard and Beck 1993, Mason et al. 1999, Judd et al. 2005). Surface waters receiving groundwater flow may also have elevated concentrations of metals, due to the changes in soil chemistry and physical properties caused by chemical deicer that result in increased mobility of metals

(Löfgren 2001). Similarly, pH may decrease because of increased mobility of H^+ ions in soil (Shanley 1994, Löfgren 2001, Thunqvist 2003)

There are significant differences in the phenology and impact to surface waters of deicing chemical contamination between flowing waters (rivers and streams) and still waters (ponds and lakes), e.g Hoffman et al. (1981). In flowing waters, the water column mixes well and residence time is short. As a consequence, concentrations of chemical deicer are uniform throughout the water column at a given sampling site, and concentrations will drop relatively quickly once loadings are eliminated. Thus, flowing waters that receive loadings primarily from surface sources will spike in concentration of chemical deicer around the spring thaw, and quickly decline once applications have stopped (Scott 1981). Because water volume and water velocity peak with snow melt and spring rain, the potential to transport chemical deicer away from a source is high. On the contrary, when a principal loading source is groundwater, loadings will be less variable seasonally, and concentrations of chemical deicer in flowing waters may actually increase in summer and fall in small streams, during low (base) flow conditions when the principal source of water is groundwater rather than precipitation (Mason et al. 1999).

For deicer-impacted rural streams and rivers, chloride concentrations range 10 – 100 ppm above background (background = under 2 ppm chloride in boreal shield waters; Environment Canada and Health Canada 1999). Road-salt impacted Adirondack streams along a rural highway had a mean chloride concentration of 5.2 ppm, and at times were up to 33× higher than the background concentration of 0.6 ppm (Demers and Sage 1990, Demers 1992). Streams in urban areas can have much higher levels of contamination, especially those that receive substantial stormwater runoff (Environment Canada and Health Canada 1999)

With lentic waters, the potential for persistent accumulation of chemical deicer is possible, because the residence time of water is generally much longer than flowing waters. Thus, chronic concentrations may become much higher than streams. Moreover, gradients of concentration by depth can form in still waters, because the density of chloride-contaminated water is higher than uncontaminated water. It is common for strong gradients in chloride concentrations to form in lentic systems (e.g. ponds, lakes) where mixing is relatively low (e.g. Hoffman et al. 1981, Kjensmo 1997, Judd 2005). Severely deicing salt-contaminated rural lakes have chloride concentrations of 150 – 350 ppm, and urban areas are an order of magnitude more contaminated (Environment Canada and Health Canada 1999). Ponds near two-lane highways in southern

Ontario have mean chloride concentrations = 95 ppm (maximum = 368 ppm), 3-5 lane highway = 124 ppm (maximum = 620 ppm), and six lane or greater = 925 ppm (maximum = 3950 ppm; Evans and Frick 2001).

The strongest gradient forms when the water body has a small surface area but is relatively deep (depth/mean width > 0.05, Kalff 2002), and the water body is protected from wind (Hakala 2004). Under such conditions, it is possible for normal fall and spring 'turnover' (vertical mixing) to fail to occur. Under normal (viz. dimictic) conditions in small temperate lakes, cooling air temperatures in fall and warming air temperatures in spring cause surface water to increase in density and sink, resulting in vertical mixing throughout the water column. This carries dissolved oxygen down to the oxygen-depleted depths, and nutrients concentrated at the bottom of the water body are transported up to the surface. When strong density gradients form because of deicing salt contamination, resistance to mixing increases. This can prolong the period of thermal stratification (the period when there is little mixing between surface and deep waters, corresponding to summer and midwinter conditions in the northern hemisphere), and thus prolonging the duration of low-oxygen conditions in the depths, and reducing transport of nutrients to the surface where they can be assimilated by phytoplankton (Judd 1970, Hoffman et al. 1981). Under the most extreme conditions, the density gradient can be strong enough to prevent turnover altogether, and thus preventing vertical mixing, a condition termed *meromixis* (Kalff 2002). With meromixis, deep waters tend to become hypoxic or anoxic, and nutrient availability is reduced near the surface (Judd et al. 2005). Examples of chloride concentration gradients caused by chemical deicer contamination in lakes for which mixing is impaired include: (1) surface (mixolimnion) = 293 ppm; bottom (monimolimnion, 15 m) = 446 ppm (Judd et al. 2005), (2) bottom (17 m) = 104 ppm (Smol 1983), (3) surface = 190 ppm, bottom (24 m) = 400 ppm (Bubeck et al. 1971). Under hypoxic, chloride rich conditions, mercury can be released from sediments (Feick, 1972, Wang et al. 1991). Phosphorus and other metals may also be released (Wetzel 1983). The normal dimictic condition has returned to lakes after reductions in deicing road salt loading (Evans and Frick 2001, Judd et al. 2005).

Section 1-2. **THE EFFECTS OF DEICING ROAD SALT ON TERRESTRIAL PLANTS**

One of the most easily detected effects of heavy applications of deicing road salt is the decline in health of roadside vegetation. Roadside vegetation is an important barrier and sink for salt spray from the road (Viskari and Karenlampi 2000). Because of the impacts on agriculture (especially orchards), commercial forestry, and ornamental vegetation in residential areas, there is a large body of literature on the impact of deicing road salt in terrestrial plants. Major reviews include Hanes et al. (1970, 1976), Dobson (1991), Transportation Research Board (1991), Blomqvist (1998), Environment Canada and Health Canada (1999), and Cain et al. (2001).

Damage caused by deicing road salt can potentially be confused with damage caused by insect or fungal pathogens or drought, but can be distinguished by the spatial pattern of damage. (Cain et al. 2001). The following are indications that vegetation has been damaged by road salt (Lumis et al. 1973, Dirr 1976, 1978, Cain et al. 2001).

- Damage to vegetation is localized to the vicinity of the road or areas that receive road run-off.
- Damage declines with distance from the road.
- Damage is greatest on the downwind side of the road.
- A plant is more severely damaged on the side facing the road.
- Parts of a plant that are below the snowline, and therefore sheltered from road spray, are less damaged.
- Parts of a plant that are above the zone of road spray are less damaged.
- Damage only extends a short distance into dense plant cover (because salt spray is blocked by the first line of vegetation).
- For conifers, damage becomes evident in late winter and continues into the growing season.
- For conifers, damaged needle tips are brown or reddish, and there is a sharp line of demarcation between the damaged tips and the undamaged (green) base of the needles.
- For deciduous trees, damage becomes evident in spring when buds swell and continue into the growing season.

Forms of Damage to Plants

Plants are exposed to deicing road salt (or other chemical deicers) in the soil at the roots (i.e. the rhizosphere). Aboveground plant tissues are also exposed to salt spray originating from the road, and aerial deposition from wind blown aerosol mists containing sodium chloride or other chemical deicers. Cain et al (2001) and Hofstra et al. (1979) review plant damage caused by road salt. Damage to plants from deicing salt exposure includes inhibition of root proliferation or root necrosis (death), reduction in leaf size and leaf growth, leaf wilting, leaf chlorosis (yellowing), leaf burn (often starting at the leaf margins), and tissue necrosis of aboveground and belowground plant tissues. Seed germination can be delayed and seed viability can be reduced. With aerial deposition, stems (particularly young growth), leaf buds, and flower buds are killed. Conifers show reddening or browning of needles and premature leaf drop. 'Witches brooms' of adventitious shoots are produced where terminal buds and shoots have been killed.

Tree species vary in their tolerance of sodium chloride; tolerance depends on the ability to exclude salt and tolerate osmotic stress from salt that has been taken up (Greenway and Munns 1980, Paludan-Müller et al. 2002). Mechanisms of salt tolerance fall into two classes: avoidance and tissue tolerance. Avoidance refers to the ability to keep salt away from the parts of the plant that are more susceptible. Tissue tolerance refers to situations where salt is stored in vacuoles within the plant tissues (Allen et al. 1994). There is potential for the successful development of salt-tolerant lines of some forest tree species; however there has been little progress to date (Allen et al. 1994).

Direct Damage to Plants due to Deicing Salt Exposure

There are multiple mechanisms by which deicing salt can damages plants, reviewed in Hanes et al. (1976), Jennings (1976), Dobson (1991), Morin et al. (2000), and Cain et al. (2001). Deicing salt directly harms plants in the root zone by affecting water balance of cells, causing dehydration and collapse of plant tissues. Similar damage is caused to aboveground tissues that are exposed to aerial deposition of road salt. Water uptake at the roots can be disrupted. Sodium and chloride are readily transported into the plant and distributed throughout the living tissues, and can concentrate to high levels, affecting cell physiology (Greenway and Munns 1980). The toxic effects of sodium chloride on cell physiology are enhanced when a plant is deficient in nutrients such as calcium, as can occur in salt-contaminated soils (Bogemans et al. 1989). At high levels of salt uptake, crystals can form within cells and desiccate a plant, and consequently killing it through physiological drought (Hutchinson 1970). Road salt can therefore limit the water availability in both the soil and the tree, causing a drought in two different ways. Sodium and chloride ions can also directly interfere with uptake of nutrients, affecting both passive and active transport mechanisms. There is evidence that cold-hardiness can be reduced by uptake of salt (Sucoff and Hong 1976).

Road salt impacted soils can lower water potential and transpiration rates, resulting in reduced photosynthetic rates and ultimately reduced tree height and girth (Kayama et al. 2003). Salt toxicity can result in shorter roots with few laterals; saplings with reduced root structures are subject to higher levels of water stress during drought periods (Croser et al. 2001). In conifers, salt toxicity can cause premature needle drop, reducing tree condition and slowing growth (Cain et al. 2001).

Indirect Damage to Plants due to Changes in Soil Fertility

Deicing salt damages plants indirectly by altering soil physical structure and chemistry, and by impacting soil biota. Thorough reviews of the impact of deicing road salt on soils, and the consequences of the alterations to plants, are found in Morin et al. (2000), Transportation Research Board (1991), Butler and Addison (2001) and Cain et al. (2001). The results of these four reviews are summarized in Environment Canada and Health Canada (1999).

Sodium and chloride move differently through the soil since sodium is attracted to and absorbed by the soil colloids, whereas the chloride is not absorbed (Scott 1980). Since the chloride anion is not retained in the soil, it may infiltrate into the groundwater or run off over the surface of the frozen soils (Labadia and Buttle 1996, Norrström and Bergstedt 2001). Levels of sodium and chloride are highest in early spring, due to the melt period, but decrease over time as the ions are gradually leached deeper into the soil profile (Hanes et al. 1976, Scott 1980, Hofstra and Smith 1984). The chloride usually flushes out of the soil system while the sodium competes with other base cations (magnesium, calcium and potassium) for exchange sites on soil particles (Mason et al. 1999). Upper soil is assumed to have the greatest potential to retain deicing salts, since it usually has greater organic matter content, a finer texture, and larger cation and anion exchange capacity than sand and gravel found at depth (Labadia and Buttle 1996).

One indirect way that deicing road salt damages vegetation is nutrient starvation caused by loss of nutritive cations in soils receiving heavy application of sodium chloride. Accumulation of sodium causes the leaching of base cations, which are important nutrients for tree growth. In undisturbed soil systems, calcium and magnesium are more strongly attracted to the cation exchange sites of clay particles in the soil since they have a higher valence than sodium (i.e. they are divalent cations, whereas sodium is monovalent). However, if the sodium concentration in the soil is high enough, the sodium can out-compete calcium, magnesium, and (monovalent) potassium for cation exchange sites (Howard and Beck 1993, Norrström and Bergstedt 2001, Löfgren 2001; Bryson and Barker 2002). If calcium, magnesium and potassium are not bound to soil particles they can be flushed out by soil water, e.g. transport via groundwater flow (Mason et al. 1999).

There is also a strong relationship between trace metal mobilization and sodium chloride (Granato et al. 1993; Norrström and Jacks 1998, Bäckström et al. 2004). Sodium tends to destroy soil structure and increase organic matter mobility, thereby increasing the chance of metal mobilization (Amrhein et al. 1992). Metal mobilization occurs through different mechanisms such as ion exchange, lowered pH, chloride complex formation and possibly colloid dispersion (Bäckström et al. 2004). Ion exchange can increase the mobility of H+ and trace metals such as Zn and Cd (Löfgren 2001). The heavy metals that are mobilized can be highly toxic to plants, soil biota, and aquatic organisms. Chloride forms complexes with toxic heavy metals (e.g. cadmium, zinc, copper, aluminum, mercury, and lead), making them more water soluble, and as a consequence more readily taken-up by plants (Bäckström et al. 2004). Roadside environments typically have elevated concentrations of heavy metals originating from fuel and lubricants, worn vehicle parts, and road construction materials (Forman and Alexander 1998, Forman et al. 2003, Trombulak and Frissell 2000). Chloride - containing chemical deicers (sodium, magnesium, or calcium chlorides) may act synergistically with these metals to expose plants to toxic levels of metals via root uptake.

Excessive exchangeable Na, a consequence of heavy applications of road salt, causes breakdown of soil aggregates, decreased pore size and reduced permeability of the soil to air and water (Jones et al. 1998, Morin 2000). This reduces the capacity of the soil to retain water, nutrients, and organic matter. Soil infiltration rates are reduced and surface runoff is increased because of the greater impermeability of the soil. Because of the breakdown in soil structure, the soil becomes more prone to erosion. Thus, the soil is more unstable, infertile and drought-prone. This, in turn, impedes root growth and inhibits germination of seeds. Plants can become water and nutrient deficient, and structural stability is impaired.

Healthy soil microbial communities are essential for plant health. They play a key role in nutrient cycling, converting decaying organic matter into forms that can be assimilated by plant roots for plant nutrition. Some fungi and bacteria are mutualistic associates of plant roots (e.g. mycorrhizal fungi, which are associated with the roots of most tree and shrub species in boreal forests), facilitating nutrient transport into the roots. Accumulation of sodium chloride in soils may negatively impact microbial communities to the extent of affecting plant health, but there is little data on the impact of chemical deicers on soil microbial communities. The few relevant studies that have been published indicate that these organisms can be affected by sodium chloride concentrations similar to what is found within the vicinity of roads (e.g. Czerniawska-Kusza et al. 2004); literature on this topic is reviewed in Environment Canada and Health Canada (1999), and Butler and Addison (2000).

Soil microfauna (e.g. nematodes, mites) and macrofauna (e.g. earthworms, insects) are also important for nutrient cycling and for maintaining good soil structure (e.g. high porosity), and therefore for healthy plants. Again, it is likely that chemical deicers negatively impact these members of the soil community, but there is little data on the impact of chemical deicers them. However there are a few studies showing that these organisms can be affected by sodium chloride concentrations similar to what is found within the vicinity of roads; what is known is reviewed in Environment Canada and Health Canada (1999), and Butler and Addison (2000).

Plants will vary in their ability to deal with the stresses caused by changes in the physical, chemical, and biological changes to the soil caused by chemical deicers and sand. As a consequence, the competition between species will be altered, and thus some species will increase in abundance as others decline (Isabelle et al. 1987, Wilcox and Andrus 1987). This process often results in the proliferation of a few species that tolerate salty, infertile soils including a few native plant species (e.g. narrow-leaved cattail *Typha angustifolia*) and several invasive exotics (e.g. common reed-grass *Phragmites australis*) (Catling and McKay 1980, Wilcox 1986, Isabelle et al. 1987, Richburg et al. 2001).

Sand is used in conjunction with chemical deicers, and acts synergistically with them to reduce soil fertility to plants. Unlike sodium chloride, which either infiltrates into the soil or is transported away with water flow, the sand is less mobile. Accumulations of sand can alter the bulk density and porosity of the roadside environments. Sand deposition reduces soil organic matter, which lowers cation exchange capacity and therefore the ability of soil to retain nutrients. Sand does not retain water as well as silts and clay; therefore, water can flow through the sand carrying the essential cations with it. A study performed in Norway found that deicing road salt killed spruce seedlings grown in sand, whereas plants grown in peat, loam, and silt loam soils were only moderately affected (Fostad and Pedersen 2000).

Finally, there is a positive feedback between degradation in soil fertility and vegetation loss due to chemical deicers. The direct impacts of deicing road salt on plant health and the indirect effects due to changes in soil fertility can result in a loss of vegetative ground cover. This, in turn, can exacerbate the physical and chemical changes in the soil, causing further declines in plant cover (Environment Canada and Health Canada 1999).

Toxicity and the Zone of Damage

The most severe cases of salt damage to vegetation are within five meters of a road (Bryson and Barker 2002), but sensitive species may be affected by chemical deicer at distances greater than 200 m from a highway (Kelsey and Hootman 1992, Cain et al. 2001). The effect zone for vegetation typically extends up to 80 m distance from major four-lane highways, and up to 35 m for two-lane highways (for a graphical example of the shape of the road effect zone for a boreal plant, lowbush blueberry *Vaccinium angustifolium*, see Eaton et al. 1998). Roadside trees are exposed to chemical deicers at the roots in the soil, and at aboveground tissues via splashing and longer distance aerial transport and deposition of aerosolized deicer (Transportation Research Board 1991, Environment Canada and Health Canada 1999, Blomqvist 2002). Up to 60% of applied road salt is transported to the soil and vegetation within 40 m of a highway, and 90% of that deposition is within 20 m of the road (Blomqvist and Johansson 1999). Methods to protect roadside plants from salt exposure (e.g. protective straw mats) have been ineffective (Pedersen et al. 2000).

Chemical deicer enters soil incidentally as salt scatter during road application, via runoff water from the road surface, from melting contaminated snow transported from the road via plowing or wind, and by aerial deposition (Blomqvist 1999, Cain et al. 2001). For woody vegetation, damage can occur from soil sodium concentrations ranging from 67.5 – 300 ppm (depending on the species), soil chloride concentrations of 215 – 500 ppm, and NaCl concentrations of 600 – 5500 ppm. Herbaceous species (e.g. grasses) are somewhat more tolerant (Cain et al 2001, Richburg et al. 2001). Very high levels of soil sodium are found within 10 m of highways, and damaging levels (> 67.5 ppm) have been detected as far as 30 m (Hofstra and Smith 1984, Cain et al 2001). Damaging levels of soil chloride (> 215 ppm) have been detected as far as 200 m from a highway (Cain et al 2001). The highest levels recorded in roadside soils subject to deicing road salt exposure were 1050 ppm Cl and 890 ppm Na (Cain et al 2001).

Aerial transport occurs from vehicle tire passage stirring up road brine into a salty spray, and by dry salt residue being up by air currents from passing vehicles. Once aerosolized, salt may be transported by wind until deposited onto the surface of plants or soil. Aerial salt spray is considered to be more important in salt accumulation than salt uptake through the roots (Viskari and Karenlampi 2000). The distance transported depends on the topography of the site, prevailing wind direction, the amount of roadside vegetation, and traffic speed and volume. In general, the effect zone (distance at which plant damage occurs from aerial deposition) is 40 m – 100 m from the road, but damaging levels of salt in plant tissues receiving aerially deposited sodium chloride can occur as far as 200 m distance (Cain et al. 2001). In some cases, aerial transport of road salt may extend up to 400 m (Blomqvist 2003).

Under some conditions, the effect zone of aerially transported deicing salt can be even farther. At an arboretum bordered by a very high volume multilane highway, white pines (*Pinus strobus*) were damaged at distances greater than 378 m from road. The principal zone of deposition in this study was within 122 m of the road, and deposition on trees occurred up to 15 m in height (Kelsey and Hootman 1992).

Toxicology studies of the effects of aerial exposure to deicing road salt or other deicers measure exposure by quantifying plant tissue concentrations, which are a function of the concentration and frequency of wet or dry deposition. Threshold values for woody plants are 200 - 16,100 ppm Na, and 800 – 70,700 ppm Cl. Foliar spray concentrations of 7140 ppm are known to damage woody vegetation (Cain et al 2001). The magnitude of damage caused by aerial transport depends on weather; frequent fluctuations in temperature and high atmospheric humidity increase damage, whereas dry air or rain can reduce damage (Viskari and Karenlampi 2000).

Sensitivity of Plants in the Adirondack Park

There is considerable disagreement among studies on the tolerance of trees to sodium chloride exposure via soil or aerial spray. This appears to be both a consequence of (1) variation among researchers in experimental design and measured responses, (2) variation in environmental conditions among studies (e.g. water availability, soil type), and (3) within species variation in salt tolerance (Allen et al. 1994, Cain et al 2001). In general, deciduous species are more tolerant to road salt than coniferous species (Bryson and Barker 2002).

For aerial exposure, on a scale of 1 (least harmed) to 5 (most harmed), the most salt-sensitive species (ranks 4 and 5) include several important trees and shrubs in the Adirondack Park (see Environment Canada and Health Canada 1999): American beech (*Fagus grandifolia*), red-osier dogwood (*Cornus sericea*), red pine (*Pinus resinosa*), white pine (*P. strobus*), white cedar (*Thuja occidentalis*), and eastern hemlock (*Tsuga canadensis*). Intermediate sensitivity to aerial exposure (rank 3) include red maple (*Acer rubrum*), sugar maple (*A. saccharinum*), paper birch (*Betula papyrifera*), gray birch (*B. populifolia*), speckled alder (*Alnus rugosa*), and basswood (*Tilia americana*). The most tolerant species (ranks 1 and 2) include white ash (*Fraxinus americana*), trembling aspen (*Populus tremuloides*), red oak (*Quecus rubra*), and staghorn sumac (*Rhus typhina*).

We did not encounter a similar ranking of relative sensitivity of Adirondack trees and shrubs to road salt exposure in soil. However, based on the compilation of data presented in Cain et al. (2001), it appears that most Adirondack species are relatively sensitive to sodium chloride in soil. In Canada, 53% of the principal forest tree species and 73% of the genera are classified as sensitive to road salt.

Many bog and fen plants found in the Adirondack Park appear to be sensitive to salt runoff, and may be replaced by cattails (*Typha)* or common reed-grass (*Phragmites australis)* where deicing salt runoff is high (Wilcox 1984, 1986, Richburg et al. 2001).

Mitigation Using Salt-Tolerant Native Plants

Most research on salt tolerant plants has focused on ornamental plants for residential areas, plants in saline soils in arid lands, or agricultural crops (Allen et al. 1994). There has been relatively little work on native plants of cold boreal forests, except to document sensitivity of tree species to exposure to sodium chloride via soil and aerial pray (reviewed in Cain et al 1999).

There is evidence that boreal species have considerable genetic variability for salt tolerance, and thus it may be possible to identify seed sources to restore native vegetation using salt-tolerant genotypes (Khasa et al. 2002). Recently, a series of studies have been published that evaluate species for remediation of highly saline mine-tailings that result from oil sands mining in northeastern Alberta, Canada. Under these conditions, plants are exposed to salt mainly via the substrate (Renault et al. 1998, 2000, 2001, 2004). Renault et al. (2001) identified red-osier dogwood (*Cornus sericea* = *C. stolonifera*) as the most suitable native woody species, and Renault et al. (2004) identified slender wheatgrass (*Agropyron trachycaulum* = *Elymus trachycaulus*) as the most suitable native grass species for remediation of saline soils with low organic matter in a cold boreal region. Both species grow in the region of the Cascade Lakes (Weldy and Werier 2005, see also Kudish 1992, Magee and Ahles 1999). Red-osier dogwood is classified as very sensitive to airborne highway spray (Environment Canada and Health Canada 1999), but occurs near Route 78 within the Cascade Lakes gorge (T.A. Langen, *personal observation*).

Section 1-3. **THE EFFECTS OF DEICING ROAD SALT ON AQUATIC ORGANISMS**

Modes of Impact

Constituents of deicing salts (sodium, magnesium, calcium, potassium, chloride) are essential nutrients for living organisms; each plays an irreplaceable role in cellular biochemistry. Increases in the concentration of ions caused by road salt contamination of freshwater affect the health of organisms by disrupting osmotic (water) balance and by directly interfering with cellular biochemistry (Evans and Frick 2001). Aquatic organisms have various ways of dealing with increased salinity, including passive tolerance, barriers to absorption, rapid excretion, or behavioral avoidance (Hart et al. 1991, Evans and Frick 2001). Closely-related freshwater organisms may vary widely in their tolerances of salinity (United States Environmental Protection Agency 1988, Evans and Frick 2001, Blasius and Merritt 2002). The most dramatic (and easily detected) consequence of increased salt concentration in water is death, but a subtler consequence is increased energy expenditure caused by physiological mechanisms to cope with increased salinity and by increased movement to avoid contamination. This leads to slower growth and lower reproductive output. Community composition can change, as less tolerant species are out-competed by more tolerant, and less-tolerant prey are more easily captured by predators (Evans and Frick 2001).

Not surprisingly, organisms can survive higher concentrations of contaminant during brief *acute* exposure than long-term *chronic* exposure (United States Environmental Protection Agency 1988, Evans and Frick 2001). In terms of chemical deicer toxicity, acute exposure typically occurs from contaminated runoff and snow-bank melt-water during thaw events. Acute exposure studies are most relevant to understanding exposures in drainage structures, streams, soil pore water, and some roadside wetlands. Chronic exposure studies are more relevant to groundwater, streams during base-flow, rivers, wetlands, ponds, and lakes (Environment Canada and Health Canada 1999, Evans and Frick 2001).

The vast preponderance of studies on toxicity of road salt or its chemical constituents are acute toxicological assays, with durations of 24 hours to 1 week. This is unfortunate, since most organisms endure chronic exposures due to contamination, so risks associated with chronic exposure are the most meaningful. The United States Environmental Protection Agency (USEPA) and Environment Canada have estimated the risks associated with chronic exposure by using the few studies that have comparable data on acute and chronic exposure (United States Environmental Protection Agency 1988, Evans and Frick 2001). For these studies, the factor difference is calculated between concentrations causing similar health affects at acute and chronic exposure (Acute to Chronic Ratio or ACR). Because the ACR varies across studies and species (ranging from 2 to 15), there remains considerable uncertainty about the effects of chronic exposure to chemical deicers for most organisms.

Other complications to interpreting toxicological studies include the type of ‘salt’ used, e.g. pure sodium chloride versus sodium chloride rock salt, which contains various contaminants and anti-caking agents (Environment Canada and Health Canada 1999). Toxicity results vary somewhat depending on the concentration of dissolved oxygen in the water (tolerance increases with dissolved oxygen concentration) and water temperature (Evans and Frick 2001).

Finally, toxicity varies depending on the type of salt applied. Generally, from most to least toxic: potassium chloride > magnesium chloride > calcium chloride > sodium chloride (United States Environmental Protection Agency 1988, Evans and Frick 2001). Calcium chloride and sodium chloride together are less toxic (for a given concentration) than either alone. Fish may be less sensitive to calcium chloride than sodium chloride, but invertebrates appear to be the opposite (Evans and Frick 2001).

Based on a review of toxicological studies, USEPA has produced some general guidelines that have been adopted by many other agencies in the US and Canada: (1) the 4-day average concentration of chloride (when associated with sodium) should not exceed 230 ppm more than once every three years. (2) The maximum 1-hour exposure should not exceed 860 ppm once every three years. (3) These guidelines may not be sufficiently protective for potassium chloride, calcium chloride, or magnesium chloride exposure (United States Environmental Protection Agency 1988). Some US state agencies have updated and reanalyzed USEPA (1988), and have recommended slightly different guidelines, while acknowledging that these guidelines do not protect the most sensitive species (Iowa Department of Natural Resources 2003). Evans and Frick (2001) estimate that 5% of freshwater species will be severely affected (i.e. mean lethal exposure) at chronic chloride concentrations of 213 ppm (95% confidence intervals (CI): 136 – 290), 10% affected at concentrations of 238 ppm (CI: 162 – 314), 50% affected at concentrations of 563 ppm (CI: 505 – 622), and 90% affected at concentrations of 1341 (CI: 1254 – 1428). Most organisms that have been tested can survive acute exposures greater than 1000 ppm chloride (Evans and Frick 2001).

For human drinking water, the USEPA recommended chloride concentration is 25 ppm, and the maximum concentration for potable water = 250 ppm chloride (the threshold for taste). The American Medical Association suggests a maximum sodium concentration = 22 ppm for people on salt-restricted diets (United States Environmental Protection Agency 1988, Moran et al. 1992).

Comprehensive reviews of the impacts of chemical deicer on freshwater organisms include Hanes et al. (1970), United States Environmental Protection Agency (1988), Transportation Research Board (1991), Hart et al. (1991), Lewis (1999), Environment Canada and Health Canada (1999), Evans and Frick (2001), Fischel (2001), Blasius and Merritt 2002, and Ramakrishna and Viraraghavan (2005).

Phytoplankton and Periphyton

Primary producers, including cyanobacteria and eukaryotic algae, are more responsive to chloride contamination than larger organisms. Phytoplankton communities can change markedly in composition as salinity rises, due to direct toxic effects on sensitive species, increases in concentrations of toxic metals, and changes in the availability of phosphorus or other nutrients. Contamination from chemical deicers can reduce or increase primary productivity (Dickman and Gochnauer 1978, Dixit et al. 1999, 2000; Frick and Evans 2001, Judd et al. 2005). Some phytoplankton species have elevated mortality to chronic exposures as low as 71 ppm chloride, but the most marked changes occur when chronic chloride concentrations reach around 580 ppm (United States Environmental Protection Agency 1988, Environment Canada and Health Canada 1999). Increases in cyanobacteria production due to road salt contamination have been reported (Ramakrishna and Viraraghavan 2005). Low nutrient lakes, such as those located on the boreal shield, are at the greatest risk of changes in primary producer community abundance and composition due to road salt contamination, and changes may occur with increases as low as 2 – 10 ppm chloride (Dixit et al. 1999, Environment Canada and Health Canada 1999).

Zooplankton

Zooplankton are more sensitive to deicing salt contamination than fish, but contamination due to chemical deicers in most water bodies is usually well below the level that is toxic for these organisms (Evans and Frick). Increased salinity and meromixis has resulted in alterations of a zooplankton community toward small-body forms, which may have been caused by alterations in vertical migration patterns due to oxygen depletion at depth, and due to changes in the phytoplankton community (Judd et al. 2005). Zooplankton hatching from sediments is reduced at chronic exposures of over 200 ppm artificial sea salt (predominantly sodium chloride; Nielsen et al. 2003).

Macroinvertebrates

Macroinvertebrates are generally more sensitive to sodium chloride contamination than fish (Evans and Frick 2001), but concentrations of chloride in road-salt contaminated streams and lakes are typically well-below levels that are toxic to aquatic insects and other macroinvertebrates (Blasius and Merritt 2002). Demers (1992) found that road salt contaminated streams had altered aquatic insect communities, whereas Blasius and Merritt (2002) did not detect any difference in benthic macroinvertebrate communities between road-salt contaminated and uncontaminated streams. As in other water-quality impairments (e.g. Lewis et al. 2001), Ephemeroptera are among the most sensitive insects to road salt contamination, whereas Diptera are the least (Evans and Frick 2001).

Williams et al. (1997, 1999) sampled benthic macroinvertebrates in 23 springs (chloride concentrations ranging 2 ppm to 1345 ppm). They found that benthic macroinvertebrates provide a valid predictor of chronic chloride contamination, and created a quantitative ‘Chloride Contamination Index’. Of the 34 taxa collected among the springs, the most salt-tolerant included oligochaetes, copepods, snails, nematodes, and ceratopogonid and tipulid dipterans. The least tolerant included Tardigrada, Turbillaria, an amphipod, a caddisfly, and two stonefly species.

Benbow and Merritt (2004) conducted acute (24 h) and chronic (15 d) exposure to sodium chloride road salt to an ephemeropteran insect, a dipteran insect, a snail, and an amphipod. The LC_{50} estimates for these species was well above the measured chloride concentrations in 43

Michigan wetlands sampled by the researchers, indicating that road salt contamination does not place these species at heightened risk of mortality.

In experiments on stream drift, a behavioral response of macroinvertebrates to environmental stress, there was no impact until acute chloride concentrations were above 800 ppm (Crowther and Hynes 1977). Blasius and Merritt (2002) concluded that there is no effect on leaf-pack colonizing macroinvertebrate composition and abundance due to road salt contamination in Michigan streams, and that acute exposure to sodium chloride concentrations caused by road salt contamination are not high enough to affect the health of stream macroinvertebrates.

It is likely that benthic macroinvertebrates are severely impacted when lakes develop meromixis due to road salt contamination, but there has been little research published on this topic (Judd et al. 2005).

Fish

Freshwater fish appear to tolerate salinities in large streams and lakes that are much higher than occur due to road salt contamination. Fish with requirements for high oxygen concentrations and low water temperature (e.g. lake trout) are at risk if salt-impacted lakes become meromictic (Evans and Frick 2001).

Amphibians

Few studies have examined the impact of chemical deicers on amphibians, but given their sensitivity to most forms of chemical pollution (Semlitsch 2003), there is reason for concern. Sanzo and Hecnar (in press) found deleterious effects to development of wood frog (*Rana sylvatica*) tadpoles from chronic exposure to sodium chloride at concentrations as low as 78 ppm. Vernal pools near roads can be contaminated by road salt to the extent that salamander and frog embryonic survivorship is impaired (Turtle 2001, Karraker 2006.).

Acute exposures to magnesium chloride at concentrations expected from deicer applications had no effect on boreal toads *Bufo boreas.* Levels of magnesium chloride contamination at two roadside breeding sites of the endangered toad were below the measured no-effect concentration (Lewis 1999).

Section 1-4. **ENVIRONMENTAL IMPACTS OF CHEMICAL DEICING ALTERNATIVES TO SODIUM CHLORIDE**

Chemical deicers are evaluated based on effectiveness at expected temperatures and other weather conditions, ease of application, cost, corrosiveness to metal and other materials, and environmental impact. Damaging environmental contamination may be caused by the principal component (e.g. magnesium chloride), additives (e.g. anti-caking or anti-corrosion compounds), and contaminants (e.g. metals, phosphorus, nitrogen). Throughout the previous sections of this review, the impact of the main ingredient of the most widely used chemical deicer, sodium chloride, was emphasized. In this section, we will focus on the known environmental impacts of other chemical deicers used on highways, deicer additives, and contaminants. Published research on the environmental impact of alternative chemical deicers is much less complete and comprehensive than for sodium chloride. Reviews of the environmental impacts of chemical deicers and their additives and contaminants include Transportation Research Board (1991), D'Itri (1992), Environment Canada and Health Canada (1999), Lewis (1999, 2001), Fischel (2001), and Ramakrishna and Viraraghavan (2005). Information on chemical deicer composition and performance can be found at Pacific Northwest Snowfighters http://www.wsdot.wa.gov/partners/pns/ .

Additional Concerns with Rock Salt

Besides the environmental impacts described in the previous sections, there are a few other potential impacts resulting from additives and contaminants to rock salt. These are poorly investigated relative to the direct effects of sodium chloride contamination.

Sodium chloride, in the form of rock salt applied to highways, includes trace contaminants comprising up to 5% of the total salt weight (Oberts 1986, Environment Canada and Health Canada 1999). As assayed by the Michigan Department of Transportation (1988), substances that can be present include phosphorus (14 – 26 mg/kg salt), nitrogen (7– 4200 mg/kg), copper (0 – 14 mg/kg), and zinc (0.02 – 0.68 mg/kg). Phosphorus and nitrogen may promote growth of algae and macrophytes in water bodies, whereas the metals may promote or inhibit the growth of microbes.

Sodium ferrocyanide and ferric ferrocyanide are added to sodium chloride and other chloride salts as an anticaking agent, in amounts of 30 – 124 mg/kg salt (Environment Canada and Health Canada 1999). In soil and groundwater, sodium ferrocyanide forms a stable, nontoxic precipitate (ferric ferrocyanide). In solution under direct sunlight, however, it can photodecompose into cyanide to form volatile, highly toxic hydrogen cyanide. The rate of the reaction depends on temperature, light intensity, pH, and initial concentration. There are multiple pathways for oxidation, chelation, and adsorption that result in less toxic species (Environment Canada and Health Canada 1999).

The risk assessment conducted by Environment Canada concludes that at present levels of road salt use, sodium ferrocyanide is unlikely to have adverse effects on land plants or soil organisms. It may potentially harm aquatic microorganisms, aquatic plants, macroinvertebrates and aquatic vertebrates in water-bodies with the most severe contamination (e.g. urban runoff) and favorable conditions for conversion to hydrogen cyanide species (Environment Canada and Health Canada 1999). Unfortunately, to date there has been little field monitoring and impact assessment of ferrocyanide (Novotny 1998, Paschka et al. 1999, Ramakrishna and Viraraghavan 2005)

Other Chloride Salts: Magnesium Chloride, Calcium Chloride, Potassium Chloride

Chloride salts used in conjunction with or in place of sodium chloride have some similar environmental effects, but also vary in terms of the impact on soil properties and toxicity to aquatic organisms. Chloride deicers are applied as a solid or liquid. Magnesium chloride and liquid calcium chloride are frequently applied with anti-corrosive substances that can impact the environment.

Toxicity varies depending on the type of salt applied. Generally, from most to least toxic: potassium chloride > magnesium chloride > calcium chloride > sodium chloride (United States Environmental Protection Agency 1988, Evans and Frick 2001, Fischel 2001). All chloride salts are non-toxic to aquatic organisms at typical exposure levels caused by use as chemical deicers. Calcium chloride has higher chloride per unit mass than sodium chloride, and is more corrosive. Calcium may be beneficial to some soils (Moran et al. 1992). Calcium chloride and sodium chloride together are less toxic (for a given concentration) than either alone. Fish may be less sensitive to calcium chloride than sodium chloride, but invertebrates appear to be the opposite

(Evans and Frick 2001). Sodium chloride is the only chloride deicer that attracts wildlife (birds and mammals) to roadsides, increasing the risk of collision (Fischel 2001).

Lewis (1999) reported on an extensive study of the environmental impact of magnesium chloride deicer. The impact depended on the source (some sources had significant contamination of metals and phosphorus) and whether anti-corrosive substances were present. He concluded that significant impacts were limited to 20 m from the road, and even within this distance other environmental impacts of roads were more noxious than chemical deicer contamination. Contamination of water bodies from runoff was dilute enough to have no impact on aquatic organisms, at least when no anti-corrosive additives were used.

Anti-corrosive additives to magnesium chloride and calcium chloride include various proprietary organic substances, including synthetic organic compounds and waste products from manufacturing agricultural products. A review of the composition, properties, and known environmental risks of these products is found in Fischel (2001). Baroga (2005) presents some field experiments comparing sodium chloride road salt with various salts containing corrosion inhibiters, and concludes that there is no significant difference in environmental impact.

Organic byproducts of fermentation, corn, barley or milk are combined with magnesium chloride, in a product referred to as Magic^{TM}. This product is applied directly as a liquid or as a pre-wetting solution with sodium chloride. Although less corrosive than sodium chloride, it can have high levels of phosphorus (70 – 100 ppm), depending on how it is manufactured (Albright 1992). The phosphorus is usually in the form of phytic acid, which chelates with calcium and iron ions and becomes insoluble in water. Laboratory experiments suggest that the phosphorus may not be bioavailable for algal assimilation (Albright 1992).

Albright (1992) examined the impacts to surface water of a switch from use of ‘abrasives’ (sand and sodium chloride, with phosphorus, silt and clay as contaminants) to sodium choride treated with Magic^{TM}. Total phosphorus and sediment load declined significantly after the switch, which was the management objective.

A similar product to Magic^{TM}, marketed as Ice Ban ^{TM}, was found to have unacceptable levels of copper, zinc, sulfate, nitrate, and phosphorus. The pH was unacceptably low (less than 4.0), and biochemical oxygen demand was unacceptably high. Contamination from this product may result in eutrophication of roadside water bodies. At low concentrations, this product may act as a fertilizer for land plants, but at higher concentrations may be toxic due to excessive ammonia (Fischel 2001).

A magnesium chloride additive product manufactured from corn, marketed as Caliber ^{TM}, contains very high concentrations of phosphorus, ammonia, and organic matter, and is likely to result in eutrophication and high biochemical oxygen demand of contaminated water bodies. At low concentrations, this product may act as a fertilizer for land plants, but at higher concentrations may be toxic due to excessive ammonia (Lewis 2001, Fischel 2001).

A recent comparison of phosphorus levels in deicer products indicated that corn-based products have unacceptably high phosphorus levels whereas levels in additive-free sodium chloride and magnesium chloride are low (New York State Department of Environmental Conservation 2002). One synthetic chemical deicer product (Ice B Gone, Sears Ecological Applications) had acceptably low phosphorus, as did one magnesium chloride + additive product (Magic-0, Sears Ecological Applications, formulation from Venezuelan sugar cane only).

Acetates: Calcium Acetate, Magnesium Acetate, Sodium Acetate, Potassium Acetate

Calcium acetate and magnesium acetate are mixed to form a chemical deicer (calcium magnesium acetate or CMA) that has been developed as an environmentally – benign alternative to sodium chloride (Transportation Research Board. 1991, Moran et al. 1992). CMA does not generally increase toxic metal mobilization in soils, and may in fact stabilize metal ions by forming less-toxic complexes between metals and a decomposition product of acetate, bicarbonate (Elliott and Linn 1987, Amrhein and Strong 1990, Amrhein et al 1992, Backstrom et al 2004). CMA preserves soil structure and permeability, reducing runoff and erosion and preserving organic matter and clay particles (Amrhein and Strong 1990, Amrhein et al 1992). Nevertheless, there remain some concerns about the effect of CMA on soil chemistry, and research is ongoing (Transportation Research Board. 1991, Ramakrishna and Viraraghavan 2005).

Eighteen species of plants were exposed to CMA and sodium chloride, either through the soil or aerial spray. Experiments were performed in greenhouses, outdoors in pots, and in the field. For most species, CMA was far less toxic to plants than sodium chloride, regardless of exposure route. Nevertheless, some plants did suffer adverse effects from CMA at the highest exposure levels (Leiser and John 1990, McFarland and O’Reilly 1992).

Goldman et al. (1992) added CMA at realistic concentrations (i.e. likely resultant contamination from use on roads) to water from high altitude, oligotrophic lakes and concluded that CMA had little effect on microbial communities. There was a small increase in primary production at the highest concentrations, and some small loss in productivity in experiments with bacteria and phytoplankton, possibly due to augmented bacterial growth. Other studies also report little impact on phytoplankton and periphyton communities from CMA (McFarland and O'Reilly 1992). However, some manufacturing processes to produce CMA result in high phosphorus contamination, and CMA from such a source may result in algal blooms (Horner 1990, Goldman et al. 1992, McFarland and O'Reilly 1992).

In most experiments, CMA exposure has little impact on zooplankton, macroinvertebrates or fish except at very high concentrations. In some experiments, CMA additions have resulted in increased mortality to zooplankton and macroinvertebrates, possibly due to lowered dissolved oxygen from increased bacterial respiration (McFarland and O'Reilly 1992).

The greatest risk from CMA contamination is increased BOD (biochemical oxygen demand) to water bodies. CMA degrades in about 3 - 6 d in soil, and therefore rarely enters groundwater (McFarland and O'Reilly 1992). Whereas McFarland and O'Reilly (1992) concluded that CMA is unlikely to contaminate groundwater, Horner (1990) came to the opposite conclusion and was concerned about high BOD causing anoxic conditions to develop in groundwater. McFarland and O'Reilly (1992) also concluded that most CMA will be degraded before reaching surface water, and under most types of loadings will not unduly impact dissolved oxygen levels in ponds and lakes. Under most scenarios, the worst case was a drop in dissolved oxygen equivalent to 2.5 ppm. Horner (1990) concluded that there can develop severe oxygen depletion under high but plausible loadings to ponds. However, there is great uncertainty about the impact of CMA on any particular water body, because of CMA's variable impacts on primary production, in part due to variation in the amount of phosphorus among different manufactured types of CMA. It is also unclear about the impact of CMA loadings on ice-covered lakes, where increased BOD may be problematic. Horner (1990), Transportation Research Board (1991), and McFarland and O'Reilly (1992) agree that use of CMA should be avoided when increased BOD is likely to have severe ecological consequences to a water body. Most at risk are poorly flushed ponds, watersheds where there is little dilution of road runoff into receiving waters, and where there is likely to be significant CMA loading beneath ice cover (Transportation Research Board. 1991).

Bang and Johnston (1998) evaluated a cellulose-derived product developed by the South Dakota Department of Transportation that is a mixture of sodium acetate and sodium formate, marketed as Ice Shear ^{TM}. The product decomposes rapidly in soil and water, similar to CMA. Bang and Johnston's (1998) limited toxicological testing and review of the literature indicated that sodium acetate/formate is relatively non-toxic to aquatic organisms and land plants. It may act as a fertilizer of land plants, but results were idiosyncratic; lettuce was negatively impacted by this product. Sodium acetate/formate increased biochemical oxygen demand in a way that is similar but not as severe as CMA. No research has been published on the effects of sodium acetate/formate on soil, but it is likely to cause similar negative physical and chemical changes as sodium chloride (Fischel 2001).

All acetate-based deicers are expected to result in some oxygen depletion in soils and waters, and therefore present an environmental risk to slow-flowing streams and ponds near highways where application rates are high (Fischel 2001). Toxicity to aquatic organisms appears to be low, with CMA the least toxic, and potassium and sodium acetates somewhat more toxic. Potassium acetate was found to inhibit plant growth at the high exposure levels expected under heavy road application (Fischel 2001).

Sand

Sand is used typically as an abrasive, but is also used as a deicer at low temperatures at which other deicers are ineffective, or because it is the least expensive option for snow and ice control. Sand is quite variable in composition, depending on the source, and may include a substantial portion of soil particles that are clay or silt (Oberts 1986, Albright 2005). Sand usually has sodium chloride as an additive, and may have phosphorus and metals as contaminants (Oberts 1986, Fischel 2001, Albright 2005). In some instances, sand may be a significant source of phosphorus to roadside water bodies (Albright 2005).

Sand has four principal impacts on the environment (Molles 1980, Fischel 2001, Staples et al. 2004, Shi et al. in review). (1) Refuge sand collects on roadside soils and plants, altering soil physical and chemical structure, and reducing photosynthetic efficiency of plants. Roadside soils subject to heavy sand use are less stable and highly subject to erosion. (2) Metal contaminants in sand may negatively impact communities of soil microorganisms. (3) Sand use dramatically increases airborne particulate matter, particularly particulate smaller than 10 μm diameter, and is a principal cause of deteriorating air quality in some urban and rural regions of North America. (4) Sedimentation of water bodies from sand use increases suspended solids and turbidity, and buries benthic habitat for macroinvertebrates, periphyton, and fish spawning sites. Increased turbidity reduces primary production, and sand particles negatively impact fish respiration.

Unacceptable levels of sedimentation caused by use of sand has resulted in the decline in its application (Albright 1992, Fischel 2001, Staples et al. 2004, Shi et al. in review). Particle sizes less than 6.35 mm are considered to be a problem in terms of environmental impact on streams due to excessive sedimentation, and particle sizes less than 2 mm are especially problematic. In samples of traction sand applied in Montana, 90% of the particles were smaller than 6.35 mm and 60% were smaller than 2 mm (Staples et al. 2004). There is ongoing research on recovering sand abrasive or structural measures to prevent contamination of water bodies (Staples et al. 2004, Shi et al. in review).

CHAPTER 2

WINTER EVENT MANAGEMENT METHODS & TECHNOLOGIES – A REVIEW

Motorist's view of Lower Cascade Lake, while driving east (in the direction of Keene, NY). Photo by Michael Twiss.

Chapter 2. **WINTER EVENT MANAGEMENT**

Report prepared by Kerop D. Janoyan and Benjamin T. Green.

INTRODUCTION

The purpose of this review is to identify winter management methods that are capable of reducing the need to use deicing chemicals. The review has been arranged into six sections which represent different fundamental approaches to environmentally sustainable winter maintenance. The sections are *Management Practices*, *Chemical/Material Spreading Methods*, *Snow Removal Machinery/Methods*, *Chemical-free Methods*, *Structural Solutions/Drainage*, and *Increased Traction*. The references that were used in the research of each of these topics are presented at the end of each section.

The intent of this review is to assist managers at identifying winter event management strategies and techniques that fit their particular situation, and thus consider more environmentally sound options for snow and ice removal. The twenty-two different ideas that are presented in the review differ in magnitudes of commitment and effectiveness. Before the main review is a brief description of each option including fiscal concerns, effort of implementation, and environmental impact.

OVERVIEW OF METHODS AND TECHNOLOGIES

2-1. MANAGEMENT PRACTICES

Better management practices such as the reduction of chemical deicer use through public awareness of road conditions and better understanding of deicer's abilities by removal crews are vital to an improved environmental outlook. Exploration into winter road quality standards and the acquisition of enhanced weather information to efficiently battle snow and ice are also helpful measures.

Winter Traffic Management

Through the strategic use of automated variable condition and speed signs, driver awareness of hazards is enhanced leading to greater public safety and allowing snow removal crews to do their jobs quickly, efficiently, with less material waste.

Public Safety: Increased driver awareness, reduced speeds, fewer careless accidents

Fiscal Implication: Require purchase and installation of automated signs, education of employees, powering signs in remote locations, reduced chemical use saves money

Effort of Implementation: Sign installation not difficult, need educated staff to be on call to keep signs accurately up to date

Environmental Issues: Less deicing chemicals applied, reduced speeds lends to less wasted chemicals, reduced speeds means less harmful traffic emissions

Deicing Road Salt Application Management

Through better organization of road salt application techniques, less chemical can be spread and greater effectiveness can be obtained.

Public Safety: Better road salt management means better road conditions, better organization of removal efforts reduce delays, increase safety

Fiscal Implication: Require the time to reorganize from the top down and educate removal crews of new policies, less chemical use saves money

Effort of Implementation: Time and effort to devise new management plan, put plan in working order, make continued improvements

Environmental Issues: Less deicing chemicals applied

Level of Service

Through reductions in levels of service where applicable, less material deicer is used and wasted.

Public Safety: Decreased public safety would not be tolerated, although greater driver responsibility is necessary

Fiscal Implications: Less removal and deicing effort saves money

Effort of Implementation: Removal efforts curbed

Environmental Issues: Reduced removal efforts and chemical usage

Road Weather Information Systems (RWIS)

Through the use of up to the minute weather data provided by RWIS, sound decisions and proper action can be implemented for every condition and situation.

Public Safety: More accurate weather information improves snow removal crew timing improving road conditions

Fiscal Implications: Requires purchase and installation of weather stations and pavement sensors, education of personnel to interpret weather data, savings in reduced deicer usage

Effort of Implementation: Weather station installation, required education and restructuring of removal efforts

Environmental Issues: Concise removal efforts leads to reduced chemical usage

2-2. CHEMICAL / MATERIAL SPREADING METHODS

Chemical deicer use can be reduced through the implementation of new equipment and techniques for spreading solid material and liquid chemicals.

Prewetting Equipment

Solid material deicers can be prewet before application to improve melting time, increase chemical bond to ice, and reduce losses to bounce and scatter.

Public Safety: Improved melting times and less material loss leads to cleaner, safer roadways

Fiscal Implication: Choice of prewetting technique effects cost, prewetting stockpile, prewetting in truck hopper, prewetting from truck prior to application, less chemicals used

Effort of Implementation: Require installation and maintenance of equipment if necessary

Environmental Issues: Less deicing chemicals applied, reduced loss of deicing material, residual effects of previously applied salts further reducing usage

Solid Material Spreaders

Properly functioning, automated spreaders are capable of providing excellent deicer coverage while minimizing material waste.

Public Safety: Improved material coverage means cleaner, safer roads

Fiscal Implication: Requires purchase of new equipment or calibration of existing spreaders, reduction in material usage, savings in chemical use

Effort of Implementation: Requires understanding of proper spreading rates for material and conditions, calibration, and cleaning/maintenance

Environmental Issues: Less deicing chemicals applied, reduced loss of deicing material

Anti-icing

Anti-icing is the technique of applying deicing chemicals before a winter event. This prevents bonding between snow and the roadway facilitating easy removal.

Public Safety: Snow and ice are not allowed to bond to road improving driving conditions and safety

Fiscal Implication: Requires purchase of anti-icing equipment such as tanks and storage equipment, although solid material can be used, cost of education of personnel, savings in chemical use

Effort of Implementation: Requires understanding of anti-icing and restructuring of winter maintenance

Environmental Issues: Less deicing chemicals used, reduced losses due to chemical bonding to roadway

Abrasives

When temperatures are below the effective temperature of deicing chemicals, abrasives can be applied to increase the coefficient of traction on the roadway

Public Safety: Increased traction can lead to greater public safety if temperatures are too low for chemicals to be effective, air born sand particles can lead to respiratory problems

Fiscal Implication: Abrasives are cost effective and readily available

Effort of Implementation: Already in use to some degree in all winter maintenance circles, traditional spreading although spring cleanup necessary

Environmental Issues: Air born particles can cause air pollution, rough on roadside vegetation and nearby surface water aquatic life

Automated-Fixed Chemical/Hot Water Sprayers

Mounted to guardrails or in the road surface, these deicing sprayers provide timely chemicals to prevent snow pack lending to easy removal.

Public Safety: Properly timed application of chemicals keeps roads clear and safe

Fiscal Implication: Extensive installation and equipment purchase is necessary, savings in chemicals

Effort of Implementation: Once in place, requires simple maintenance, weather information reviews, and deicing chemical refills.

Environmental Issues: Like anti-icing, less chemical material is used and wasted

2-3. SNOW REMOVAL MACHINERY & METHODS

Through the efficient use of snow removal equipment and methods, negative effects of deicing chemicals can be lessoned.

Snowplows

Extensively used anywhere regular snow accumulations occur, proper selection of plows and cutting edges to match conditions can make them more effective.

Public Safety: Plows designed to match the condition can lead to better maintained roads

Fiscal Implication: Cost of purchasing new plows, training operators to use them effectively, savings in reduced deicing material usage.

Effort of Implementation: Training personnel, trucks readily available

Environmental Issues: Reduced use of deicing chemical

Snowblowers

Capable of removing accumulated snow from the roadway either far to the sides or into a truck for off hauling. Good for used around guardrails and for removing contaminated snow banks

Public Safety: Efficiently remove accumulated snow or drifts

Fiscal Implication: Cost of purchasing snow blowers, training operators to used them effectively, loss of money in off winters

Effort of Implementation: Transporting snowblower to site, training personnel

Environmental Issues: Remove contaminated snow

Hauling Snow

Removing contaminated snow from a site is a good way to prevent roadsalts and other pollutants from reaching watercourses in spring time thaw.

Public Safety: Remove accumulated snow and snow banks leading to cleaner roads and better visibility

Fiscal Implication: Costs of manpower and machinery to remove and haul snow

Effort of Implementation: Organize manpower and machinery

Environmental Issues: Remove contaminated snow

Snow Melting Machinery

Loader fed or self propelled machines are capable of melting contaminated snow leading to a reduction in handling.

Public Safety: Remove accumulated snow and snow banks leading to cleaner roads and better visibility

Fiscal Implication: Cost of purchasing or leasing snowmelter, training personnel to operate it, transportation costs

Effort of Implementation: Training, coordination with other machinery

Environmental Issues: Remove contaminated snow, melted water an issue if not sent into a municipal storm water system

Snow Fences

By reducing wind velocity over the road, these fences have traditionally stored snow. New style snow fences accelerate wind over the roadway to prevent accumulations.

Public Safety: Prevent snow from accumulating on road, thus making roads less slippery

Fiscal Implication: Cost of purchasing and installing snow fences, reduced chemical usage and mechanical removal methods

Effort of Implementation: Installation and maintenance if needed

Environmental Issues: Reduced use of deicing chemical, less removal effort

2-4. CHEMICAL - FREE METHODS

These solutions boast the complete elimination of dependency on deicing chemicals to keep roadways clear of snow and ice.

Electrically Conductive Concrete Overlay

Snow and ice are not able to bond to the roadway because the surface is heated by the electrical resistance of steel fiber reinforced concrete in line with a power source.

Public Safety: Prevent snow from accumulating on the road making road surfaces clear and dry

Fiscal Implication: Cost of construction only requires installation of overlay, cost of powering units, continued operational power costs

Effort of Implementation: Construction effort, monitoring its effectiveness

Environmental Issues: Deicing chemical use eliminated, negative environmental impact of producing electricity to power conductive overlay

Geothermal Heat Pumps

Geothermal hot water that can be recharged during summer months is circulated through pipe networks under the road surface by pumps thus heating the roadbed and preventing snow and ice buildup.

Public Safety: Prevent snow from accumulating on road making road surfaces clear and dry

Fiscal Implication: Cost of construction is very high due to necessary excavation, monitoring system

Effort of Implementation: Construction effort, excavation intensive, monitoring its effectiveness

Environmental Issues: Deicing chemical use eliminated, no major environmental issues if operating properly

Covered Highway

A structure completely enclosing the highway would prevent the elements from coming into contact with the road surface and therefore eliminate the need to use deicing chemicals in that area.

Public Safety: The road surface would constantly be in a clear and dry state

Fiscal Implication: Cost of design of such a structure and construction would be expensive, routine maintenance on the structure would also have to be performed, no winter maintenance needed in that area

Effort of Implementation: Construction effort, structure maintenance

Environmental Issues: Deicing chemical use eliminated, no major environmental problems, air pollution from ventilation system

2-5. STRUCTURAL SOLUTIONS/DRAINAGE

By controlling the path and ultimately the end location of roadway runoff, contaminates can be kept from entering natural water supplies.

Drainage System/ Runoff Control

A closed drainage system in conjunction with other components such as infiltration trenches and detention basins can control the outcome of contaminated runoff.

Public Safety: Snow and ice removal methods would remain unchanged

Fiscal Implication: Cost of design and construction would be expensive

Effort of Implementation: Construction effort, routine maintenance such as cleaning structures would also have to be performed

Environmental Issues: Contaminated runoff would be controlled and not allowed to pollute adjacent water sources

Vegetated Swales

Salt tolerant plants are capable of shielding vehicle spray, and in conjunction with a well drained soil, filter salt laden runoff.

Public Safety: Snow and ice removal methods would remain unchanged

Fiscal Implication: Reasonable cost of construction and maintenance

Effort of Implementation: Installation effort, routine maintenance

Environmental Issues: Contaminated runoff would be controlled, spray would be shielded

Runoff Filters

Filtering the roadway runoff ensures that many runoff pollutants, such as roadsalt, would not find their way into adjacent water sources.

Public Safety: Snow and ice removal methods would remain unchanged

Fiscal Implication: Cost of installation and cleaning/changing filters

Effort of Implementation: Construction effort, routine maintenance

Environmental Issues: Contaminated runoff filtered, reducing contaminates in the surrounding waters

2-6. INCREASE TRACTION

Promoting good drainage and high skid resistance are important factors to safety as are increasing surface traction.

Skid Resistance

Properly graded mix designs and grooved pavement can increase surface traction and leading to less deicing chemical need.

Public Safety: Snow and ice removal methods would remain unchanged

Fiscal Implication: Cost of installation of new overlay

Effort of Implementation: Construction effort

Environmental Issues: Small reductions in deicing chemicals needed

Tire Chains/Studded Snow Tires

Surface traction is increased while stopping distances are decreased when tire chains or studded snow tires are used in inclement weather conditions.

Public Safety: Increased surface traction and stopping distance make roads safer

Fiscal Implication: Cost of road maintenance due to damage caused by chains/studs, enforcement

Effort of Implementation: Enforcement and maintenance of road damage

Environmental Issues: Less deicing chemicals are needed

Section 2-1. **MANAGEMENT PRACTICES**

Winter Traffic Management (Signage)

Traffic monitoring and management in winter conditions can increase public safety by alerting motorists to possible dangers. Due to increased driver awareness, less deicing chemicals need to be applied to the roadway leading to a less severely impacted roadside environment. These information systems receive data gathered from road weather information stations and post it via electronic signs and broadcast it over the radio waves for public benefit. In places like Sweden and Switzerland, motorists are encouraged and even paid to use cellular phones to call in inclement weather conditions to aid broadcasters in accessing accurate information (Research Results Digest 1999).

In many places, newspapers provide information from authorities pertaining to safe travel and safe driving practices. French road authorities even use the newspaper, internet, and other forms of public announcement to educate motorists concerning the meaning of road sign message board postings (Research Results Digest 1999). It is believed that this additional driver understanding of road dangers will reduce accidents while allowing crews to spread less deicing chemicals.

Vehicular traffic affects the pavement surface by compacting snow, making surfaces more slippery. This effect is worsened by heat from tire friction and vehicle operation. Traffic also tends to blow deicing chemicals and abrasives from roadways. For these reasons, up to the date road conditions provided to motorists are important to public and environmental well being (FHWA Effective Anti-Icing Program 2004). Variable speed limit and condition signs remotely connected to road weather stations are vital to providing reliable information to drivers. Winter maintenance vehicles should also have on board sensors that project conditions to rear mounted message boards, alerting motorists of conditions and when deicing chemicals are being applied to roadways (Research Results Digest 1995). With this information, motorists are less upset by delays and may plan to use alternative routes.

Information from weather stations should be reviewed by a meteorologist before being posted on variable message signs. Road information should be simple messages to avoid driver information overload. Categories such as *good*, *possibly slippery*, and *verified slippery* are currently in use in the United States (Technical Research Centre of Finland 2002). In France, a four category system in used with conditions ranging from *normal*, to *delicate*, to *difficult*, to *impossible*. It is vital to use categories that closely resemble conditions while being understandable to motorists (Research Results Digest 1999).

It has been found that slippery road condition signs do lower mean speeds on highways. Variable message signs warning of hazards that are not obvious, such as black ice, have had the greatest effect on motorists. Polls taken with motorists who had driven by the signs proved that they had paid closer attention to changing conditions, passed more carefully, and even tested roadway traction (Technical Research Centre of Finland 2002).

Example of a variable message board (http://www.oksolar.com/visual_signals/heavy-traffic-signs.htm)

Variable speed signs have also been tested and implemented. Variable speed signs can be utilized in the case of accidents, congestion, construction, snow, ice, and fog (CTC and Associates et al. 2003). These signs were found to have a greater effect then the variable condition signs. Experiments conducted in Wisconsin showed 83%-91% of drivers who had passed the signs remembered the posted speed limit while only 66% recalled seeing the variable condition sign. 95% of drivers felt, however, that the signs were useful (Technical Research Centre of Finland 2002). On most variable speed signs, computers process the weather data and usually provide ample information for meteorologists to make decisions concerning postings. The most effective variable speed signs are those that display a light that means speeds are radar enforced.

These signs have potential to educate drivers of hazardous conditions, thus leading to higher public safety. With accurate weather information and road conditions, removal crews can streamline deicer use leading to greater environmental acceptance. At times, these variable message signs can cause a greater range in motorist speeds leading to potentially dangerous outcomes. It has been found that when exiting the area of minimized speed or hazardous conditions, rapid acceleration without heed of current conditions can lead to problems. In order for this system to be effective, drivers must be willing to have a less aggressive behavior towards winter driving and signs must be strictly enforced by police.

References

CTC & Associates LLC; WisDOT RD&T Program. April 9, 2003. Variable speed limit signs for winter weather. *Transportation Synthesis Report,* pp 1-4.

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm

Research Results Digest. January 1995. Winter maintenance technology and practices - learning from abroad. Number 204: 7-8.

Research Results Digest. April 1999. Report on the 1998 scanning review of European winter service technology. Number 238: 4-5.

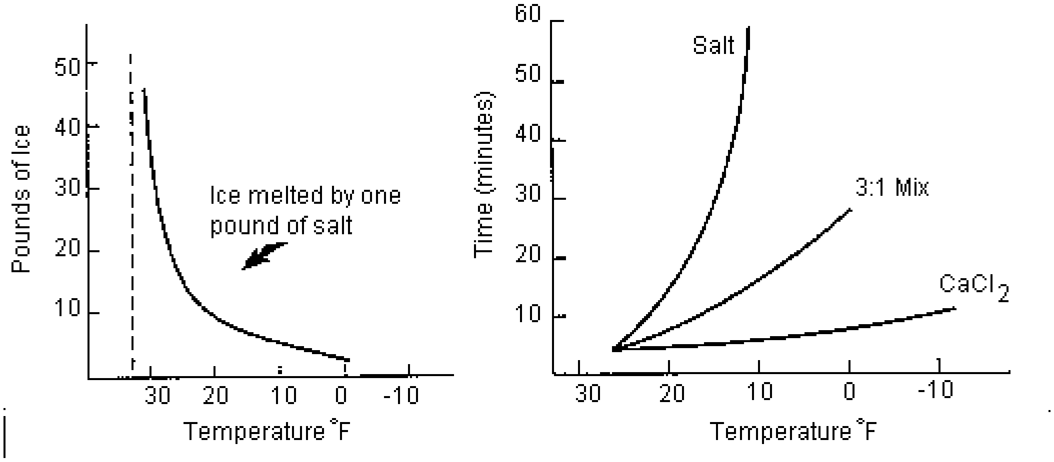
Technical Research Centre of Finland (VTT). 2002. Effects of weather-controlled variable message signing on driver behavior. Number 1: 1-4.

Deicing Road Salt Application Management

The application of road salt has long been the norm for winter deicing operations in the United States. In recent years, negative environmental impacts have developed through the long term use of these chemicals. Many alternatives have been explored but it has been difficult to match the convenience and low price of using deicing salts. Through better road salt management practices, reductions in application, and an improved training on best practices, many of the negative impacts can be decreased.

Objectives for road salt management should be focused on safety, environmental protection, continued improvement, economic issues, efficient transportation systems, accountability, measured progress, and communication (Transportation Association of Canada 2002). Safety is always the main priority of any department of transportation.

The easiest way to reduce the environmental impacts of roadsalt is to reduce the amount that is being applied. The United States alone used 20 million tons of deicing salt in 1994 (Road salt and Winter Maintenance for British Columbia Municipalities 1998). Pavement temperature and timing are the most important factors in the application of deicing roadsalts. As the pavement temperature decreases, more roadsalt is needed to clear the roadway. Salt can melt five times as much ice at 30^{o}F then at 20^{o}F. The following graphs show the relationship between pavement temperature and effect of roadsalt on ice (U.S. Roads 2004).



Road salt melting ability by temperature and time for various compounds to melt 1/8" of ice (U.S. Roads 2004)

Understanding of the left-hand side plot is vital to proper road salt management practices. In addition, if too much chemical is used, not all of it will dissolve into solution, thus leaving wasted amounts. The application of too little chemical will not lower the freezing point of the ice so it will either not melt or it may melt only to refreeze. Either problem is a waste of chemical resources. In terms of timing, early application is essential, as spreading a small amount of road salt when the snow is loose reduces it to slush which is cannot be compacted by traffic and can be easily plowed away (U.S. Roads 2004).

In order to maximize the level of service while minimizing roadsalt dependency, certain measures should be taken. The level of service of every roadway should be documented. Type of deicer, quantity, and application rates should be calculated and clearly defined for every location and situation. Available deicing equipment should be organized by prewetting ability, type of application, and type of spreader control (electronic of manual). The equipment should then be assigned a suitable route matching its abilities. Road weather information should be intimately used in all decisions (Transportation Association of Canada 2002). The roll of prewetting deicing materials is vital to application accuracy and reduction.

In addition to road weather information systems, pavement temperature sensors should be installed on all removal equipment. Effective plowing practices should be exercised and the use of snow drift control in the form of snow fences should be utilized (Transportation Association of Canada 2003). Snow removal personnel should be trained in roadsalt conservation. They should be made aware and instructed to follow all standards to ensure proper applications (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). The following application guidelines are an example of desired application rates per condition and temperature.

Application	Description	Application Rate
light application	to prevent black ice when the surface temperature is near freezing with light snow or sleet	212 pounds per two lane mile (about 0.065 cubic yards)
average application	early in the day when the surface temperature is 25° F and rising under snow, sleet or freezing rain conditions	300 pounds per two-lane mile (about 0.093 cubic yards)
heavy application	early in the day when the surface temperature is 25° F and stable or when the surface temperature is 21° F and rising or late in the day when the surface temperature is 25° F and rising, under conditions of packed snow or ice on the highway surfaces	460 pounds per two-lane mile (about 0.145 cubic yards)

Roadsalt application guidelines (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998)

References

Road salt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html

Transportation Association of Canada. September 2003. Vegetation management.: 1.

Transportation Association of Canada. July 2002. Salt management plans.: 1-5.

U.S. Roads. 2004. *Using Salt and Sand for Winter Road Maintenance*. http://www.usroads.com/journals/p/rmj/9712/rm971202.htm.

Level of Service

In the United States, a goal of winter snow and ice removal is to prevent accumulation of snow and ice on the road surface, with a goal of maintaining 'bare pavement'. The policy was instituted and is followed because it is believed that public safety is much greater on bare, dry pavement (FHWA Effective Anti-Icing Program 2004). All roads generally have this policy, although secondary roads and residential roads do not generally receive the maintenance to achieve this goal in as timely a manner as highways. Using preset levels of service, most American municipalities include 24 hour maintenance until bare pavement is achieved.

A comparison conducted concerning the available snow removal resources and funding of the United States, Japan, and Europe yielded interesting results. Scaled comparisons between the countries found that Japan and Austria allotted 2 to 3 time more funds then the United States respectively. Equipment and field employee responsibility is approximately 8 and 16 times greater in areas of the United States then in Austria (Research Results Digest 1995). This proves that with better funding and management, the United States could provide a higher level of service to the public while maintaining safety and reducing chemical usage.

Generally a municipality or state DOT will define levels of service for given roads they maintain in order to organize their deicing and snow removal efforts. For example, some organizations scale their level of services from "A" through "F", others use a numerical system. Level "A" would be a major route given excellent maintenance care as the surface is kept clear and the road is delay free. Level "F" would be a poorly maintained, little used road in which delays would be prominent and reaction time to plow and apply deicing chemicals would be unknown or even indefinite (Washington State DOT 2005).

Winter road management standards and the surface levels

Standards for highways and trunk roads	Standards for sub-arterial roads A and B	Standards for residential streets	
Surface level 4	**Surface level 3**	**Surface level 2**	**Surface level 1**
Powder snow, large snowflakes, slush	Compacted snow, larger snowflakes with underlying ice layer	Ice film, ice sheet, powder snow with underlying ice layer	Very slippery ice film, ice sheet, compacted snow

Numerical Level of Service definition (City of Sapporo 2005)

In areas of Japan, snow and ice removal crews are not deployed until several inches of snow has accumulated on the roadways. This level of service, however, is known and expected in those regions so drivers take more responsibility and exercise caution. In parts of Germany, snow-clearing operations are only done from 4:00AM to 8:00P.M, when the majority of the motorists are on the road. In Scandinavian countries, the level of service that is provided is a function of coefficient of traction measurements that are taken using a truck mounted friction meter (Research Readers Digest 1995).

There are many ways to define and reach levels of service. The “bare pavement policy” is in effect for any sizable road in the United States. Reducing the levels of service on these roads would reduce the manpower, equipment costs, and environmental impacts inherent in snow removal. However, given the current attitudes of drivers, this option would lead to a great decline in public safety which will not be tolerated. The only chance for reducing levels of service is to increase driver awareness and education in winter conditions.

References

City of Sapporo. 2005. *Establishing a Level of Service for Winter Road Management.* http://www.city.sapporo.jp/kensetsu/yuki/english/plan/plan3-2.htm.

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program.* http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 8.

Washington State Department of Transportation. 2005. *Highway Maintenance Levels of Service*. http://www.wsdot.wa.gov/maintenance/service.htm.

Road Weather Information Systems

Road Weather Information Systems (RWIS) supply accurate weather information along road sections. This helps maintenance managers and road crews to efficiently battle winter events as duration and magnitude of inclement weather can be precisely forecasted (Research Results Digest 1995). Information provided by such weather systems include real-time pavement temperatures, dew point, humidity, air temperature, wind velocity and direction, and even the amount of deicing chemical on the road surface (Roadsalt and Winter Maintenance for British Columbia Municipalities 2004).

The emphasis of these systems is to provide accurate snow, ice, and black ice conditions. Sensors are imbedded in pavement courses to provide temperature information while more traditional weather equipment is utilized to provide data at certain intervals along the roadway. All systems in the proximity should use common data standards to promote communications between systems and uniformity of collected weather information (Research Results Digest 1999). Even video cameras have been used to provide managers with visual conditions of the roadways.

Remote Weather Station (Federal Highway Administration 2004)

With the collected weather and temperature data, thermal mapping of the road surface can be performed (Research Results Digest 1999). Knowledge of pavement temperatures is the most crucial piece of weather information as it is the most important consideration in the bonding that occurs between precipitation and the road surface (Roadsalt and Winter Maintenance for British Columbia Municipalities 2004). This allows managers to see a projected thermal model of road surface temperatures along a given stretch of highway.

RWIS provide managers with the essential information to make educated decisions concerning maintenance and safety. The precise information aids in the decisions of quantities, types, and duration of resources needed to battle the event. RWIS helps removal crews to properly time deicing applications to maximize effectiveness while minimizing material usage, thus both saving money and reducing environmental impacts (Research Results Digest 1999). While services are improved with the use of RWIS, maintenance costs are reduced due to reduction in material usage and the elimination of unnecessary callouts (EPA 1999). Weather stations are typically spaced between 2 and 10 miles and general cost of upkeep of these stations are $275 to $500 per year per station depending on numbers of sensors and detectors (Research Results Digest 1999).

References

Federal Highway Administration. 2004. *Protecting a National Forest With New Snow Removal Methods*. http://www.fwha.dot.gov/winter/roadsvr/CS018.htm.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 7.

Research Results Digest. Apr. 1999. Report on the 1998 scanning review of European winter service technology. 238: 23.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality.* http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

United States Environmental Protection Agency. Sept. 1999. *Storm Water Management Fact Sheet: Minimizing Effects from Highway Deicing*.

Section 2-2. **CHEMICAL/MATERIAL SPREADING METHODS**

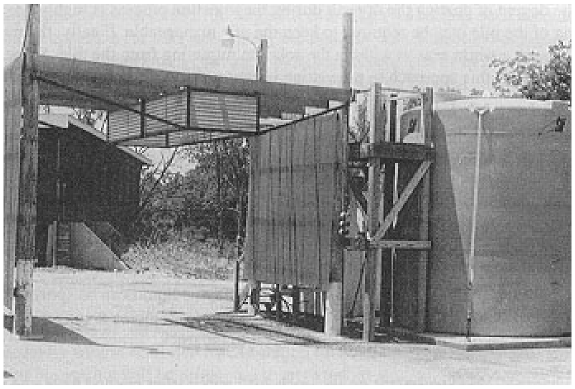
Prewetting Equipment

Prewetting is the process of adding water or a deicing solution to road salt or abrasives before or during application. The applied liquid covers the solid material deicer so that when it is applied, it embeds itself into snow and ice (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). A solid chemical particle needs energy to convert into a solution. Therefore, dry deicing particles must first form this coating before deicing can begin. By prewetting, this stage is eliminated leading to faster deicing times (FHWA Effective Anti-Icing Program 2004).

Prewetting solutions include water, sodium chloride, calcium chloride, magnesium chloride, potassium chloride, and calcium magnesium acetate in brine solutions (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). Water should only be used at higher temperatures soon before application as there is a risk of freezing at some point in the handling operations. Calcium chloride is quite commonly used as it has a lower freezing point then salt brine, thus lowering the effective temperature of road salt. Calcium chloride will also absorb moisture from the air helping to keep it on the road surface after deicing is complete by reducing the likelihood of it being wind blown off the road surface (FHWA Effective Anti-Icing Program 2004).

There are three main techniques that are used to prewet deicing materials. The first is to inject a prewetting chemical into a stockpile of deicing material. The second is the spray the prewet liquid into the spreaders hopper, and the third is to wet the material via a spray system as it is spread (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998).

Stockpile prewetting is advantageous because there is no need to purchase sprayer equipment, train personnel, or install storage tanks as the liquid retailer delivers and applies the prewetting agent. However, stockpiles must be kept inside to prevent dilution of prewetting agent, and even distribution of the agent throughout the pile is a common problem. Prewetting individual loads is another approach that requires only minimal equipment. Equipment includes a storage tank, centrifugal pump, piping, open spray area and a metering device at a general cost of $8,000 to $10,000. However, this procedure can be highly corrosive to the hopper and equipment and uniformity of application is still difficult to obtain.



Overhead spray bar for prewetting material while in hopper (FHWA Effective Anti-Icing Program 2004)

A spreader equipped with prewetting equipment is the most common and most effective alternative. Prewetting agents are applied in a uniform manner, directly to the deicing material as it is applied to the road surface. The only disadvantage of this method is the failure of the prewetting equipment on the truck due to operation in such extreme conditions (FHWA Effective Anti-Icing Program 2004).

Prewetting can lead to a savings of 20% to 30% of applied material due to reduced losses through bounce and scatter and the ability of the prewet material to remain on the road surface. This reduces salt that finds its way into spring time runoff and ultimately into the surrounding waters. In general, 6 to 10 gallons of prewetting liquid are applied per cubic yard of salt of deicing solid (U.S. Roads 2004). Additional advantages of prewetting include accelerated melting speeds due to additional moisture, wet particles adhere to pavement thus requiring less frequent applications, and effective temperature range of deicer is increased (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998).

References

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

U.S. Roads. 2004. *Using Salt and Sand for Winter Road Maintenance*. http://www.usroads.com/journals/p/rmj/9712/rm971203.htm.

Solid Material Spreaders

Spreaders are one of the most important pieces of equipment to reduce the impact of deicing chemicals on the surrounding environment. An evenly distributed deicing chemical applied at the proper rate can greatly decrease the amount of chemicals needed (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). For this reason, spreader calibration is of utmost importance. Different materials will require different spreading rates and care should be taken to match materials and conditions with proper calibration. Spreaders function in a hostile, low temperature, wet, and corrosive environment so they should constantly be cleaned and checked to ensure proper performance (U.S. Roads 2004).

There are several different types of spreaders that have all proved effective in deicing applications. The three types discussed here include the chute spreader, hopper spreader, and under-tailgate spreader. Chute spreaders dispense deicing chemicals in windrows on the road. This type of spreader is fed by raising the dump truck bed, and distribution is usually focused on the centerline of the road (U.S. Roads 2004). Hopper spreaders are usually an external V-box design that sets in a dump truck bed and feeds material though the use of full-length belts, chain-drag belts, or longitudinal augers at the base of the V-box. The material is spread by a semicircular spinner capable of spreading widths of 3 to 40 feet. The under-tailgate spreader is also an external unit that sets in a dump truck bed. The system includes a hopper, an auger feed mechanism, and a spinner disk (FHWA Effective Anti-Icing Program 2004).

State of the art spreaders boast noncorrosive parts such as stainless steel and various plastics (Research Results Digest 1995). Automated spreader control systems are also common. They are capable of maintaining constant application rates despite variable match belt or auger speed as well as truck speed (FHWA Effective Anti-Icing Program 2004). Air foils are being employed on the back of spreaders to keep deicer spray off of lights and message boards, reduce loss of material from bounce and scatter upon application, and improve visibility for following motorists (Research Results Digest 1999).

Air Spoiler on back of spreader (Research Results Digest 1999)

Properly calibrated spreaders are a key step to reducing material usage and negative environmental impacts. It was been discovered that between 100 to 300 pounds of material per lane mile is optimal. Care should be exercised to not apply any material when temperatures are below its melting abilities. Spreading operations should be focused on the centerline or crown of two lane roads and on the high side of curves to optimize deicer usage (U.S Roads 2004). Properly calibrated spreaders and care in application can greatly reduce the negative environmental impacts of salt usage.

References

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices - learning from abroad. 204: 3.

Research Results Digest. Apr. 1999. Report on the 1998 scanning review of European winter service technology. 238: 23.

U.S. Roads. 2004. *Using Salt and Sand for Winter Road Maintenance*. http://www.usroads.com/journals/p/rmj/9712/rm971203.htm.

Anti-Icing

There has been much confusion concerning the differences between anti-icing and deicing. Deicing is the common procedure practiced by most agencies today. It is the process of spreading chemicals to break the bond between ice and the roadway after accumulations have occurred. Conversely, anti-icing is the preemptive procedure of spreading chemicals before a winter event occurs to prevent ice bondage and facilitate easy removal (FHWA Effective Anti-Icing Program 2004). Anti-icing allows for a high degree of public safety while reductions in chemicals used are favorable monetarily and environmentally (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998).

Anti-icing is most efficient on heavily-used roadway. In order to implement the method, accurate climate and pavement temperature information is essential (U.S. Roads 2004). With anti-icing, timing is everything, and the proper weather data equipment and understanding is crucial to making proper decisions. The four main operations that make up anti-icing include solid chemical application, chemical solution application, pre-wetted solid chemical application, and plowing capabilities (FHWA Effective Anti-Icing Program 2004).

There are many different preventive materials that can be spread before a storm to keep roadways clear of snow and ice. Dry solid chemicals such as sodium chloride or calcium chloride can actually be effective for anti-icing as long as the roadways are damp. Moisture on the roadway is crucial because it is necessary to prevent material loss during spreading, wind, and blowing action of vehicles. The moisture also helps to form salt brine. This effect can also be obtained on dry roads by using prewet solid chemicals (FHWA Effective Anti-Icing Program 2004).

Liquid chemicals are ideal for temperatures of around 23°F and can be applied at up to 30 mph with uniform distribution (FHWA Effective Anti-Icing Program 2004). Liquids can be applied at low rates of just 25 to 50 gallons per mile, providing a damp surface. Liquid anti-icing chemicals remain on the pavement for up to several days (U.S. Roads 2004). Liquid chemicals can be used for temperatures below 23°F by increasing applications levels. Commonly used chemicals include sodium chloride, magnesium chloride, calcium chloride, calcium magnesium acetate, and potassium acetate (FHWA Effective Anti-Icing Program 2004).

Solid chemicals can be spread through the use of conventional mechanical spreaders, as mentioned previously. It should be noted again that the calibration of those spreaders to match materials and conditions is of utmost importance to maximize safety while minimizing chemical use. For liquid application, spreaders are either of the spinner or nozzle distributor bar variety. Tanks can either be mounted in the bodies of dump trucks, on flat bed trucks. Chassis mounted tanks are available although they are not as versatile for off-season use. All tanks and distribution equipment should be made of non-corrosive material and regularly tested for proper calibration and function (FHWA Effective Anti-Icing Program 2004).

Liquid spread unit set in dump truck body (FHWA Effective Anti-Icing Program 2004)

Most important to anti-icing success is the education of mangers and operators. Many concepts of anti-icing are against traditional techniques and etiquette. For this reason, education is vital to break old habits and shape an understanding of anti-icing procedures and concepts. Understanding of the method and how it works will only help mangers and operators to better perform their jobs (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998).

Anti-icing has the capability, if managed correctly, to achieve bare pavement results while demanding fewer snow removal hours, using fewer materials, and reducing negative impacts on the environment. Anti-icing with prewet solid materials requires little to no equipment upgrades while liquid spreaders can be purchased or even built cost effectively through the conversion of other liquid spreading machines (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). Different material storage and distribution units may be necessary.

Anti-icing can become ineffective in the cases of heavy precipitation or when crews were unable to apply anti-icing chemicals before accumulations. If this situation arises, traditional deicing techniques should be employed (U.S. Roads 2004). Losses of liquid anti-icers in windy conditions can also be an issue and care should be exercised when winds are strong or gusty. Solid material and liquid material spreaders alike operate in extreme conditions so should routinely be inspected and cleaned to ensure long reliable service lives. Without proper education and understanding of anti-icing by managers and operators, the program will not be successful.

References

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

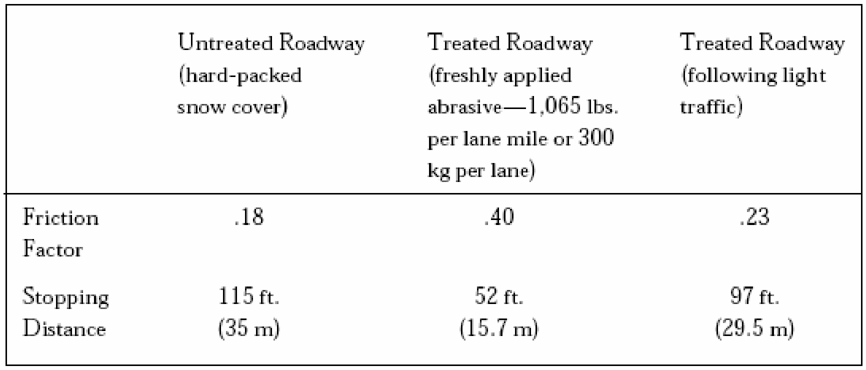
Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

U.S. Roads. 2004. *Using Salt and Sand for Winter Road Maintenance*. http://www.usroads.com/journals/p/rmj/9712/rm971203.htm.

Abrasives

Abrasive use such as sand, slag, cinders, and bottom ash from power plants have long been used to increase surface traction in the winter months. Sand is the most commonly used abrasive. Abrasives are most useful to apply when it is too cold for chemical deicer to be functional or when snow and ice are bonded to the road surface and cannot be removed. Abrasives have also traditionally been applied at dangerous locations such as curves, hills, and intersections, typically at a rate of 500 to 1200 pounds per mile (U.S. Roads 2004). They provide no deicing capabilities of any sort and are only useful for increasing the coefficient of friction between the road surface and vehicle tires (FHWA Effective Anti-Icing Program 2004).

Abrasive quality is important to its effectiveness. Material with angular or crushed particles provide better traction. Small particles are ineffective and more environmentally harmful. Material should ideally be larger then the #50 sieve but smaller then 3/8”. Treating abrasives with 50 to 100 pounds of salt per cubic yard is needed in order to prevent the material from becoming frozen and difficult to work with. Without the salt, abrasives can become difficult for the mechanical spreader to handle. The added salt also helps to imbed the sand into the ice surface, particularly if the material was prewet, making it more effective and less likely to be blown off the road surface (U.S. Roads 2004). Despite being prewet and mixed with deicing chemicals, it has been found that sand is blown off of the highway by as few as 10 to 12 passing vehicles (Technology News 2001). The following chart shows how ineffective salt can be in areas of moderate to heavy traffic.



	Untreated Roadway (hard-packed snow cover)	Treated Roadway (freshly applied abrasive—1,065 lbs. per lane mile or 300 kg per lane)	Treated Roadway (following light traffic)
Friction Factor	.18	.40	.23
Stopping Distance	115 ft. (35 m)	52 ft. (15.7 m)	97 ft. (29.5 m)

Added friction of abrasives in low and high traffic areas (Traffic News 2001)

However, it should be noted that a mix of abrasives and chemicals are no more effective, baring extreme temperatures, then pure chemicals alone. It fact, the presence of abrasives have a negative impact on the effectiveness of the chemicals (FHWA Effective Anti-Icing Program 2004).

Abrasives actually have far more negative impacts then positive ones. In fact, sand has more unfavorable impacts on the environment than road salt (Shi et al. 2004). Sand commonly requires spring cleanup and has been known to clog drainage pipes and inlets (U.S. Roads 2004). Sand particles surround and coat ground cover vegetation, leading to damage. Sand also finds its way into lakes and streams often carrying other roadway pollutants thus contaminating water sources. Sand particles less then 2mm in diameter are the most dangerous as they prevent oxygen movement into streambed gravels thus stressing fish and other aquatic animals (Shi et al. 2004).

The small sand particles on the order of 10 microns in size can become air born and contribute to air pollution (U.S. Roads 2004). This dust problem can lead to respiratory problems and lead to general air quality issues (Shi et al. 2004). The air born dust remains to be a problem after spreading as vehicles drive over the sand until spring clean up can be completed. Overall, abrasives cause many more problems then they solve, but due to their low price and high availability, are still exclusively used in some regions.

References

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

Shi, X., J.M. Staples, O. Stein. 2004. Highway runoff BMPs and winter traction materials management: A cold regions perspective. In review, see *www.coe.montana.edu/wti/wwwshare/Corrosion/Winter_Traction_Materials.pdf*

Technology News. Jan-Feb 2001. Dry sand on winter roads provides little benefit.: 1-2.

U.S. Roads. (2004) *Using Salt and Sand for Winter Road Maintenance*. http://www.usroads.com/journals/p/rmj/9712/rm971203.htm.

Automated-Fixed Deicing Chemical/Hot Water Sprayers

This system consists of a weather-sensing unit that activates a high pressure nozzle and deicing sprayer system mounted along a guardrail or in the roadway itself. (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). This type of system is available for problem areas such as bridge decks, on/off ramps, tunnel entrances, and intersections. The monitoring systems are capable of figuring whether further treatment is necessary in case of dilution or in the case of heavy snowfall, shutting down the system entirely (Ministry of Transportation Ontario 2001). The system provides needed deicing chemicals at the right moment thus increasing public safety and cutting down on excess material usage (Research Readers Digest 1999).

The system is made up of a road weather information system, much like the ones discussed previously. Pavement surface temperatures, subsurface temperatures, moisture, ice, snow, frost conditions, concentration of applied chemicals, and freeze point detection are the data that is necessary for this system. This information is needed for the automated computer arrangement to activate the hydraulic and fixed sprayer system. Spray nozzles, distribution piping, valves, large storage tanks, and the necessary automation must all be properly designed for loadings common to the climate (Ministry of Transportation Ontario 2001). The computer system generally has numerous preset programs to match various weather conditions that may occur. Ideally, the system can be turned on in sections, entirely, or sporadically. The systems can usually also be activated by radio control from trucks or manually from the control area (Research Results Digest 1999).

Nozzle heads can be located on guardrails, in concrete barriers, or recessed in the pavement. Nozzles recessed into the pavement boast up to two lanes of coverage. Nozzles are commonly spaced in the order 40 ft on average depending on gradation of the road surface. Common deicing chemicals compatible with these systems are sodium chloride, calcium chloride, magnesium chloride, magnesium acetate, potassium acetate and mixtures of the previous constituents (Ministry of Transportation 2001). These systems can also be attached to a variable message sign to inform motorists of conditions and deicing efforts. In the case of malfunction or power failure, the system sends a warning so that deicing vehicles can service the area (Research Results Digest 1999).

Example of in roadway sprinkler head (Ministry of Transportation 2001)

Decrease in accidents where systems like this have been installed nears 50% (Research Readers Digest 1999). The precise timing of application that this system provides reduces overuse of chemicals or loss of chemicals from traffic. Furthermore, the optimal timing results in more effective removal of snow and ice from roadways. It also prevents use of chemicals in the event inclement weather does not occur as originally forecasted (Ministry of Transportation Ontario 2001). This system is not effective for use on porous pavements due to refreeze in voids (Research Readers Digest 1999). Nozzle and piping system are also subjected to hostile conditions and are prone to failure. Maintenance costs are generally low for this type of system, barring unforeseen failures. Cost-to-benefit ratios for these systems have been around 2 to 1.

Costs per mile for construction and installation are around 1 million dollars (Research Reader Digest 1999).

Similar systems have been employed in Japan to remove snow and ice from roadways. However, the low pressure, low velocity sprinklers distribute hot water over the road instead of deicing chemicals. The climate-sensing units in theory are identical, except that geothermal hot water or heated well water is drawn and applied. These systems have proven capable of melting snow accumulation, but can severly damage drainage systems (Research Results Digest 1995).

References

Ministry of Transportation Ontario. May 1 2001. *Anti-Icing on Structures Using Fixed Automated Spray Technology (FAST)*. http://www.mto.gov.on.ca/english/engineering/anti_ice/anti_ice.htm.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 3-5.

Research Results Digest. Apr. 1999. Report on the 1998 scanning review of European winter service technology. 238: 5-7.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

U.S. Roads. 2004. *Snow and Ice Control Materials and Methods*. http://www.usroads.com/journals/p/rmj/9712/rm971203.htm.

Section 2-3. **SNOW REMOVAL MACHINERY/METHODS**

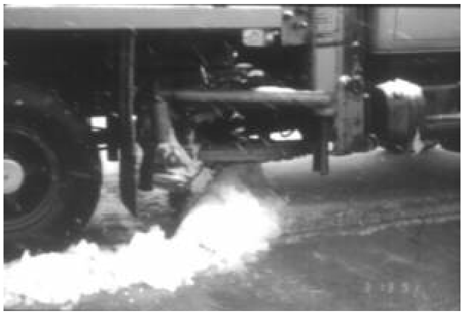
Snowplows

The purpose of snowplowing is to remove as much loose snow and ice from the road surface as possible before applying deicing chemicals. It reduces the amount of melting that the deicing chemical must perform by preventing excessive dilution. In cases of very dry and cold snow, snow will not stick to the roadway, thus making plowing the only necessary measure (FHWA Effective Anti-Icing Program 2004).

There are many kinds of snowplows in use today. They include one-way plows, reversible plows, four-way articulating plows, underbody plows, side wings, and hydraulically extendible plows (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). One way plows can only be raised or lowered by the operator, and the plow is angled so as to leave the snow back (typically) on the right side of the truck. Reversible plows are the most common. They are simply a straight blade that can be angled either direction by the operator. Four-way articulating plows are jointed (articulation point) in the middle of the plow, allowing the operator to angle the plow left, right, in a V shape or in a backwards V shape. This plow is the most versatile as it allows snow to be moved to either side of the vehicle, both sides of the vehicle, or snow can be captured by the plow.

One-Way Snowplow (www.hankstruckpictures.com), Reversible Snow Plow (www.hankstruckpictures.com), and a Four-Way Snowplow (www.curtisplow.com)

Underbody plows are mounted between the front and rear axles. Due to the position on the truck, it requires less down force then conventional plows. The downward pressure-sensing unit on the plow allows it to be optimally positioned vertically to remove snow and ice while not destroying the road surface (Research Result Digest 1995). Middle mount plows are therefore excellent at plowing hard packed snow and ice. The use of this kind of plow leads to a reduction in deicer application (Federal Highway Administration 2004).



Middle Mount Snowplow (Federal Highway Administration 2004)

Side wing plows are mounted on the side of the truck and are used in conjunction with the front mounted plows. The hydraulically actuated plow has automated height adjustments that allow it to be used on the road surface or at elevated levels. Side wing plows are most useful for widening the plowing area or to push back snow banks especially around and above guardrails (Research Results Digest 1995). Hydraulically extendable plows can be broadened from side to side as much a 9ft to 12ft and are best utilized on roads with varying widths (FWHA Effective Anti-Icing Program 2004).

Truck with side mounted wing plow (www.hankstruckpictures.com)

Cutting edges for snowplows are available in a variety of materials depending on application. Materials include synthetic polymers, rubber, steel, and carbide. Generally, synthetic polymer or rubber cutting edges are the best for slush as they scrape the light material from the roadway. In heavier snow, however, steel cutting edges have been found to work better as the rubber edges fold back. For ice and packed snow, carbide has been found to be the best material for cutting edges (FHWA Effective Anti-Icing Program 2004).

Ideally, every truck should have several different plows available to match the conditions of various storms. This would optimize truck use while improving service (Research Results Digest 1995). Care should be taken to plow minimally over areas where deicing chemicals have been applied to prevent removing the deicing chemicals from the roadway. However, in cases where deicing chemicals were applied during a storm event, this is not always avoidable (Roadsalt and Winter Maintenance for British Columbia Municipalities 1998). Proper snowplowing equipment and techniques can reduce the need to apply deicing chemicals thus reducing contamination of surrounding waters.

References

Federal Highway Administration. 2004. *Advanced Cutting Edge Clears More Ice in NH*. http://www.fwha.dot.gov/winter/roadsvr/CS041.htm.

FHWA Effective Anti-Icing Program. 2004. *Manual of Practice for an Effective Anti-Icing Program*. http://www.fhwa.dot.gov/reports/mopeap/mop0296a.htm.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: -3.

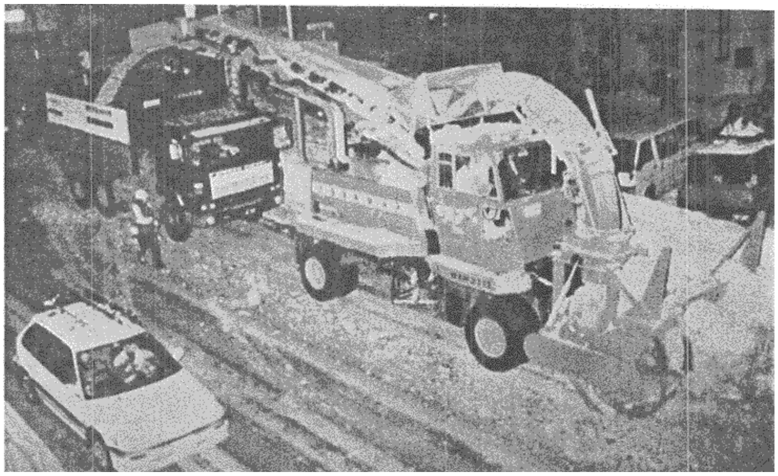
Research Results Digest. Apr. 1999. Report on the 1998 scanning review of european winter service technology. 238:23.

Snow Blowers

Snowblowers, or rotary snow-plows, have been used in snow removal efforts for many years especially in the Western United States. Snowblowers have front mounted units containing an auger and impeller system that blows snow out of a chute high into the air and away from the roadway. Snowblowers are most well known for their effectiveness at removing snow along guardrails or where snow banks have been deposited by snow plows (Tan 2004).

To be effective, snowblowers must operate along the edge of the road or along guardrails with high precision to prevent left over snow from falling back into the roadway. This requires the operator to drive very close to the guardrail using it for a guide. The drawback of this technique is that accidental damage to the guardrail is possible, operating such a large powerful machine in such close proximity to it. However, new automated systems are being installed on snowblowers to improve lateral control of the machine thus reducing contact with guardrails (AHMCT 2004).

A rearward conveying system has also been invented for attachment to snowblowers. This allows snow collected by the snowblower to be placed into a trailing truck for offsite hauling. The snowblower and tag behind truck can clear one lane of traffic while moving forward at a rate of 5 to 10 mph.



Snowblower with tag along truck (Research Results Digest 1995)

The two-man crew (snowblower operator and truck driver) must work together although articulation between the front and rear axles make this operation quite manageable (Research Results Digest 1995).

Snowblowers are especially adept to removing large drifts of snow or snow banks from a site. This ability leads to less need to apply chemicals to clear the roadway. Snow banks are also commonly laden with salt, and their removal to an offsite location would ensure that spring runoff into surrounding bodies of water is reduced. The disadvantage of this kind of machinery is the need to have trained operators on staff. Mobilization of snowblowers to a site can be quite an undertaking. Lastly, given the inconsistency in severity of New York winters, their may be years when winter precipitation is not heavy enough to warrant a snowblower.

References

Advanced Highway Maintenance and Construction Technology. 2004.UC Davis Caltrans. http://www.ahmct.ucdavis.edu/roadview/r_res4.htm..

Tan, H.S. 2004. An automated snowblower for highway winter operations. *California Path,* 10:1-6.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 2.

Hauling Snow

Plowing operations during significant snow accumulations can leave large roadside snow banks. These snow banks can obstruct the motorists view, infringe on the edges of the roadway, cause drifts to form in roads, and make later plowing difficult. These roadside snow banks are often contaminated with roadsalt, abrasives or other deicing chemicals. Furthermore, oil, grease, heavy metals, litter, and dust are all commonly found in the banks. For this reason, collecting this contaminated snow and hauling to a prepared site is often the safest and most environmentally friendly option (Transportation Association of Canada 2003).

Traditional loader and truck units can be used for snow collection and hauling. Snow blowers have also proved to be efficient for this use. By relieving the site of snow banks and piles, a reduction in pollutants such as road salts reaching watercourses is acheived. In urban areas, hauled snow is often taken to the municipal wastewater facility. In more remote areas, sites should be chosen and prepared to minimize negative environmental impacts (Research Results Digest 1995).

Short haul distances are ideal for minimizing equipment and road wear while reducing time and cost of completion. Wetlands, flood prone areas, and contaminated sites should not be chosen for snow storage, and the use of solid waste sites should be closely examined. Sites with overhead or underground utilities should also be avoided for safety and logistic reasons (Transportation Association of Canada 2003). Ideally, an impervious surface that has drainage with some form of contaminate filtration would be preferred.

Removing the contaminated snow from the roadside does have positive environmental impacts, especially in the case of a site with nearby surface water. However, a great deal of equipment and labor is required to facilitate the removal on an as needed basis. Equipment in the road will lead to traffic delays and safety issues. While contaminated snow can be hauled offsite, it still can have negative impacts at the dump-site, and thus utmost care should be taken in the sites selection and care. Spring cleanup of all haul sites is mandatory.

Using a snow blower to load trucks of hauling contaminated snow banks
(http://www.ogra.org/lib/db2file.asp?fileid=9198)

References

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 2-3.

Transportation Association of Canada. Sept. 2003. Snow storage and disposal. 1-2, 4.

Snow Melting Machinery

Melting snow rather then hauling it to an offsite location is another option in winter event management. Both alternatives are effective for removing contaminated snow from an environmentally sensitive site. Snowmelters come in many shapes and sizes. The trailer-mounted units are the most common but self propelled machines also exist. Snowmelters offer a fast, safe, reliable, and cost effective way to avoid the over handling of snow. Major airports and now even cities and municipalities have been exploring the capabilities of snowmelters (Snowmelter 2005).

The trailer style snowmelters are operated in conjunction with loaders that perform similar functions as in snow hauling. Instead of loading dump trucks for offsite hauling, loaders feed the snowmelter. The enclosed water tank is powered by turbo diesel motors that run a combustion air fan which, in conjunction with a burner, heats the water. The hot water and hot water spray then melt snow and ice placed in the snowmelter (Haughton 2005). Melt water can be discharged into a sanitary or storm sewer system or perhaps into detention ponds. The melt water is screened to separate out all floating solids, and additional filters are optional. One ton of snow generates approximately 240 gallons of water and one ton of snow with these systems requires 1.5 gallons of diesel depending on conditions (Snowmelter 2005).

Trailer type snow melting machine in action (Snowmelter 2005)

Self propelled snowmelters collect and melt snow by themselves, making the need for additional equipment unnecessary. A submerged combustion technique is used on these machines to melt the snow. A high velocity, high heat release burner is fired directly into a full pool of water, creating a warm water bath that instantly melts snow and ice. As with the trailer-mounted system, melt water is discharged into storm, sanitary or other water retention system (City of Toronto 2005).

Self Propelled, self feeding snowmelter (City of Toronto 2005)

Snowmelters allow for great levels of flexibility as they can be operated at any time of day in any condition. Rated capacities depend on machine size, but around 135 tons/hr is an average output. These machines require minimal staff to operate, in comparison to hauling away contaminated snow. The city of Barrie in Ontario, Canada, noted savings of 7 to 8 times when using snowmelters when compared to traditional snow removal techniques (Haughton 2005). Snowmelters are better suited for urban areas, especially parking lots as they can require a lot of space to operate. Also, discharge water is best suited for discharge into a town's waste water system, which may not be an option in more rural areas.

References

City of Toronto: Transportation. 2005. *Snow-The Toronto Snow Melting Machine*. http://www.city.toronto.on.ca/transportation/snow/torontomelt.htm.

Haughton, Fred. 2005. *Getting Rid of the Stuff, Snow Disposal Versus Snow Melting*. http://www.ogra.org/lib/db2file.asp?fileid=9198.

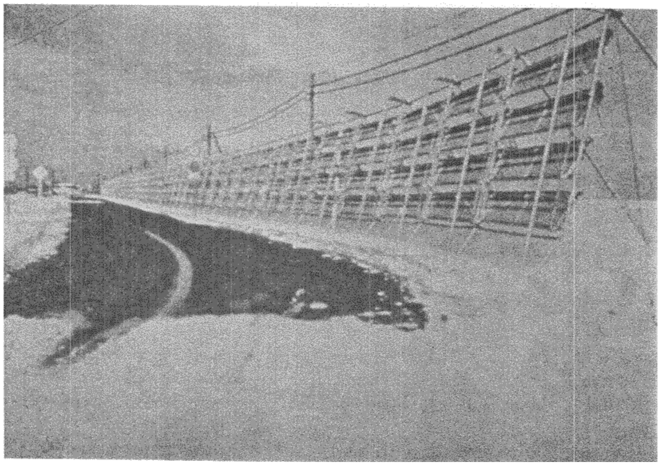
Snowmelter. 2005. *TRECAN Combustion Limited*. http://www.snowmelter.com/faq.html.

Snow Fences

Snow fences have been in use for many years to prevent snow from blowing across and drifting upon roadways. Snow fences are generally designed to effectively store snow on the side of prevailing winds, leaving the leeward or roadside free from significant deposition. This kind of snow fence is commonly of the picket variety and works by decelerating the airflow on the leeward side (Research Results Digest 1995).

Snow fences have been cited to minimize costs of snow removal, reduce the formation of snow on the roadway, and reduce the need to apply deicing chemicals (EPA 1999). In addition, increased visibility has made for a safer working environment for removal crews and has also reduced wear and tear on removal equipment (Federal Highway Administration 2004).

Picket style snow fences tend to become full, leading to losses of effectiveness (Research Results Digest 1995). The fences can even buckle under lateral snow loads (EPA 1999).



Blower Type Snow Fence (Research Results Digest 1995)

A blower style fence has been developed that is designed to accelerate the airflow around the roadway to reduce snow accumulation. Forceful wind action keeps the leeward side of the fence snow free and enhances visibility. The fence can also be placed close to the roadside, taking up very little right of way, thus making it useful on space-constrained sites (Research Result Digest 1995).

Snow fences offer a cost effective solution to using deicing chemicals and mechanical removal techniques. It was been discovered that trapping snow in fences is approximately 100 times more cost effective then mechanical removal (EPA 1999). Disadvantages of snow fences are their lack of ascetic qualities. Some deicing chemical is still needed to prevent ice formation from freeze thaw cycles.

A snowbreak forest is a natural “snow fence”. A snowbreak forest is simply a group of trees intentionally planted or naturally occurring that decreases wind speed and therefore snow accumulations on roadways. The line of trees causes wind blown snow to be deposited in the forest rather then in the roadway. Snowbreak forests are also capable of increasing visibility, air purification, noise reduction, fog prevention, conservation of vegetation, and adding aesthetic beauty to the roadside. The drawbacks of snowbreak forests include the amount of land needed adjacent to the roadway and the challenges of sustaining tree growth in extreme conditions (Highway Snowstorm Countermeasure Manual 1990).

References

Federal Highway Administration. 2004. *Making Snow-Covered Roads Easier to Open*. http://www.fwha.dot.gov/winter/roadsvr/CS009.htm.

Highway Snowstorm Countermeasure Manual. March 1990. *Snowbreak Forest Book*: 2-3, 5-7.

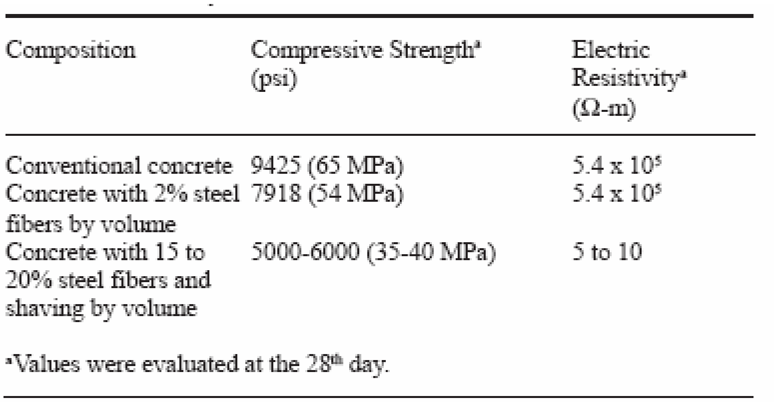
Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204: 5,7.

United States Environmental Protection Agency. Sept. 1999. *Storm Water Management Fact Sheet Minimizing Effects from Highway Deicing*: 4-5.

Section 2-4. **CHEMICAL-FREE METHODS**

Electrically Conductive Concrete Mix Overlays

This solution uses concrete to bind together electrically conductive materials which, in formation with electrodes and a power source, is capable of heating the roadway for deicing or anti-icing applications. There are two main types of conductive concrete: concrete that incorporates conductive fibers and shavings, and concrete that uses conductive aggregate. Conductive fibers and shavings have been found to foster higher mechanical strengths but a lower conductivity then conductive aggregate. The lower conductivity is due to the small fiber to fiber contact areas. The reason for low mechanical strength in conductive aggregate based overlays is the high water to cement ratio required to compensate for the water absorbed by the conductive aggregates (Yehia and Tuan 1999a).

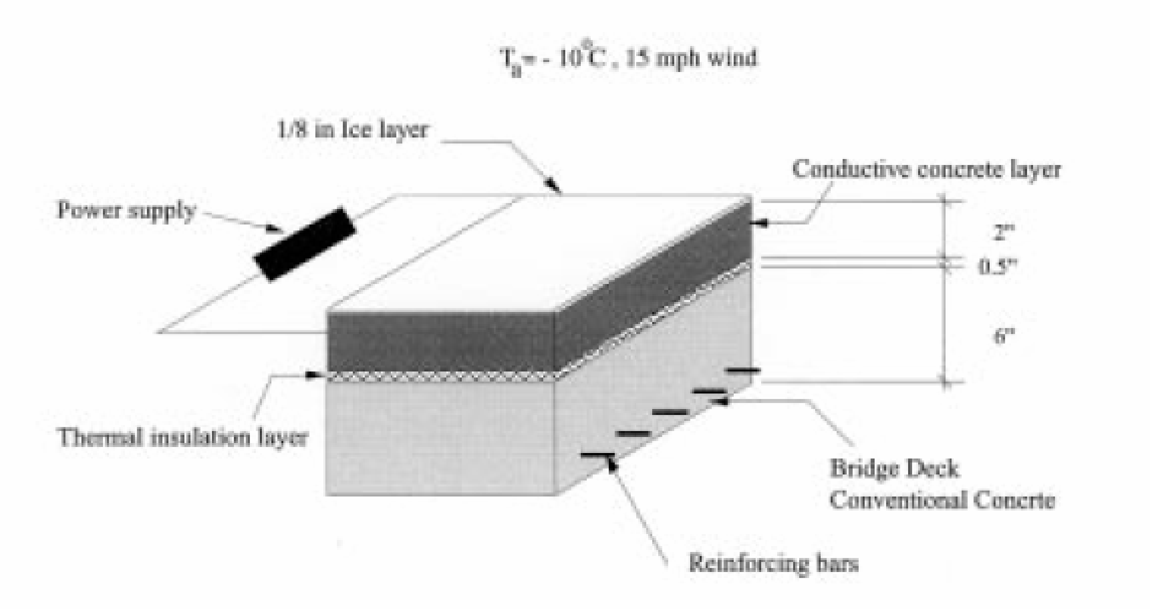


Composition	Compressive Strength[a] (psi)	Electric Resistivity[a] (Ω-m)
Conventional concrete	9425 (65 MPa)	5.4 x 10^5
Concrete with 2% steel fibers by volume	7918 (54 MPa)	5.4 x 10^5
Concrete with 15 to 20% steel fibers and shaving by volume	5000-6000 (35-40 MPa)	5 to 10

[a]Values were evaluated at the 28th day.

Comparison Conventional Concrete and Conductive Concrete (Yehia and Tuan 1999a)

Materials used in fiber and shaving based conductive concrete include steel fibers, steel shavings, carbon fiber, graphite, and other manufactured waste products. Conductive aggregate is generally a form of carbon black or coke breeze. Although fiber based conductive concrete is much more common due to its high mechanical strength, strides have been made to produce a more durable conductive aggregate mix (Tuan 2004a). Due to electrical resistance and impedance, enough heat can be generated in the concrete overlay to prevent ice formation when connected to a suitable power source. The key to successful conductive concrete no matter the type is that it operates at the electrical threshold for the circuit while maintaining mechanical strength and also proper mixing conditions. As electric current flows through conductive concrete, the heating rate increases as does the temperature of the overlay. Similarly, the conductivity of the element also increases allowing more current to flow though the specimen at a constant voltage. Applied voltage load must be controlled to maintain gradual heating so as to prevent thermal shock to the concrete (Emerging Construction Technologies 2004).



Conductive Concrete Cross Section (Yehia and Tuan 1999a)

The mix design for such overlays includes adding conductive concrete particles and fibers to conventional aggregate. Approximately 20% by volume of conductive shavings and 1.5% conductive fibers by volume of concrete have been found to be the optimal ratio. It has been found that the use of just fibers or just shavings alone does not provide a resistivity low enough for deicing. Any ratio of conductive materials greater then 20% and 1.5% have been found to encompass unacceptable workability and finish ability qualities. It is still required that the concrete overlay meet all AASSHTO requirements of compressive strength, flexural strength, modulus of elasticity, resistance to freezing and thawing, shrinkage, and permeability (Yehia and Tuan 2002b).

There are two popular means to prepare conductive concrete which are simply through conventional mixing practices or through slurry infiltration. Conventional mixing yields higher resistance and a higher compressive strength in the overlay while the slurry infiltration method usually results in increased flexural and compressive strengths while the resistance is lower.

Conductive concrete is typically a lightweight mix design but can be used structurally as it bonds well with conventional concrete and has excellent stability as well as load bearing capacity (Emerging Construction Technologies 2004).

Typical conductive concrete overlays that have been constructed, mostly for testing purposes, have a general thickness of 2.5" to 3.5". Electrodes are simply perforated thin steel plates located along the length of the slab that allow the concrete to flow through to ensure quality bondage. Maximum aggregate size of most conductive overlays researched is ½" as this size offers quality mechanical strength and also the flow ability in addition to desired workability (Yehia and Tuan 2002b). When connected to a power source, the power enters the overlay via the electrodes and the overlay acts as a resister in which the heat loss due to the impedance of the concrete is capable of deicing or anti-icing applications. Direct Current (DC) is the simplest power source. A safety note, the voltage should be kept under 48 Volts as that is the safe threshold for human beings (Yehia and Tuan 1999a).

Electrode Arrangement (Tuan 2004a)

In order to have a successful conductive concrete overlay, a collection of properly calibrated devices are necessary. First, a temperature sensing unit is needed in order to identify when the temperature of the overlay drops below the inactive temperature or rises above the operating temperature, which are commonly set at approximately 40^{o}F and 55^{o}F respectively. Next, a power switching unit is necessary to activate or deactivate the overlay based on the signal sent from the temperature unit. Furthermore, a current sensing monitor and an operator interface unit are needed to complete the system, to prevent unnecessary failure and allow manual operation and calibration if needed (Tuan 2004a).

Due to the uniformity of the conductive concrete as a heating material, any breech in the slab will not affect the heating capabilities of this system. As with a coil or loop system, a failure of any part of the loop could lead to failure of the entire circuit. This issue does not arise, because the concrete doubles as the road surface overlay and the heat source for deicing.

Conductive concrete is a system that can be used for anti-icing or deicing. In anti-icing, the conductive overlay is powered several hours before the storm event is due. This allows the overlay to obtain a constant temperature at which the snow and ice will turn to water upon contact and drain off the roadway or be evaporated. The target temperature is often just above freezing, but depending on wind conditions may need to be higher. In deicing, the overlay is not powered until the road surface is already covered with snow or ice. This tests the ability of the conductive concrete to remove snow and ice in bulk quantities. Anti-icing, in addition to requiring less power, is the more desired applications for roadway safety (Tuan 2004b). The goal of winter road maintenance is to keep the roadway clear to dry pavement and anti-icing better attains this goal.

A properly operating conductive concrete overlay is capable of keeping a road subject to snow and slush during a winter event clear, thus eliminating the need to plow and spread salt on the road. In a worse case, high snow rate scenario in which the conductive concrete is having difficulty keeping up, a drop of the snow plow to remove the slush is all that is needed to help the system catch up. This is desirable because it prevents salt placed on the road to increase traction from draining into the surrounding environment, especially the natural water supply. In addition to the environmental advantages of not using salt, conductive concrete is not corrosive to structural steel, maintenance equipment, or automobiles. It also reduces or nearly eliminates snow removal costs of labor and materials. Conductive concrete is also capable of reducing a wet surface to dry pavement thus increasing traction, and therefore safety.

Conductive concrete has excellent mechanical and electrical conductivity properties making it both an effective roadway clearer and a sound structural road component. It is produced easily as no special concrete equipment is necessary. Usually a thermal insulation layer to prevent heat losses into the ground is laid upon the base material and the conductive concrete is applied over the insulator (Zuofo et al. 2003). No extra space requirements or heavy construction techniques are required to install conductive concrete. Care must be taken however in the assembly of the electrodes and electric circuitry to ensure a reliable long lasting system. Power can be taken offline using alternating current and a 120 volt plug or powered by direct current, both of which are capable, reliable, and simple. However, a constant power source is necessary for conductive concrete to function properly, thus potentially creating a high energy-consumption bill.

References

Emerging Construction Technologies. 2004. *Conductive Concrete*. http://www.new-technologies.org/ECT/Civil/conductive.htm.

Tuan, C. 2004a. Conductive concrete. *Concrete Technology Today* 25:1-2.

Tuan, C. 2004b. Electrical resistance heating of conductive concrete containing steel fibers and shavings. *ACI Materials Journal* 101:65-71.

Yehia, S, C. Tuan. 1999a. Conductive concrete for bridge deck deicing. *ACI Materials Journal* 96:382-391.

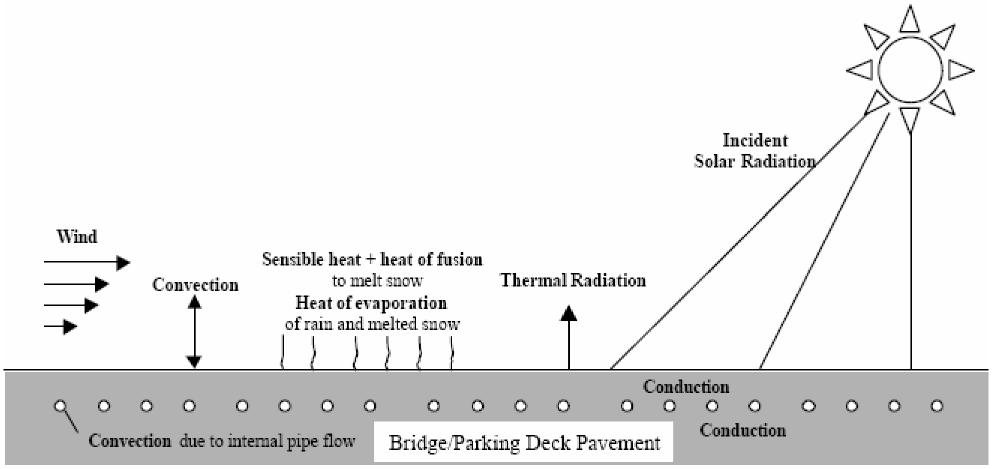
Yehia, S, C. Tuan. 2002b. Conductive concrete overlays. *Concrete International*: 56-60.

Zuofu, H., L. Zhuoqiu, T. Zuquan. 2003. Finite element analysis of electrically conductive concrete for roadway deicing or snow-melting system. *ACI Materials Journal* 100:469-476.

Geothermal Heat Pumps

Geothermal heat pumps are an effective means of using a heated road bed to clear snow and ice from the roadway. There are two main geothermal systems available. Direct use systems utilize the heat in the ground water directly for energy. Low to moderate water temperatures (100^{o}F to 300^{o}F) are required to supply direct heat to applications. The other type of system, ground-source heat pump, utilizes the heat in the ground water during winter months and stores heat in the ground water during the summer months (Geothermal Heat Center 2004b).

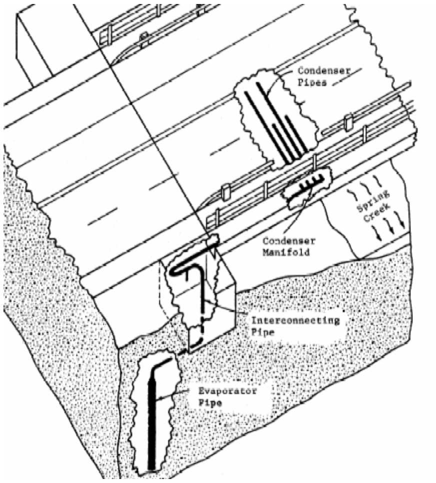
Unfortunately, the majority of the resources necessary for direct use systems in the United States are available only in the west. The primary applications of this geothermal energy have been in space heating, green houses, and aquaculture facilities. A direct use system is made of three parts: a production facility or a well that brings the hot water to the surface; a mechanical system which generally includes a heat exchanger, piping, and controls to transfer the energy to useful applications; and a disposal system where the cooled fluid can be dispensed (Geothermal Technologies Program 2004a).



Geothermal Heated Surface (Spitler & Ramamoorthy 2000)

The ground source heat pump method, however, can be installed anywhere because the system recharges itself, making it self-sufficient. This means the heat pump takes heat out of the ground in the winter to heat the roadway, and then removes heat from the roadway in the summer and recharges the ground. This system includes a geothermal earth connection system, a geothermal heat pump subsystem, and a geothermal heat distribution subsystem (Geothermal Technologies Program 2004c). The loop or series of pipes can be buried either horizontally or vertically, and they either give heat to or take heat from the ground depending on the respective season (Spitler & Ramamoorthy 2000).

In winter, heat is extracted from the earth using a heat exchanger and is upgraded by a heat pump to a pipe network under the roadbed. The heat is carried through an anti-freezing liquid and is capable of heating a roadbed and clearing a highway like conductive concrete. In the summer time, heat is recovered from the hot pavement and stored by connecting heat exchangers directly to heat pipes (WIRE 2004).



Schematic of a bridge heat pipe system (Lund 2001)

The process is most commonly done with glycol solution hot water (Geothermal hot water) or steam (Geothermal steam) circulated in the pipes within or below the pavement surface. There is one documented case where the hot water was actually sprinkled onto the pavement surface itself. Heating requirements for this method depend on four factors: rate of snowfall, relative humidity, wind velocity, and air temperature. It is crucial to have accurate weather data on these variables in order to properly design a system capable of first melting the snow and then evaporating the resulting water film all while operating under specifications (Lund 2001).

In terms of roadway surface to be used with this system, both Portland cement concrete and asphalt concrete can be used. However, Portland cement concrete has better then normal conductivity when compared to asphalt concrete. The concrete overlay slab can be poured above the heat pipes meaning common design practices and repairs can be utilized. Furthermore, asphalt concrete is less preferred due to its high laying temperatures and need for compaction. The temperature of asphalt mix can be over 300^{o}F at time of paving and has been known to deform or break heat pipes and connections (Lund 2001).

The benefits of geothermal heat pumps are their ability to remove snow and resulting water from the roadway during temperatures below the freezing point. This provides greater safety for the public, and also reduces costs associated with snow plowing and road salting. Low running costs and a simple mechanical system make it both cost effective and reliable. The negative environmental impacts of road salt are taken out of the equation entirely. Only normal ground temperatures are necessary for this system, which is capable of recharging itself in the summer months for winter applications. Winter heating loads and summer recharge rates need to be intimately studied and understood to properly design the system. The tremendous amount of excavation necessary to place heat pipes coupled with the shear number of units make this possibility on a long site impractical (Geothermal Technologies Program 2004b). Pipe corrosion has also been cited as a problem, and any such problem with this kind of system would require removing the road surface to service the system, which would be costly and impractical.

References

Geo-Heat Center. 2004. *What is Geothermal?* http://geoheat.oit.edu/whatgeo.htm.

Geothermal Technologies Program. 2004a. *Direct Use of Geothermal Energy*. http://www.eere.energy.gov/geothermal/directuse.html.

Geothermal Technologies Program. 2004b. *Economic Impacts of Using Geothermal Energy*. http://www.eere.energy.gov/geothermal/econ_impacts.html.

Geothermal Technologies Program. 2004c. *Geothermal Heat Pumps*. http://www.eere.energy.gov/geothermal/heatpumps.html.

Lund, J.W. *Pavement Snow Melting*. 2001. Klamath Falls: Geo-Heat Center, Oregon Institute of Technology.

Spitler, J.D., M. Ramamoorthy. 2000. *Bridge Deck Deicing using Geothermal Heat Pumps*. Proceedings of the Fourth International Heat Pumps in Cold Climates Conference, Aylmer, Québec. August 17-18, 2000.

WIRE. 2004. *Melting Snow on Roads Using Geothermal and Solar Heat*. http://www.wire0.ises.org/wire/exchanges/Projects.nsf/0/do64dc41a5070793c12567a1003adab6?.

Covered Highway

Poor road conditions occur because precipitation from cold weather events is deposited onto the road surface. Snow, ice, sleet, and rain all fall freely onto road and immense efforts are put forth to maintain safety no matter the weather conditions. An enclosed highway would keep the elements off the road increasing safety and decreasing maintenance efforts. No matter the weather conditions outside, bare, dry, safe pavement conditions and normal driving speeds could be maintained inside the enclosed sections.

A covered highway is an option to prevent the need to use deicing chemicals. If the road is kept free of snow and ice, routine winter maintenance would become unnecessary. An enclosed highway would even reduce storm water runoff during summer time rain events. Live loads such as snow and rain would have to be intimately studied as the enclosed structure would be used as a temporary storage area for these materials. The construction materials would most likely include structural steel in some capacity. The structure would have to be routinely inspected to insure that corrosion from deicing chemicals used on adjacent parts of the roadway would not have a detrimental effect on the structure.

Drawbacks of an enclosed highway include the loss of aesthetics of traveling in a tunnel like structure. Especially on space restricted sites, accidents or problems inside the enclosed structure could have the potential to lead to closures or long traffic delays. Transition zones, where motorists drive from snow covered sections to dry pavement of vice versa are often the most dangerous sections of highway. Transition zones entering and leaving these enclosed structures could be dramatic, and measures should be taken to provide ample winter maintenance on the sections adjacent to the enclosed highway. Motorists should also be warned via variable message signs of the transition zones when they do in fact exist. If the highway were completely enclosed, ventilation systems would be necessary significantly adding to the cost of this solution. Structurally, lengthy enclosed structures would be expensive to construct.

Section 2-5. **STRUCTURAL SOLUTIONS/DRAINAGE**

Drainage System/ Runoff Control

Controlling the runoff resulting from deicing operations is of utmost importance to maintaining unpolluted watercourses. Runoff enters the environment through three means. The first way is by overland drainage or discharge of drainage systems into surface water. The second is by infiltration into the ground with the possibility of reaching the groundwater. The third way is by spray of runoff caused by wind or vehicular traffic (Transportation Association of Canada 2003).

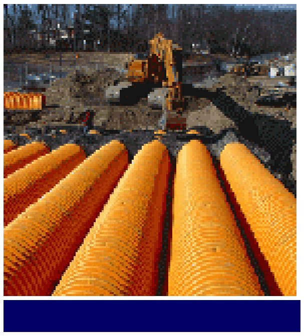
A closed drainage system or storm sewer system is a common structural design that is capable of alleviating the chloride content in the surrounding waters. In this system, runoff is directed away from the natural surface water with negligible infiltration potential. It has been found that a storm sewer system can reduce the impact of salt laden runoff on ground water, wetlands, and wildlife (Transportation Association of Canada 2003). Additional elements that can be used in conjunction with this kind of system include roadside ditches, sheet flow, infiltration trenches and basins, detention and retention basins, vegetated swales, contaminate traps, and various filters (Western Transportation Institute 2004).

Sheet flow is the most common means of controlling runoff, for example when water is dispersed off of the roadway across grass strips. This technique requires ample space adjacent to the roadway, and has been found to have negative impacts on ground water. A V-ditch is a common roadside gulley designed to channel the runoff to another location. Ditches lined with impervious materials such as geosynthetics or clay prevent contamination of groundwater. Flat bottom ditches also channel runoff, but typically at lower velocities leading to the possibility of sediment trapping of infiltration. Some flat bottom ditches are designed to store runoff making sediment trapping and infiltration inevitable (Transportation Association of Canada 2003). All of these options require plentiful non-rugged land area surrounding the roadway.

Detention basins or dry ponds are designed to store runoff for a preset period of time to allow pollutants to separate or settle from the water. They are not designed to always have water in them, but infiltration in these basins is still possible (California Stormwater Quality Association 2003). Retention basins or wet ponds, however, are meant to have water in them all the time, leading to a greater chance of infiltration. Both systems work on the concept of trapping runoff and releasing at slow rates to allow pollutants to settle, evaporate, infiltrate, or become absorbed

(FHWA Environmental Brief 2004). Detention and retention systems are simple to design and build but are somewhat ineffective at removing soluble pollutants (California Stormwater Quality Association 2003). Ample land area is also necessary for these options to be feasible. As with any structural drainage component, freezing can become an issue in the winter months, greatly reducing effectiveness.

A closed drainage system is entirely self-contained, meaning that no contaminants in the water should escape the system. Catch basins and drainage swales direct the roadway runoff into detention basins where the contaminated water can due less harm. The most effective closed drainage systems also have an infiltration trench beyond the shoulder of the road to ensure that the chloride is removed from the runoff. The runoff, unless in the case of a storm of severe conditions, never reaches the natural water bodies. These types of systems work by trapping runoff for a long time until contaminants can be separated through evaporation, settling, or filtering (FHWA Environmental Brief 2004).



Stormtech Chambers being installed (Stormtech 2004)

When space is limited, however, underground water storage may be the most practical solution. Underground water storage tanks are manufactured by several companies including Stormtech™ and are capable of serving the same functions as a detention basin while not wasting any surface space. These runoff storage tanks would be placed under the roadway, as they are a capable road foundation structure (Stormtech 2004).

This solution takes a different approach to the problem of winter event management. Rather then trying to reduce or eliminate the levels of harmful deicing chemicals used, this system attempts to control the chemicals and prevent them from reaching adjacent water courses. This method is costly to implement as large amounts of reconstruction are necessary to facilitate the components of this system. However, once in place, minimal maintenance and upkeep is all that this method requires. When used in conjunction with other methods of reducing deicer usage, it can be quite effective.

References

California Stormwater Quality Association. Jan. 2003. *Extended Detention Basin*: 1-2.

FHWA Environmental Brief. 2004. *Is Highway Runoff A Serious Problem*? http://www.saltinstitute.org/runoff.html.

Roadsalt and Winter Maintenance for British Columbia Municipalities. December 1998. *Best Management Practices to Protect Water Quality*. http://wlapwww.gov.bc.ca/wat/wq/bmps/roadsalt.html.

Stormtech. 2004. *Wastewater Management-Stormtech*. http://www.stormtech.com/product.htm.

Transportation Association of Canada. 2003a. *Drainage and Stormwater Management*: 1-4.

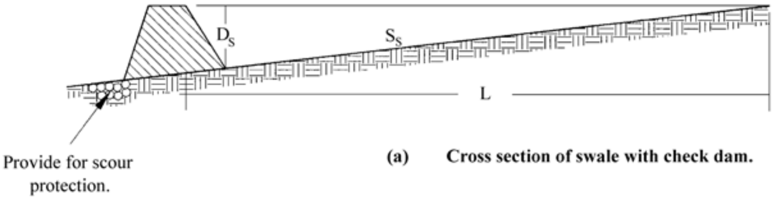
Western Transportation Institute 2004. *Recommendation for Winter Traction Materials Management on Roadways Adjacent to Bodies of Water*. http://www.coe.montana.edu/wti/wti/display.php?id=120

Vegetated Swales

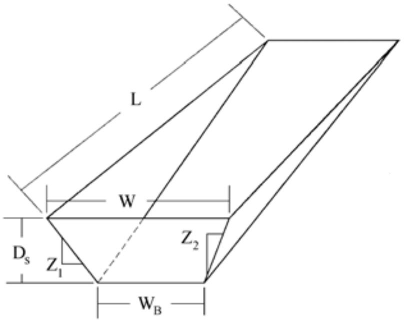
Vegetated swales are shallow channels covered in plant life designed to disperse and filter runoff. Vegetated swales are generally open trapezoidal or parabolic shaped channels used in conjunction with or replacements for traditional drainage systems (Managing Stormwater 2004). Filtering is performed by trapping the pollutants in the vegetation in the swale, through the filtering capabilities of the soils in the swale, or by infiltration into the soil (California Stormwater Quality Association 2003). Vegetated swales can be used where space permits and the soils supports dense ground cover. Success of a vegetated swale hinges on the area, slope, and volume of runoff the swale is subjected (EPA 1999).

Vegetated swales are capable of reducing peak flows, removing pollutants, and promoting infiltration (EPA 1999). They can be aesthetically pleasing, and can be natural or manmade. In New York State, salt tolerant plants should be used in vegetated swales. This would ensure long service life and proper filtering of the swale. To prevent salt spray from vehicles, the wind, or removal equipment from reaching water courses, salt tolerant plants or shrubs should be tightly planted as a barrier in or along the vegetated swale. Common salt tolerant species include Silverberry, Sea Buckthorn, Common Ninebark, Choke Cherry, Staghorn Sumac, Snowberry, and Japanese Tree Lilac (Transportation Association of Canada 2003).

Vegetated swales are impractical in extremely flat sites or sites with rugged topography. Land must be available for them, and the soils must be able to support heavy vegetation. High runoff flow rates can erode vegetated swales, a problem that can be amplified by poorly draining soils. In many areas, vegetated swales are restricted by law, as curb and gutter systems are mandatory. Failures of vegetated swales are most often attributed to soil compaction of sub-base, steep slopes, and short vegetation height (California Stormwater Quality Association 2003). Infiltration through the vegetation may cause pollutants to reach ground water (EPA 1999).



(a) Cross section of swale with check dam.



(b) Dimensional section of swale impoundment area

Vegetated Swale Cross Section (EPA 1999)

A design height of the grass is recommended to be around 6 inches and swales should never be less then 100 feet in length (California Stormwater Quality Association 2003). The parabolic or trapezoidal channel should not have slopes steeper then 1:3 in order to maximize the wetted channel perimeter of the swale. Commonly, vegetation in a swale is fine, dense, grass that is water resistant. Common plants used in swales include canary grass, grass-legume mixtures, and red fescue (EPA 1999).

Maintenance of vegetated swales is important to keep them in proper working order. Inspection for erosion, removal of waste and sediments, and mowing should generally be conducted several times a year (Managing Stormwater 2004). The service life of a vegetated swale is related to the quality of upkeep it receives. If well kept, the service life could be forever. Construction costs generally range from $8.50 to $50.00 per linear foot depending on dimension and conditions. Annual maintenance costs generally hover around $0.50 per linear foot depending on size and shape (EPA 1999).

References

California Stormwater Quality Association. Jan. 2003. *Vegetated Swale*: 1-2, 6-7.

Managing Stormwater. 2004. *Vegetated Swales*. http://www.greenworks.tv/stormwater/vegetatedswales.htm

Transportation Association of Canada. Sept. 2003. *Vegetation Management*: 1-5.

United States Environmental Protection Agency. Sept. 1999. *Storm Water Management Fact Sheet: Vegetated Swales*: 1-5.

Runoff Filters

Filtering systems for the salt laden runoff are another possible solution. These kinds of systems are particularly useful on space limited sites. Filtering systems have no soil restrictions like infiltration technologies. Filtering systems can also be more environmentally sound because they do not depend on the inherent filtration abilities of soil, thus reducing the risk of groundwater contamination (FHWA Environmental Brief 2004).

There are several kinds of runoff filtration systems, including traditional style filters. These filters are all similar in general design as they have a sedimentation area to retain large particles and a secondary filter area that removes the remaining pollutants (FHWA Environmental Brief 2004). Sand filters are one such filter that are emerging for this kind of application. A typical sand filter contains two to three chambers, the first for removing floating sediments, the second for the removal of additional pollutants by filtering the water through a sand bed, and the third chamber for discharge. The discharge is either into a closed drainage system or into surface waters depending on the design and abilities of the system (United States Environmental Protection Agency 1999).

Sand filters are capable of removing high quantities of roadway pollutants such as roadsalt. Most of these systems are below grade, meaning excavation and extensive pipe fitting would be necessary in the case of new construction or in the cast of retrofitting. Costs of installation of these systems is anywhere from $7,000 to $20,000 per acre depending on the system and the site conditions. The system does require routine maintenance to prevent clogging and to replace filters, which generally last 3 to 5 years. Sand filters and other types of runoff filters are still rather unproven in northern climates, however. It is not yet known the level of performance or associated problems that would result in the use of the systems in freezing temperatures (United States Protection Agency 1999).

Another kind of filter they may prove to be useful for removing pollutants and roadsalt from the runoff is woodchip filters. Many of the contaminants in runoff are attracted to organic material. Filters made from woodchips trapped in a wire mesh and dropped in drainage ditches have been found to remove nearly 97% of pollutants. The filters are non-toxic, cheap, readily available, and have high public acceptance. Constant replacement is an issue, however, as filters must be replaced on the order of every 30 to 60 days for best performance. As with sand filters, these filters still require further testing to determine exact levels of effectiveness (Edie News 2004).

References

Edie News. 2004. *Woodchips Remove Toxins From Roadway Runoff.* http://www.edie.net/news/archive/4522.cfm.

FHWA Environmental Brief. 2004. *Is Highway Runoff A Serious Problem*? http://www.saltinstitute.org/runoff.html.

United States Environmental Protection Agency. Sept. 1999. *Sand Filters*: 1-3.

Section 2-6. **INCREASE TRACTION**

Increase Skid Resistance

Increasing skid resistance of the roadway is another way to reduce salt usage while increasing surface traction for vehicles. These methods can be applied in conjunction with any of the three previous solutions to further enhance the quality of the final product. The use of soft densely graded aggregates that easily polish can make a wet road surface dangerously slippery. Friction between vehicle tires and a wet road surface depend on the hardness of aggregate minerals and their micro-textural roughness (Better Roads 2003). Whether a flexible pavement (bituminous pavement) or rigid pavement (Portland cement concrete pavement) is used, skid resistance can be enhanced through proper mix design, the use of quality aggregates, and appropriate installation techniques (U.S. Roads 2004).

The micro-texture of the pavement is important to surface friction, while the macro-texture of the pavement is important for removing water from the road surface. In terms of minerals, the harder the material, the better. Anything with a Mohs hardness number of 6 or greater is ideal, with feldspar and quartz being commonly used. For the aggregate, a sound micro-structure is also important, with sandstone and slag being the ideal candidates for use. Furthermore, flat, jagged aggregates are preferred to round or smooth surfaced material to increase friction. High frictional durability and polish resistance are the solution to skid problems (Better Roads 2003).

For flexible pavements, the mix design should only be used after a study has been conducted to ensure that there will be sufficient stability and skid resistance durability. A bituminous pavement should contain non-polishing aggregates. The mix design should allow for good surface exposure to coarse aggregates. Whether open or densely graded, the asphalt friction course should be a mix concept that has a large quantity of one size aggregate. This provides a uniformly coarse texture at the road surface, thus greatly increasing skid resistance (U.S. Department of Transportation 1980).

For rigid pavements, it is important to make sure that finishing procedures, mix designs, and aggregates are all suitable to provide satisfactory skid resistance and surface durability. The quality of skid resistance depends greatly on the fine aggregates used in rigid pavements. The fine aggregate should be tested to ensure suitable wear and polish resistance capabilities. Research by the Portland Cement Association has shown that the siliceous particle content of the fine aggregate should be greater then 25 percent. As with flexible pavements, coarse aggregates on the surface should have high durability and proper polish resistance capabilities (U.S. Department of Transportation 1980).

There have been varying results concerning the success of open-graded surface courses versus densely graded surfaces. Densely graded surfaces were generally preferred in the past due to their high durability and long service life. However, densely graded surfaces can become dangerously slippery due to the water film that forms on the smooth surface. Open-graded mixes have been preferred by some U.S. states due to the high amount of same sized aggregate, providing a quality coarse structure, and thus exposing a large surface area while water more easily drains through the porous surface. Some states, however complained about sub-par durability in the open-graded mix over time. It was concluded that in states with good material quality control and testing, sound designs, and technologically advanced construction practices, open-graded surface courses could successfully be used to provide proper skid resistance and durability (U.S. Roads 2004).

Skid Resistance Testing (http://www.highwaysmaintenance.com/skidtext.htm)

Grooving of pavements has been studied in an attempt to increase skid resistance and encourage quality drainage. It has been found that the greater the number and depth of grooves in relation to the dimensions of the roadway, the more enhanced the drainage and skid resistance capabilities of the road surface. A fine graded flexible asphalt pavement can be ideal for this application. The main downfall of grooved pavement is the noise pollution generated by vehicle tires passing over the surface (Hayasaka and Takemoto 2004).

The additional costs incurred by this method of increasing roadway friction are mainly spent in research and the formulation of the mix design. The issue of obtaining high quality aggregates could also be an issue depending on geographical availability. It is also recommended that a reputable paving contractor equipped with state of the art automated equipment be hired to ensure a long lasting effective result

References

Better Roads. 2003. *Creating Friction Where Rubber Meets the Road.* http://www.betterroads.com/articles/cot03b.htm.

Hayasaka, Y., H. Takemoto. 2004.*Performance of Grooving Pavement and Optimal Grooving Pattern.* http://www3.ceri.go.jp/english/0207e/020702E.htm.

U.S. Department of Transportation. Dec. 23, 1980. *Skid Accident Reduction Program*: 4

U.S. Roads. 2004a. *Forty Eight State Survey.* http://www.usroads.com/journals/rej/9706/re970603.htm.

Tire Chains/Studded Snow Tires

Deicing chemicals are applied to the roadways to clear snow and ice down to clear pavement. This increases traction for vehicles thus making for safer travel. This possibility has the potential for unsafe conditions for the public. If traction can not be increased by reducing snow and ice with chemicals, perhaps traction should be secured through the individual use of tire chains or studded snow tires. As in the Western United States, roads would be closed to vehicles that do not have tire chains or studded snow tires.

Tire chains are used extensively in the western part of the United States and perhaps should be considered for use in low salt, high snow areas in the East. Traditional steel tire chains are still the most effective, increasing traction by up to 140% while decreasing stopping distances by 20% (Valvoline 2004). When installed on all four tires of a vehicle, accelerating and braking can be twice as responsive as normal tires and steering can also be greatly enhanced (4x4 ABC 2004).

Tire chain diamond style

Tire chain cable style

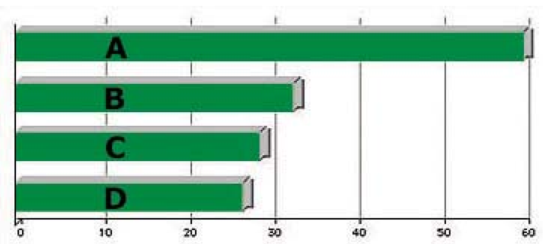
(http://tirechain.com/CarChains.htm)

Rubberized tire chains or cable chains are also commonly used today. They consist of steel aircraft cable surrounded by steel rollers or a mat of rubberized steel cable (Research Results Digest 1995). These chains are much easier to install then their steel counterparts, while causing less damage to tires and the roadway. Rubberized or cable traction chains offer a smoother ride while not sacrificing performance (Valvoline 2004). However, it has been noted that these types of chain options have a shorter service life then traditional steel chains (Research Results Digest 1995).

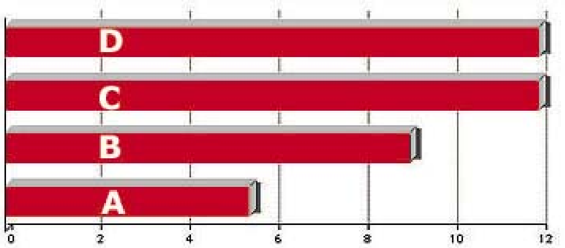
A disadvantage of mandating tire chain use is the fact that motorists must stop to install them before proceeding into the chains-only zone of road. This inconvenience is not appreciated by many of today's motorists. Chains also have the tendency to reduce road service life as the aggressive patterns can tear up pavement in low snow areas or dry pavement. Travel speed with snow chains is reduced to 30mph maximum, which promotes safety, but often frustrates overconfident, behind schedule motorists. Increased air pollution from the increased rolling resistance provided by chains has also been mentioned as an issue. Tire chains put more personal safety responsibility in the hands of the motorists. In order for this to work and safety to be maintained, a shift in driver behavior to a less aggressive mode is necessary.

Studded snow tires are another alternative to using tire chains. They increase surface traction through the use of steel studs, rubber studs, or new "studless" snow tires. Lightweight, road surface friendly studs are being developed. In many areas, studded snow tires are not permitted on freeways, making tire chains a more viable option for mountainous areas (Research Results Digest 1999).

Studded snow tires offer greater handling in winter events, including braking, accelerating, and steering. The use of studded snow tires has led to a reduction of 7,500 accidents annually in Sweden due to inclement weather conditions (Research Results Digest 1999). Unfortunately studded tires have been notoriously harsh on road surfaces, and increased air pollution from their use has also been noted. Drivers in areas of severe winter weather must suffer the inconvenience of buying and changing various tires per season.



Distance Traveled up snow packed hill (15mph)



Stopping distance on packed snow

A=Chains on all four wheels *(All distances in meters)*
B=Chains on one axle
C=Studded Snow tires
D=Snow tires *(4x4 ABC, 2004)*

The use of tire chains and studded snow tires in mountainous areas subject to severe weather should be considered as an option. It would greatly reduced the need to apply deicing chemicals, thus lowing salt content in the runoff that makes its way into the natural water supply. An improved durable pavement would make the additional wear to the highway less problematic. At accidents, police should be given the authority to issue citation for tire violations in this area, to enforce the rules and maintain safety (Research Results Digest 1999). This solution would cause some motorist inconvenience, while making them directly responsible for the safety of themselves and the safety of others on the road.

References

4x4 ABC. 2004. *Tire Chains vs. Snow Tire Comparison*. http://www.4x4abc.com/ML320/ml_chainss.html.

Research Results Digest. Jan. 1995. Winter maintenance technology and practices-learning from abroad. 204:7.

Research Results Digest. Apr. 1999. Report on the 1998 scanning review of European winter service technology. 238:5-7.

Valvoline. 2004. *Choosing Tire Chains*. http://valvoline.com/carcare/articleviewer.asp?pg=ccr20011201tc.

CHAPTER 3

THE ENVIRONMENTAL HISTORY OF THE CASCADE LAKES REGION – A REVIEW

Dr. Curt Stager and students from Paul Smiths College collect a sediment core sample from Lower Cascade Lake. Photo by Curt Stager

Chapter 3. **ENVIRONMENTAL HISTORY OF THE CASCADE LAKES REGION.**

Report prepared by Tom A. Langen, with contributions by Hanna Prutzman.

GENERAL INTRODUCTION

The Cascade Lakes and Chapel Pond, in the Town of Keene, Essex County, New York State, are popular recreation areas that also have great ecological value within the Adirondack Park. These lakes are impacted by New York State Route 73, a major highway that is sometimes called 'the gateway to the Adirondacks', but also suffer stresses from other human activities and natural causes associated with extreme winter weather. Recently, there have been concerns expressed in the press about the impacts of deicing salt and other winter road management activities on the roadside paper birch trees, *Betula papyrifera*, and the state-listed round whitefish, *Prosopium cylindraceum* (Fleck et al. 1988, Meyers 2002, Adirondack Council 2003).

In this chapter, we review the background information available about the environmental and human history of the Cascade Lakes and Chapel Pond, to help interpret the results of our research reported in Chapter 4 though Chapter 6. In this chapter, we (1) provide essential background on the natural and human history of the lakes; (2) provide background natural history on two species of most concern, the round whitefish and paper birch; (3) review previous research in the Adirondacks pertaining to the environmental impact of deicing road salt; (4) report on the history of chemical deicer and abrasive applications at the Cascade Lakes and Chapel Pond; and (5) report on changes in water chemistry over the last two decades at the lakes that can be attributed to deicing road salt.

Section 3-1. **HISTORY OF THE CASCADE LAKES REGION**

BACKGROUND NATURAL HISTORY

The Cascade Lakes and Chapel Pond lie within the High Peaks region of the Adirondack Park. They are geologically similar to other lakes in the region that were formed in the last 10,000 years after the retreat of glaciers at the end of the last ice age. Both the Cascade Lakes and Chapel Pond have steep anorthosite cliffs along the shoreline. The mountains bordering the Cascade Lakes are composed of anorthosite (igneous rock at least 90% feldspar) and syenite (silica poor equivalent of granite), and the waterfall that provides the lakes with their name exposes many minerals including calcite, feldspar, and garnet. Interestingly, the Cascade Lakes lie within a geological fault; Chapel Pond may also be within a fault zone. The Cascade Lakes are divided by what appears to be an ancient landslide (Isachsen and Fisher 1970, VanDiver 1976, 1985).

The lakes lie at 620 m above sea level (Cascade Lakes) and 400 m (Chapel Pond) at the headwaters of the East Branch of the Ausable River, which flows into Lake Champlain. The Cascade Lakes climate is similar to that recorded at the nearest weather station in Lake Placid, but winter weather is more extreme due to the higher elevation and narrow northeast – southwest orientation of the valley in which the lakes lie. The Cascade Lakes are subject to frequent high winds caused by the shape and orientation of the valley. Chapel Pond is climatically somewhat less extreme, because it is at a lower elevation and has a north – south orientation. The lakes are ecologically similar, and subject to similar anthropogenic stresses as other boreal lakes in the Adirondacks and elsewhere on the 'Boreal Shield' (Gunn and Pitblado 2004, Steedman et al. 2004).

The plant communities in the region of the lakes are classified as red spruce – northern hardwood forest, spruce-fir forest, and boreal talus woodland (Thompson and Sorenson 2000). After extirpation in the 19th century, beaver (*Castor canadensis*) returned to the Cascade Lakes by 1951 (as noted in a 1951 NYSDEC water chemistry survey of the lake). No invasive aquatic plant species has been detected at the Cascade Lakes, but Chapel Pond has not been surveyed (Hilary Oles, Adirondack Park Invasive Plant Program, *pers. comm.*). Purple loosestrife (*Lythrum salicaria*) has appeared at the Route 73 roadside at the Cascade Lakes, apparently as a contaminant in hay used in erosion control during roadwork. The infestation was eliminated using physical removal and glyphosate. Common reed (*Phragmites australis*), which can be associated with pooled water contaminated by road salt, has appeared near the Route 73 roadside at Chapel Pond. This invasive plant has also been eliminated using physical removal and glyphosate (Steven Flint, Adirondack Park Invasive Plant Program, *pers. comm.*).

A seasonal waterfall is located in the vicinity of the division of Upper & Lower Cascade Lakes. Heavy rain or snowmelt results in large transport of sediment and organic matter onto all three lakes. Major floods in the Ausable Valley that presumably affected the Cascade Lakes and Chapel Pond occurred in 1856, 1901, 1908, 1913, 1924, 1932, 1963, and 1979 (Bailey 1980).

Forest fire went though the area of the Cascade Lakes in 1903 and 1913 (Bailey 1980, McMartin 1994). Salvage logging of the mountains around the Cascade Lakes occurred after the fire (McMartin 1994). The birch stands around the Cascade Lakes are believed by residents to have established at this time, and this corresponds with tree ages reported in Fleck et al. (1988). However, there may also have been a fire in the 1940s or 1950s, but records are conflicting (Janet Hall, Keene Town Historian, *pers.comm.*).

The die-off of paper birch (*Betula papyrifera*) trees along Route 73 in the Cascade Lakes gorge was first noticed by the public around 1980, at the time of the Lake Placid Olympics. Residents attributed the mortality to deicing road salt, but exhaust emissions from motor vehicles was also blamed. Vehicle exhaust tends to accumulate in the gorge on calm days. Most of the dead trees fell and many live trees were severely damaged during the 1998 ice storm (Janet Hall, Keene Town Historian, *pers.comm.*).

HUMAN HISTORY

The human history of the Cascade Lakes and Chapel Pond is based on the unpublished *History of the Town of Keene, New York*, written in 1980 by James Bailey (Keene Town Historian), Barbara McMartin's (1994) *The Privately Owned Adirondacks*, and an unpublished report by Janet Hall, the current (2005) Keene Town Historian.

The Cascade Lakes was first settled in the 1780's or 1790's by William Edmonds, who cut the first trail into the area. The lakes were then known as the Edmonds Ponds in the Edmonds Ponds Gorge (also spelled Edmunds), but were also referred to in early maps as Long Pond. Apparently there was an iron ore bed that was mined briefly in the vicinity of the Cascade Lakes. Charcoal kilns were also operated in the vicinity of the Cascade Lakes in the 19th century. In

1891, it was reported that forests had been cut in the Cascade Lakes area 25 years before (i.e. about 1866), and in subsequent years had grown back into a dense secondary forest (McMartin 1994).

In 1878, a hotel (the Cascade House or Cascade Inn) accommodating 100 guests was built in the land between Upper and Lower Cascade Lake. The hotel was occupied year round, and a stable of horses was maintained. The lakes took their present names at this time, and the area became known as Cascadeville, with its own post office. The road apparently was extended on to Lake Placid at this time (or possibly a few decades earlier), although the main route from Keene ran behind Pitchoff Mountain on “the old military road” along what is currently the Jackrabbit Trail.

The hotel operated until 1905, and then became a private club. A hatchery was operated in association with the club, and the lakes were heavily stocked with trout. The club changed hands in 1923 and apparently the facilities were little used afterward except for one year (1927). New York State acquired the land from the Lake Placid Club in the late 1940s and the last building was demolished in 1947. A public picnic area was built by the New York State between the lakes at the site of the old hotel. It remains a popular rest stop. Also popular is the pull-off parking areas along Route 73 in the vicinity of the Cascade Lakes and Chapel Pond; these areas are heavily used by hikers and rock/ice climbers.

Since the Adirondack Park Act was passed in 1971, the guidelines for management of the lakes are described in the 1972 State Land Master Plan. NYSDEC manages the state lands that include the Cascade Lakes and Chapel Pond, and the Adirondack Park Agency provides oversight. The Cascade Lakes are governed by the Unit Management Plans for the Sentinel Range and High Peaks Wilderness Areas, and Chapel Pond is under the Dix Mountain Wilderness Area Unit Management Plan (e.g. New York State Department of Environmental Conservation 1999). As lakes within state land classified as ‘wilderness’, the Cascade Lakes and Chapel Pond receive the highest statutory protection under the State Land Master Plan.

Section 3-2. **CASCADE LAKES SPECIES OF CONCERN**

BACKGROUND NATURAL HISTORY OF THE PAPER BIRCH *BETULA PAPYRIFERA*

The paper birch tree, *Betula papyrifera*, is a medium-sized, fast growing tree that grows best on well-drained, sandy loam soils on cool, moist sites (Burns and Honkala 1990). Paper birch trees are commonly found in mixed hardwood-conifer forests, but may form pure stands as pioneer species at fire or logging - disturbed sites (Burns and Honkala 1990), as apparently has occurred in the Cascade Lakes.

Paper birch is a North American boreal species, adapted to cold climates where the average temperature in July ranges from 13 °C to 21 °C and snow is plentiful in winter (Marquis et al. 1969, Burns and Honkala 1990). Average temperatures above 21 °C can cause growth problems (Burns and Honkala 1990). Paper birch is shade-intolerant, which is why birch stands only last for one generation before being replaced by more shade tolerant species, usually conifers (Marquis et al. 1969).

Paper birch trees are a relatively short-lived species, surviving a maximum age of 140 years. Paper birch begin producing copious wind-dispersed seeds at 15 years of age, and their optimal seed-bearing age is between 40 and 70 years. Seed production averages 2.5 million seeds per hectare, and can be as high as 86 million seeds, which are dispersed from September to November (Burns and Honkala 1990). Seedbed condition is the most important factor in paper birch regeneration, and growth occurs best in organic, nutrient-rich soil. After germination, the most rapid occurs within the first 20 years, after which growth rate begins to decline (Marquis et al. 1969). Rooting is shallow and spreading, and no taproot is formed (Burns and Honkala 1990). Paper birch has been shown to be an excellent species for stabilizing coarse soil on steep slopes (Campbell and Hawkins 2003).

Paper birch roots have mutualistic ectomycorrhizal fungi (Jones et al. 1998). Ectomycorrhizae are functionally the nutrient- and water-absorbing “organs” of most woody plants, and mycorrhization of seedlings is considered to be critical in nutrient and water uptake (Jones et al. 1998). Mycorrhizae are essential for proper growth of some plant species, especially woody plants that grow in nutrient poor soils (Hausenbuiller 1985).

There are many factors that can cause a decline in abundance of paper birch trees, including mortality caused by successional replacement by shade-tolerant species, natural thinning and senescence, fungus and insect pathogens, changes in soil fertility, and climate change (Marquis et al. 1969; Burns and Honkala 1990). In the 1930s and 1940s there was a major regional dieback in paper birch due to unspecified deteriorating environmental conditions; trees that were shallow-rooted were the most vulnerable (Burns and Honkala 1990). The symptoms of declining health of birch include reduced growth, changes in foliage color, root death, and the dieback of twigs or branches that alter the tree crown (Marquis et al. 1969, Burns and Honkala 1990).

Fungal and insect infestations are one of the most common causes of paper birch tree damage and declining health. The principle fungi that cause disease in paper birch trees include *Poria obliqua*, *Fomes igniarius*, and *Pholiota spp*. (Marquis et al. 1969). The density of *Phytobia*, a dipteran stem miner, is higher on plantlets grown at high soil fertility (Ylioja and Rousi 2001). Shoestring root rot, caused by the fungus *Amelaria mellia,* can cause reduced growth rates and cause the tree to be more susceptible to decay and wind breakage (Marquis et al. 1969).

There are also many types of insects that attack paper birch trees. The bronze birch borer (*Agrilus anxius*) is the most serious insect pest of paper birch (Burns and Honkala 1990). This borer usually attacks mature trees that have already been weakened by some other agent (Marquis et al. 1969). Other insects that attack the leaves of paper birch trees include tent caterpillars (*Malacosoma disstria)*, the birch skeletonizer (*Bucculatrix canadensisella*), the birch leaf miner (*Fenusa pusilla*), the birch leaf-mining sawfly (*Phyllotoma nemorata*), and the birch casebearer (*Coleophora salmani*) (Marquis et al. 1969, Burns and Honkala 1990).

Lastly, changes in soil fertility can cause a decline in the health of paper birch trees that can lead to a decline in birch density due to mortality. Birch seeds and saplings require an organic, nutrient-rich soil to thrive (Marquis et al. 1969). Changes in the soil fertility, such as a decrease in essential nutrients or an increase in harmful metals, can have adverse affects on birch growth. However, paper birch appears to tolerate moderate levels (up to 100 mg/L) of aluminum, copper, nickel, or cobalt metals in soil (Patterson and Olson 1983). Heavy exposure to air pollution can result in declining health or death (James and Courtin 1985).

Additionally, birch seedling health and birch growth are improved by the presence of a diverse soil organism community. Experiments on European white birch (*Betula pendula*), a close relative of paper birch, show that seedling birch have lower root, stem and leaf growth in soils that are poor in soil microbes (bacteria, protozoa, and fungi) and microinvertebrates (Setälä & Huhta 1991, Liiri et al. 2001).

Paper birch has been found to be intermediately tolerant to sodium chloride, and can survive in semi-salty soil (Salt Institute 2004). An ecologically similar relative of paper birch, the yellow birch *Betula alleghaniensis*, has reduced germination success in soils with NaCl concentrations of 700 ppm (Bicknell and Smith 1975). For another ecologically similar birch species, European white birch *Betula pendula*, seedlings showed some signs of stress (leaf necrosis) in soil containing 28 mM NaCl and severe stress (necrosis and stunting) in 56 mM NaCl; the toxic effects of NaCl are exacerbated in sandy soils and under drought conditions (Fostad and Pedersen 2000). Paper birch's susceptibility to injury from aerial deposition of highway salt spray is intermediate (3 on a scale of 1 to 5; Cain et al. 2001).

In the most complete experimental study of the affects of sodium chloride and the alternative chemical deicer calcium magnesium acetate (CMA) on paper birch, Leiser and John (1990) applied deicer to soil and as a spray to vegetation on saplings. Five application levels were used (soil treatments: 0, 5, 10, 50, and 100 meq/L; foliage spray: 0, 0.1, 0.5, 1.0, 2.0 normal). Soil applications of sodium chloride caused little damage at low concentrations, but severe damage or death at the higher concentrations. Symptoms included tip, marginal and interveinal necrosis to leaves, defoliation, and stem dieback. CMA caused some decrease in leave size and shoot growth at the highest concentrations. Foliar spray applications of sodium chloride were damaging even at the lowest concentrations, and caused total stem die-back at the highest concentrations. At lower concentrations, twig die-back, leaf necrosis, and leaf abscission were evident. CMA spray resulted in some leaf malformation and spotting at the highest concentrations. Leiser and John (1990) concluded that sodium chloride is mildly to severely damaging when applied to soil or above-ground vegetation, whereas CMA did little damage. Paper birch was one of the most salt-sensitive species of the 18 tree and shrub species assessed in their study.

At the Cascade Lakes, paper birch is the dominant tree species, but is being replaced by boreal conifers (e.g. red spruce *Picea rubens* and balsam fir *Abies balsamea*). Dating via tree rings indicate that the cohort of trees established around 90 years ago (Fleck et al. 1988). Many birches are 'standing dead', and many have damaged crowns from ice storm breakage or damage caused by high winds, heavy snow and ice loads, or tumbling rocks (personal observation). The shoe-string root-rot fungus *Amelaria mellia* and the birch leaf miner *Fenusa pusilla* have also been implicated in declining health of birch trees in this region (Fleck et al. 1988). A dieback of birch along the New York State Route 73 roadside has drawn the attention of the public since the early 1980s, and road salt and sand abrasives have controversially been blamed for having caused roadside tree mortality (Fleck et al. 1988, Meyers 2002, Adirondack Council 2003).

BACKGROUND NATURAL HISTORY OF THE ROUND WHITEFISH *PROSOPIUM CYLINDRACEUM*

The round whitefish *Prosopium cylindraceum* is a member of the Salmon family (Salmonidae) in the subfamily of Whitefishes and Ciscoes (Coregoninae). The whitefish are cold-water fish inhabiting oxygen-rich, nutrient-poor water bodies of high-latitude North America and Eurasia. Few studies have been published on the natural history of round whitefish (Scott and Crossman 1973, Armstrong et al. 1977).

Round whitefish are benthic (bottom) feeders that feed on macroinvertebrates including mayfly larvae and pupae, caddisfly and chironomid larvae, small mollusks, leeches, crayfish, and fish eggs (Scott and Crossman 1973, Armstrong et al. 1977).

Round whitefish breed in November and the eggs hatch in April and May. Eggs are scattered on gravel shoals. Maturity is reached at about 3 years of age, and the fish live greater than 13 years in the wild (Scott and Crossman 1973).

We have not encountered any studies specifying the oxygen and temperature requirements of round whitefish, but the requirements are likely to be similar to lake trout (*Salvelinus namaycush*), which is the native species that most frequently co-occurs with the whitefish. Lake trout can inhabit water that is no warmer than 15.5 °C, but the optimal water temperature is below 10 °C. Adult lake trout can inhabit water with dissolved oxygen levels as low as 4 mg/L, but dissolved oxygen must be at least 5.5 mg/L to sustain all lake trout life stages, and a minimum of 6 mg/L oxygen is required if lake trout are to thrive (Ryan and Marshall 1994). Lake trout, and presumably round whitefish, can be ‘squeezed’ in late summer at small, thermally stratified lakes because well-oxygenated epilimnetic water is higher in temperature than is healthy for the fish, whereas sufficiently cold hypolimnetic water is oxygen-depleted. Late summer conditions may limit fish to a narrow band of suitable habitat in the metalimnion, resulting in extreme competition for food and other consequences of crowding. Under more extreme conditions, especially in small lakes, fish are forced into unsuitable habitat that results in accelerated mortality and reproductive failure (Dillon et al. 2003).

There have been no studies on toxicity of road salt contaminated water on the health of round whitefish. Adult round whitefish are known to inhabit brackish water at the mouth of the St. Lawrence River and near Hudson Bay (Scott and Crossman 1973). There is some sparse data on other salmonid fish that are physiologically similar to round whitefish, summarized in Evans and Frick (2001). Rainbow trout *Oncorhynchus mykiss* suffered 50% mortality from 24 hour exposure to NaCl concentrations of 5,500 mg/L (94 mM). Other studies on rainbow trout and brown trout *Salmo trutta* indicated that adults have no mortality from acute (24 hour) exposure to NaCl concentrations of 10,000 mg/L (171 mM). Rainbow and brown trout fingerlings were unaffected by a one week exposure to 800 – 1000 mg/L NaCl. Rainbow trout eggs were unaffected by 7-day exposure to NaCl concentrations up to 2,000 mg/L. A 27-day exposure at concentrations of 1,000 mg/L NaCl had no deleterious effects on rainbow trout embryos.

The Adirondack population of round whitefish is at the southern edge of this boreal species distribution. It inhabits cold, clear lakes, usually co-inhabited by lake trout. In the 19th and early 20th century, round whitefish was heavily fished for subsistence, and widely stocked. For reasons that remain unclear, the species disappeared from most lakes that it had formerly inhabited. Candidate causes of the decline include introduced competitor and predator fish species, siltation of breeding sites, lake acidification, and over-fishing by use of nets (Smith 1985, Carlson 2000). It is now listed in New York State as a statutory Endangered Species (Carlson 2000). Globally the species appears secure.

Upper and Lower Cascade Lakes have large populations of round whitefish, and eggs are harvested at the lakes for hatchery production and release in other lakes as a recovery effort for the species (Carlson 2002, Richard Preall, NYSDEC, *pers. comm.*). Population densities at the Cascade Lakes are among the highest in the Adirondacks (Geoffrey Steinhardt, Cornell University, *pers. comm.)*. Round whitefish has also been recorded at Chapel Pond.

The whitefish breed in November near the mouth of the inlet streams of Upper and Lower Cascade Lakes. Adult whitefish at the Lower Cascade Lake are much smaller and are thin relative to their size, in comparison to Upper Cascade and other Adirondack lakes (included lakes that have been stocked with fry of Cascade Lakes' fish; Richard Preall, NYSDEC, *pers. comm.*; Geoffrey Steinhardt, Cornell University, *pers. comm.*). Contaminant analysis of round whitefish at Upper Cascade Lake in 1986 indicated no detectable pesticides, whereas lead was 0.12 ppm and mercury 0.2 ppm (Carlson 2000). There has been some concern in the local Adirondack press about risks to the Cascade Lakes round whitefish populations due to contamination from deicing salt (Meyers 2002). Presently (2005), Geoffrey Steinhardt (Cornell University) is in the second year of a three-year study (funded by the State Wildlife Grant Program of the New York State Department of Environmental Conservation) on the natural history and conservation status of round whitefish in the Adirondack Park, which includes monitoring of the Cascade Lakes and Chapel Pond populations.

Water quality characteristics of lakes inhabited by lake trout (a frequent associate of round whitefish) in the Adirondack Mountains is presented in Gunn and Pitblado (2004). Environmental problems caused by roads and other anthropogenic stressors to lakes inhabited by lake trout, and by extension round whitefish, is reviewed in Steedman et al. (2004).

Section 3-3. **DEICING ROAD SALT RESEARCH IN THE ADIRONDACKS**

Overall, there has been little published research on the environmental impact of winter road management in the Adirondack Park region. Most reports have been incidental to other research on water quality, showing elevated chloride levels of streams and lakes near roads (e.g. Lawrence et al. 2004). Incidental to comprehensive research on acid precipitation, it was shown that lakes with elevated sodium and chloride concentrations in the Adirondack Park are almost always adjacent to major highways (National Acid Precipitation Assessment Program 1998). Chloride derived from atmospheric deposition originating from coal-fired power plant emissions is detectable but small, and is declining (Wagner and Steele 1989, Driscoll et al. 2001, 2003).

Recently published long-term research on water quality in the Mohawk and Hudson Rivers (rivers whose watersheds include the Adirondack Park) show that chloride levels have doubled over the last two decades (to around 40 mg/L), and the likely cause is increased loadings from deicing road salt (Goodwin et al. 2003, Kaushal et al. 2005). Over the last 50 years (1952 – 1998), in the Mohawk River sodium concentrations have increases 2.5 times and chloride levels 3.5 times, whereas other major anions and cations are unchanged or declining (Godwin et al. 2003). Godwin et al. (2003) estimate that a loading of 14 metric tons per km per year is necessary to result in the observed rise; these results parallel trends throughout the northeastern United States and eastern Canada (Environment Canada 2001, Kaushal et al. 2005).

Sediment cores of natural lakes and reservoirs in the Adirondack Park and elsewhere in the northeastern United States were collected, and sediment diatoms were used to reconstruct water quality conditions during pre-industrial times (pre 1850) and at present (Dixit et al. 1999). In the Adirondacks, 30 of 962 candidate natural lakes and 2 of 236 reservoirs were sampled. There was little change in the proportion of natural lakes classified as 'salt-impacted' (= greater than 200 μequiv/L Cl) in the Adirondack Park, and a modest increase in slightly salt impacted lakes (= 100-200 μequiv/L Cl), from 2% to 12% of sampled lakes. The proportion of lakes impacted at present, as indicated by sediment diatoms, and the increase in impacted lakes from pre-industrial conditions is much less than other regions examined by Dixit et al. (1999).

Demers (1992) and Demers and Sage (1990) reported on the effects of road salt contaminated streams along a 2 km stretch of State Highway 28 N near Newcomb, Essex County, New York during the winter of 1987-1988. The four streams were small, had low flow rates, and were bordered by northern hardwood forest. Thirty-seven metric tons of salt were applied during the study on the 2 km stretch of road, in a 7:1 ration of sand to road salt. Chloride levels were as much as 31 times higher downstream for the road than upstream, and averaged 5 times higher (5.23 mg/L versus 0.61 mg/L). Chloride levels remained elevated at least six months after the last deicing salt application. Due to inputs from the four feeder streams, the outlet lake (Rich Lake) had a slightly elevated chloride concentration at depth relative to its surface waters.

Demers (1992) assayed aquatic macroinvertebrates using Hester-Dendy samplers, and found that insect family richness was lower downstream, that Shannon-Weiner diversity was significantly lower downstream, and evenness was significantly lower downstream than upstream. The stream with the highest chloride levels had the most extreme differences between upstream (low salt contamination) and downstream (high contamination). She concluded that road salt contamination had negatively impacted the aquatic insect community of these streams.

Fleck et al. (1988) examined the impact of deicing road salt at the Cascade Lakes along New York State Route 73, during September 1985 to June 1986. The study was conducted, in part, as a response to concerns about tree mortality in response to aggressive winter road management (sand and salt applications) associated with the 1980 Winter Olympic Games in Lake Placid. Fleck et al. (1988) compared the Route 73 roadside environmental conditions to those 30 m to 40 m upslope. They documented that there was a higher mortality and lower regeneration of paper birch along the roadside that upslope. Soil calcium, magnesium, and potassium were lower along the roadside, whereas sodium was higher. The percentage composition of dry solids was higher at the roadside and organic matter was low, caused by an accumulation of sand from winter spreading as abrasive. Fleck et al. (1988) concluded that changes in the physical and chemical properties of the soil resulted in declining health of roadside birch, and that the long-term prognosis of the roadside birches was poor. They recommended research into planting salt-tolerant plants along the roadside, to reduce soil erosion. Further information on this study is reported in Chapter 4.

Current research in the vicinity of State Highway 28 N near Newcomb, Essex County, New York by Nancy Karraker (SUNY ESF) focuses on the impact of deicing road salt on vernal pool breeding amphibians. Her research indicates that road salt contaminates vernal pools up to 50 m inside a forest, and the contamination can make amphibian eggs inviable and reduce the health of amphibian larvae (Karraker 2005; Nancy Karraker, SUNY ESF, *pers. comm.*).

Vitaliano (1992) conducted a preliminary environmental economic study of the environmental costs of road salting in the Adirondack Park. Using contingent value surveys (willing to pay, willingness to accept) administered to campers on the perceived cost of tree dieback from road salt use, Vitaliano (1992) estimated that typical overnight campers value tree damage at $1.55 (1992 dollars). This can be interpreted as the amount of money campers would be willing to pay as a surcharge to implement policies that reduce or eliminate tree damage due to road salting.

Section 3-4. **WINTER ROAD MANAGEMENT OF NEW YORK STATE ROUTE 73**

HISTORY OF ROUTE 73 AT THE CASCADE LAKES

A dirt path was established to the Cascade Lakes in the late 18th century, but it was not extended up the gorge to Lake Placid until 1880 (Bailey 1980). Some widening and road improvement apparently occurred in 1927 (McMartin 2004). According to NYSDOT records (Mark Bonfey and John Remmers, NYSDOT, *pers. comm.*), Route 73 was widened and paved at the Cascade Lakes in 1932 in preparation for the 1932 Olympics. It is believed that the widening and paving of Route 73 at Chapel Pond also occurred during this period. By 1932, Route 73 had become the primary route to Lake Placid from New York downstate population centers. The road was resurfaced in 1953 and 1980. In 1979, NYS Route 73 was widened to include a climbing lane and the gabion baskets were installed to stabilize the steep slopes. Major structural work was also done in 1961. Maintenance to stabilize steep banks and manage rock falls is ongoing. Proposed road widening was not done in 1980. The road remains a high velocity (55 mph speed limit) two-lane highway at both the Chapel Pond and Cascade Lakes locations, with an average annual daily traffic volume in 2001 of 3360 vehicles per day. Route 73 is considered to be the ‘gateway to the Adirondack Park’, and as such is under special management as a New York State Scenic Corridor (Route 73 Steering Committee 1999).

HISTORY OF WINTER ROAD MANAGEMENT

Sand and NaCl have long been used on highways in the Adirondack Park. We have not seen documentation for the Adirondack highways, but inferring from historical practices in the Northeastern US it is likely that sand was first applied at the time of the paving of Route 73 in the 1930s, and deicing road salt was introduced in the 1940s. Calcium Chloride ($CaCl_2$) has been used locally to clear culverts in the last two decades, and new chemical deicer products (e.g. liquid $MgCl_2$, liquid sugar-$MgCl_2$ mixes) were introduced in the 1990.

In the early 1990s, concerns were raised about the environmental impacts of chemical deicers and abrasives to water bodies along Adirondack Park roads (Curran et al. 1990, Sutherland et al. 1990), and a recommendation was made to conduct research on the topic and develop new policies to reduce environmental impacts (Recommendation 208 in *The Adirondack Park in the 21st Century*, Commission on the Adirondacks in the 21st Century (1990)).

The Cascade Lakes section of Route 73 is exceptionally difficult to maintain during winter, due to high precipitation, low temperatures, steep valley walls (70°), poor drainage, and winds that blow snow onto the road surface. The area has averaged 77 ± 6.3 winter storm events requiring salting per year over the six-year period from winter 1998-99 to winter 2003-04. The typical 'snow season' lasts from November to April, and winter storm events can occur in October and May (Fig. 3-1). Blowing snow, melt-water runoff, and other sources of moisture result in the road surface being frequently wet, and result in lower persistence of chemical deicers, and therefore frequent reapplication.

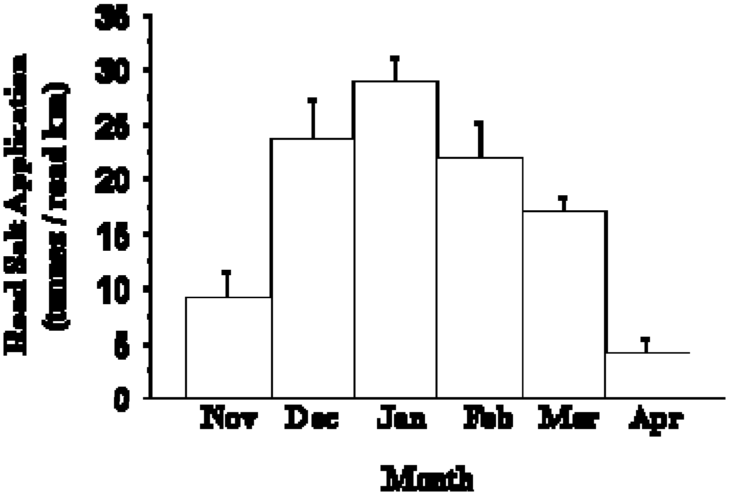


Fig. 3-1. Mean application rate of deicing road salt per month along Route 73 in the Cascade Lakes region. Data are the average of five winters (2000-2001 to 2004-2005).

NYSDOT uses an anti-icing strategy, applying materials before winter precipitation begins, to prevent a bond from forming between snow or ice and the road surface, and thus improving the effectiveness of snow plowing. Once precipitation has started to accumulate, after a plow has removed snow and ice from the road surface the chemical materials are again applied. Deicing salt (NaCl) is applied at 70.5 kg/lane km for a first run of a storm event and 31 kg/ lane km afterwards (Mark Bonfey, NYSDOT Essex County Residency, *correspondence*). Liquid calcium chloride is used to pre-wet the NaCl, which helps make the salt stick to the pavement, lowering application rates because scattering is lessened, and also results in the salt functioning at lower temperatures. Calcium chloride flakes, in conjunction with mechanical means, is used to keep open ditches and culverts that frequently become blocked with ice. Sand is currently used in low quantities as an abrasive when the temperatures are very low, when salting becomes ineffective.

In winter 1998-1999, liquid $MgCl_2$ was used, at an application rate of 71 L/lane km (Mark Bonfey, NYSDOT Essex County Residency, *correspondence*). Liquid $MgCl_2$ was also used as a pre-wetting agent for NaCl. The NYSDOT Essex County Residency deemed the product unsatisfactory for this type of environment, and discontinued use.

During winters 1999-2000 and 2000-2001, MAGIC™ brand chemical deicer (Sears Ecological Applications Inc.) was used, at an initial application rate of 71 L/lane km (Mark Bonfey, NYSDOT Essex County Residency, *correspondence*). MAGIC™ is a liquid compound containing $MgCl_2$ and an organic sugar substance (cane sugar or beet sugar byproducts) in a 1:1 mass ratio. Because this made the road surface slick, the application rate was reduced to between 35 and 47 L/lane km. The NYSDOT Essex County Residency deemed the product unsatisfactory for this type of environment, and discontinued use.

Sand was heavily used as an abrasive up to the 1980s but applications of sand was reduced to nearly nil in the 1990s (see below). When sand was used, it was reclaimed when feasible (Peter Howard, NYSDOT, *correspondence*). Sand remains a principal constituent of roadside soil at the Cascade Lakes and Chapel Pond (see Chapter 4), and large sand bars and sediment beds remain in Upper and Lower Cascade Lakes adjacent to the road (personal observation).

The NYSDOT Essex County Residency provided historical records of materials use, which we used to estimate loadings on the Cascade Lakes and Chapel Pond. The first record was total NaCl and sand applications from winters 1984-85 to 2004-05, and total center-lane road length serviced each year. The second set of data was all materials applied monthly in the 'Cascade Lakes' salt-spreader run (16.4 lane km = 8.2 center lane km) and 'Chapel Pond' salt-spreader run (18.9 lane km = 9.4 center lane km), during the winters 1998-99 to 2004-2005.

To estimate loadings during the most recent five-year period (winters 2000-01 to 2004-05), we measured the center lane distance of Route 73 through the watershed of each lake. We assumed that all materials applied on the road eventually enter the lake within whose watershed it lies. For Lower Cascade Lake, we only included loadings downstream from Upper Cascade Lake, which flows into Lower Cascade Lake and is therefore a loading source (see Chapter 6). We calculated the cumulative application rate (kg/center lane km-year) of each material on the Cascade Lakes and Chapel Pond salt-spreader runs, and multiplied these by our measured center lane distance in each lake's watershed (Table 3-1). We assume that all materials at the Chapel Pond watershed enter the lake, but it appears that a ditch along Route 73 opposite the lake transports some material away; therefore, the Chapel Pond loading is likely to be an overestimate. Because $CaCl_2$ is applied at problem spots as needed, it is unclear how evenly this substance is dispersed. Therefore, we calculated the $CaCl_2$ loading under the assumption that is evenly applied throughout a spreader-run, and under the assumption that all of it is applied within the focal lakes' watersheds. We believe that the latter is figure is probably a more accurate estimate.

Table 3-1. Loading (mean tonnes ± se) of chemical deicers and abrasives over the winters 2000-01 to 2004-05. Lower Cascade Lake loading excludes input from Upper Cascade Lake.

		Loading (tonnes)			
Lake	**Road Length (km)**	**NaCl**	**Sand**	**$CaCl_2$ (spread)**	**$CaCl_2$ (localized)**
Upper Cascade	1.45	151.88 ± 11.057	0.84 ± 0.552	0.04 ± 0.026	0.23 ± 0.162
Lower Cascade	1.95	204.25 ± 14.870	1.13 ± 0.742	0.05 ± 0.035	0.31 ± 0.217
Chapel	1.02	70.70 ± 4.489	0.07 ± 0.076	0.01 ± 0.006	0.15 ± 0.134

To estimate loadings at the Cascade Lakes over the period 1984-85 to 2004-2005, we calculated the difference in loading rate between the Cascade Lakes salt-spreader run and the overall Keene Residency average application rate, for the period 1998-99 to 2004-2005 (the period for which we had data for both). Cascade Lakes salt application was 3.33 times higher than the Keene average, and sand application was 3.80 times higher than the average. We then used the Keene Residency application rate for each material (tonnes/center lane km) multiplied by the calculated surplus loading factor and center lane distance of road in each watershed (Lower Cascade Lake distance excludes loading from Upper Cascade Lake). Sand loadings declined drastically in the 1990's to essentially nil in response to environmental concerns about siltation of water bodies (Fig. 3-2). Deicing salt loadings, in contrast, doubled over the period, apparently to compensate for terminating the use of sand (Fig. 3-3).

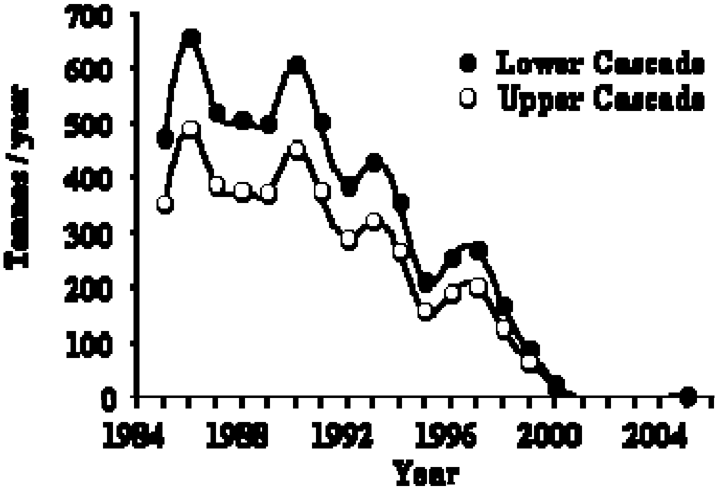


Fig. 3-2. Application rate of sand in the watershed of Upper Cascade Lake and Lower Cascade Lake (excluding Upper Cascade Lake) over the last two decades.

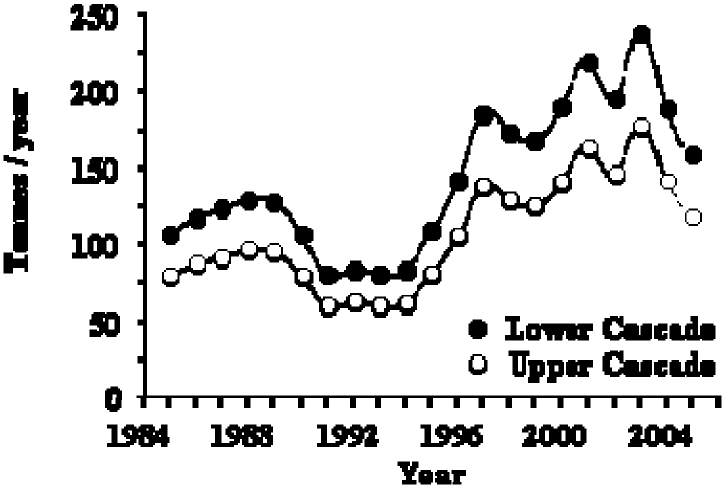


Fig. 3-3. Application rate of deicing road salt in the watershed of Upper Cascade Lake and Lower Cascade Lake (excluding Upper Cascade Lake) over the last two decades.

Section 3-5. **LONG-TERM CHANGES IN WATER QUALITY DUE TO DEICING SALT**

The concentration of chloride in a water body provides a standard indicator of deicing road salt contamination, particularly in regions such as the Adirondacks that are distant from significant natural sources of NaCl and other sources of chloride ions. Between 1984 and 1987, the Adirondack Lake Survey Corporation (ALSC) collected data on near-surface water chemistry and other limnological information on 1469 ponds and lakes sampled during July and August in the Adirondack Park (methodology and data available at *http://www.adirondacklakessurvey.org/*). Upper Cascade Lake (sampled July 1984) had the 13th highest chloride concentration, Lower Cascade (sampled July 1984) had the 21st highest, and Chapel Pond (sampled July 1986) had the 109th highest among the water bodies in the ALSC data set. We compared data from our July and August 2003-05 near-surface water samples of the three focal lakes (see Chapter 5) to the ALSC data. Upper Cascade and Lower Cascade Lakes presently have far higher chloride concentrations than typical Adirondack Lakes (Fig. 3-4). Lower Cascade Lake exceeds the lake with the highest chloride concentration in the ALSC data set (Bald Mountain Pond = 87.1 ppm, a lake that also suffers heavy road salt loadings), and the present Upper Cascade Lake mean chloride concentration is higher than all ALSC lakes, except the single highest ranked. Chapel Pond presently has a significantly higher chloride concentration than the average ALSC sample lake (Fig. 3-5), and would be ranked 121st highest of the 1469 lakes in that data set.

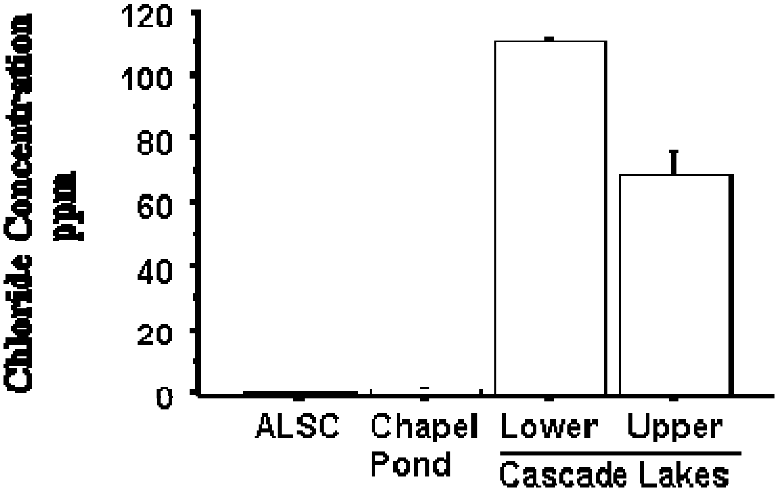


Fig. 3-4. Chloride concentrations (mean ± se near surface water) in the three focal lakes sampled in July and August 2003-05 compared to the 1469 lakes surveyed by the Adirondack Lake Survey Corporation in 1984-1987.

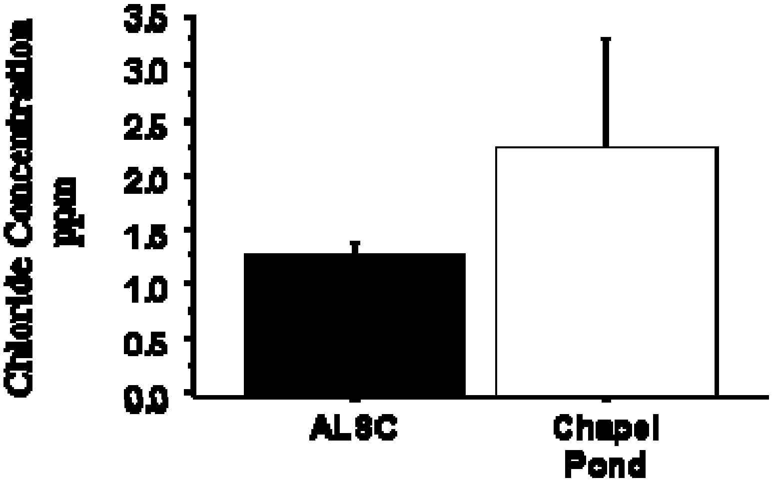


Fig. 3-5. Chloride concentrations (mean ± se near surface water) in Chapel Pond sampled in July and August 2003-05 compared to the 1469 lakes surveyed by the Adirondack Lake Survey Corporation in 1984-1987.

Besides the ALSC data (described above) and the data we collected in 2003-2005 (see Chapter 5), we received historical data on chloride concentrations in the focal lakes from three other sources. (1) James W. Sutherland (NYSDEC) provided water-quality data collected August 1985 to July 1986 from Upper and Lower Cascade Lakes, which was an unpublished component of the research reported in Fleck et al. (1988). (2) Michael DeAngelo (Paul Smiths College) provided water quality data collected by the Adirondack Watershed Institute (*http://www.paulsmiths.edu/PAGE=881/page.pl*) and collaborators, including Upper and Lower Cascade Lakes sampled April 1995 to September 2002, and Chapel Pond sampled May 1998. (3) Richard Preall (NYSDEC) provided water quality data from Upper Cascade Lake sampled August 2001. We analyzed chloride trends over time; splitting data into three depth categories because there are pronounced gradients of concentration with depth (see Chapter 5). Upper Cascade Lake (Fig. 3-6), Lower Cascade Lake (Fig. 3-7), and Chapel Pond (Fig. 3-8) all display a slight trend in increasing chloride concentration between 1984 and 2000, followed by a sharp increase that corresponds with the increased deicing salt loading in the late 1990s (Fig 3-3). Overall, each lake is around 4 times higher in chloride concentration at present than the mid 1980s, and Lower Cascade Lake is approaching the USEPA ambient water quality guidelines of 230 ppm chloride for aquatic life (United States Environmental Protection Agency 1988).

One surprising trend is that sodium chloride application rates increased in the late 1990s, but elevated chloride levels only appeared after 2000, when we initiated our study. Richard Preall's (2001) sample of Upper Cascade Lake also showed similar elevated chloride concentrations as ours, whereas the data provided by Michael DeAngelo from the late 1990s and one sample in 2002 indicated lower concentrations. Our chloride measurements were cross-validated using two methods (ion specific chloride electrode and ion chromatography) and compared to our own standards and a standard water source (see Chapter 5). We believe that the lag is real, and is caused by a lag in the system's response to loading changes, which results from movement of chloride through the soil followed by eventual groundwater discharge into the lakes. Our chloride mass balance model indicates that it can take around 5 years for lake concentrations to reach a new equilibrium after a change in loadings (see Chapter 6).

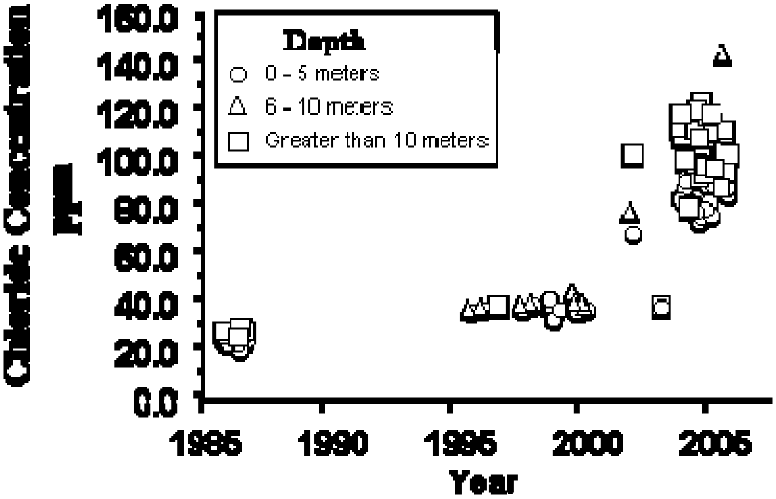


Fig. 3-6. Chloride concentrations over the last two decades at Upper Cascade Lake.

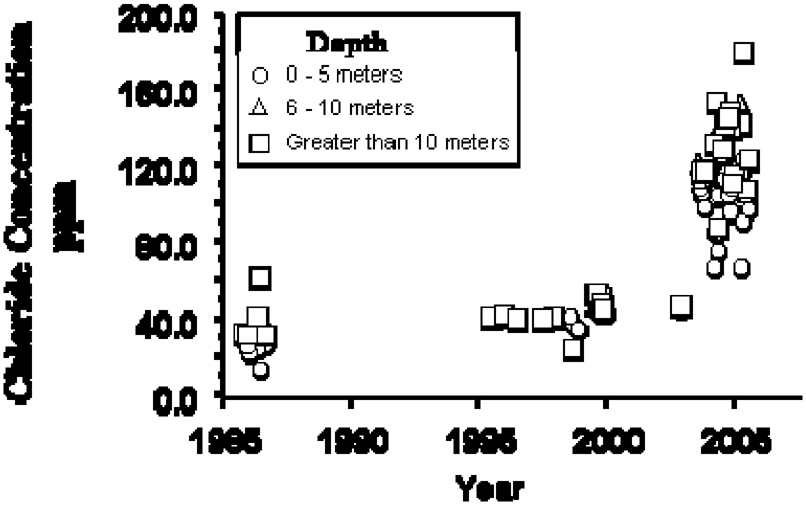


Fig. 3-7. Chloride concentrations over the last two decades at Lower Cascade Lake.

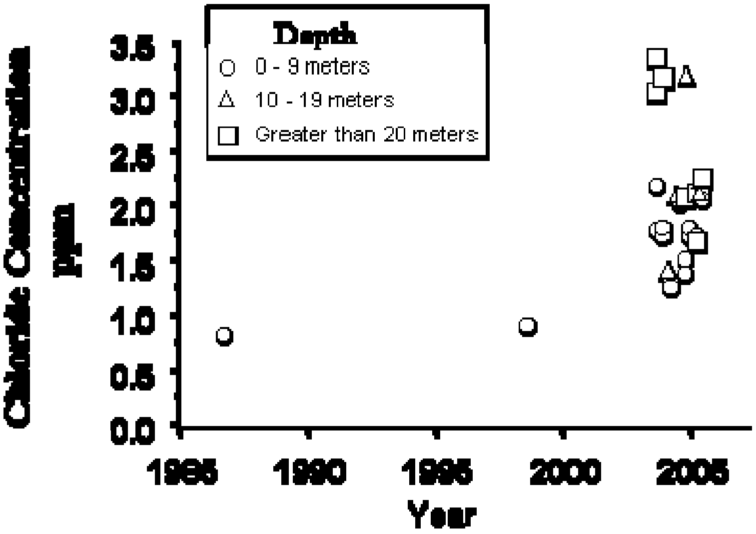


Fig. 3-8. Chloride concentrations over the last two decades at Chapel Pond.

CHAPTER 4

IMPACT OF WINTER ROAD MANAGEMENT ON SOIL AND VEGETATION COVER ALONG NEW YORK STATE ROUTE 73

*Dead paper birch (*Betula papyrifera*) and degraded soil along Route 78 at Upper Cascade Lake. View is facing west. Photo by Curt Stager.*

Chapter 4. **IMPACT OF WINTER ROAD MANAGEMENT ON SOIL AND VEGETATION COVER ALONG ROUTE 73.**

Report prepared by Tom A. Langen and Hanna Prutzman.

GENERAL INTRODUCTION

In terrestrial environments, the use of chemical deicers and abrasives as measures to control snow and ice accumulation of roads can have severe environmental effects. Changes in roadside soil structure and chemistry, and declines in the health of roadside vegetation are the most perceptable consequencs of winter road management (see Chapter 1).

Along New York State Route 73 in the vicinity of the Cascade Lakes, Essex County New York, there has long been concern about the affects of deicing road salt and sand on roadside stands of paper birch (*Betula papyrifera*). An apparent die-back of birch is readily observed by motorists, and the issue has drawn the concern of environmental advocacy organizations (e.g. Meyers 2002, Adirondack Council 2003).

Fleck et al. (1988) reported on research pertaining to the decline in paper birch and changes to soil structure at the Cascade Lakes, based on research conducted in Fall 1985 to Summer 1986. During this study, they surveyed the abundance and condition of trees, and the physical and chemical properties of soil, comparing the roadside to conditions upslope 30 m from the road. They concluded that changes in soil chemistry and physical structure caused mortality to mature roadside paper birch trees, and inhibited regeneration.

In Chapter 4, we report on an expanded follow-up study to Fleck et al (1988). In 2003-2005, we surveyed birch tree abundance and condition, and soil physical and chemical properties at the locations surveyed by Fleck et al. (1988). We also surveyed two additional locations: the lakeside forest across the lakes from the highway, and the roadside at Chapel Pond. In addition, we conducted a preliminary study on aerial transport of road salt from the highway to surrounding vegetation and snow. The questions addressed include (1) *Have conditions deteriorated due to winter road management from the conditions documented in Fleck et al. (1988)?* (2) *What are the root causes of changes in vegetation along the roadside?* (3) *What is the prognosis for future conditions of roadside soils and vegetation, given present winter management practices*?

Section 4-1. **SOIL**

INTRODUCTION

We investigated whether the application of road salt and sand has altered the soil conditions on the Route 73 roadside at Upper and Lower Cascade Lakes, with the aim of understanding whether soil chemical and physical structure had been altered to the extent of potentially affecting health of roadside plants. We compared the Cascade Lakes roadside soils to soil collected 30 m upslope from the highway, soil in forest across the lakes for the highway, and soil collected from the roadside at Chapel Pond. We also compared our results to soils collected along the roadside and upslope areas two decades previous (Fleck et al. 1988). We examined physical properties of the soil (e.g. density, particle size), amount of soil organic matter, extractible concentrations of sodium and chloride, and extractible concentrations of nutritive cations (calcium, magnesium, and potassium). All of these are features of the soil that can affect the health of plants.

Based on the history of use of chemical deicers and abrasives along Route 73, and on the known environmental effects of road salt and sand, we predicted:

- The concentration of extractable sodium and chloride in the soil would be higher along the highway than locations distant from it.
- The concentration of extractable sodium and chloride in the roadside soil would be higher in 2003 - 2005 than in 1985 - 1986, when soil properties were last measured.
- The concentrations of extractable potassium, magnesium and calcium in the soil would be lower in the roadside soils than elsewhere.
- The concentrations of extractable potassium, magnesium, and calcium in the soil would be lower along the road in 2003 - 2005 than in 1985 - 1986, when soil chemical properties were last measured.
- The roadside soil would contain more sand and less organic matter than soil collected a distance away from it.

METHODS

Study Plots

A total of thirty-one 40 m x 5 m plots were delineated in the vicinity of State Route 73 at Upper and Lower Cascade Lakes and at Chapel Pond. The long-axis of each plot ran parallel to the highway. Locations included the roadside between the road surface and the lake at the Cascade Lakes ('Cascade Lakes Roadside', *n* =14 plots) and Chapel Pond ('Chapel Pond Roadside', *n* = 5 plots), along the far shore of the Cascade Lakes ('Cascade Lakes Farside', *n* = 7 plots), and 30 m upslope from the road at the Cascade Lakes ('Cascade Lakes Upslope', *n* = 5 plots); see Fig. 4-1-1. The Cascade Lakes Roadside, Chapel Roadside, and Cascade Lakes Farside plots were established in July and August 2003, whereas the Cascade Lakes Upslope plots were delineated in April 2005.

Figure 4-1-1. General location of the three sites at which we established plots at the Cascade Lakes. This view is looking east on New York State Route 73, with Lower Cascade Lake on the left. Photo by Michael Twiss.

The plot location and spacing were chosen based primarily on a study that was performed by Fleck et al. (1988), but with the addition of the Cascade Lakes Farside and Chapel Pond Roadside sites. A Global Positioning System (Garmin GPS 76 brand with WAAS correction) was used to mark the location of each plot. The starting point for the first plot was parallel to the head inlet of Upper Cascade Lake or Chapel Pond. The starting points for the remaining plots were selected by measuring out 150 m from the previous plot. For the Farside and Upslope plots, the siting was constrained by accessibility. The only criteria for plot selection was that each plot had to be at least 150 m away from any other plot and the terrain had to be safely accessible.

Soil Preparation

Using an Edelman combination soil auger, soil samples were collected from each plot at Roadside, Chapel and Farside in October 2003, April 2004, July 2004, October 2004, and April 2005. Upslope samples were collected once, in April 2005. Samples taken from the Cascade Lakes Roadside were approximately 4 m from the road. Chapel Pond soil samples were taken between 4 m to 8 m from the road depending on the terrain. Soil samples were not collected when the ground was frozen.

Soil cores were taken at the center of the plots. Cores were taken at 15 cm from the surface. This was done by taking the first surface soil core and discarding it, then placing the corer back in the initial hole and removing a second core. This core was bagged and labeled with the appropriate plot name. Three cores were taken from each plot in the same way. After the soil was bagged, the auger was wiped clean.

Soil from each of the 31 plots was placed in brown paper bags and weighed, then dried in a drying oven at 110ºC. The samples were weighed after 24 h, and again after an additional 14 h to ensure that there was no more water loss. Samples were then sifted through two sieves in order to remove the gravel (4 mm) and fine gravel (2 mm). The remaining sample, which consisted particles smaller than two millimeters, was transferred to a sealed plastic bag and placed in a –20 °C freezer.

While wearing latex gloves, five grams of the sieved soil was mixed with 100 mL of deionized water (water which passed through four steps of purification through the Millipore Milli-Q Gradient system, also referred to as Q-water) in a nitric acid-washed Erlenmeyer flask. The flask was labeled, sealed with parafilm, and placed on a shaker for 2 h. The soil-water mix was filtered through a No. 1 fluted piece of filter paper. The remaining soil in the flask was discarded. The volume of aqueous extract was recorded and the pH was measured using a pH probe (Thermo Orion model 620). The extracts were placed into nitric acid-washed 125 mL polypropylene narrow-mouthed bottles and stored a refrigerator at 2.5 °C. This process was done in triplicate for each plot.

Soil Sieving

The April 2004 (April 2005 for Upslope), soil samples were sieved using US Standard sieves and a 152H Hydrometer, in order to obtain soil fractions for each plot (Lui and Evett 2003). For a soil sample that was predominantly silt and clay, a sample of 50 g was analyzed, whereas when the soil was predominantly sand, a 100 g sample was used. The samples were placed into a 250 mL beaker and covered with 125 mL of sodium hexametaphosphate solution (40 g/L) to suspend the particles. The samples were stirred to put into solution and then soaked for sixteen hours in the solution. At the end of the soaking period, the sample was further disaggregated by transferring the soil-slurry to a dispersion-cup. The beaker was rinsed with Q-water to make ensure the residue was collected in the dispersion-cup. Q-water was added until the cup was at least half full. The soil-slurry was then stirred for one minute using a mechanical stirring machine.

The stirred soil-slurry was then transferred into a glass cylinder and de-ionized water was added until it reached a volume of 1000 mL. A rubber stopper was used to cork the open end while the glass cylinder was inverted 60 times (about 1 min) to complete agitation. After the cylinder was inverted, it was placed in a convenient location and hydrometer measurements were taken at 2, 5, 15, 30, 60, and 250 min, and 24 h using a 152H hydrometer. The hydrometer was placed into the glass cylinder, to the appropriate depth so that the hydrometer would not agitate the slurry, for 20 to 25 s before each reading was taken, then removed and cleaned in a beaker of de-ionized water. The temperature was taken after each reading. After 24 h the suspension was then transferred to a No. 200 sieve (75µm), and washed with tap water until the wash water was clean. The sample retained on the No. 200 sieve was then transferred to a drying pan and placed into a drying oven at 110 ºC until the sample was dry. The portion retained was used to perform a sieve analysis.

Each of the following sieves was weighed prior to taking measurements: 10, 40, 60, 80, 100, 170, 200 and the bottom pan. The sieves were stacked appropriately and the dried sample was weighed and placed into the top sieve of the stack. The stack of sieves was then placed into the Ro-Tap (RX29) Sieve Shaker, which has 278 oscillations and 150 taps per min, for 9 min or until less than 1% of the mass of the soil retained on the sieve passed through the sieve in one minute. The mass of each fraction was then weighed on a balance and recorded. In the end the sum of the masses retained should equal closely the original mass of the soil that was sieved. The samples were placed in a separate bag and stored at –20 °C in a freezer.

Soil Specific Gravity

Specific gravity was measured for the April 2004 samples (April 2005 for Upslope), using the method in Lui and Evett (2003). To calibrate the pycnometers, a dry pycnometer was weighed, then filled with Q-water and re-weighed. The temperature was recorded from a thermometer to the nearest 0.1ºC. The mass of the pycnometer and water can then be calculated for any other water temperature using the following formula:

$$(1)\quad M_{pw}\ (\text{at } T_x) = \frac{\text{density of water at } T_x}{\text{density of water at } T_i}\left[M_{pw}\ (\text{at } T_i) - M_p\right] + M_p$$

Where:

T_x = any other desired temp.

T_i = observed temp.

M_p = mass pycnometer, g

M_{pw} = mass pycnometer and water, g

This calibration procedure was performed on 5 different pycnometers.

To determine the specific gravity, the mass of the pycnometer was verified within 0.6 g of the calibrated mass. If the mass was not within 0.6 g, the pycnometer was recalibrated. Then ~100 g of dry soil from a sample was added to the pycnometer. The amount of soil was recorded as M_s. Q-Water was added to the pycnometer until the water level was 1/3 to ½ of the depth of the main body. The soil was agitated until a slurry was formed. A squirt bottle of Q-water was used to rinse any soil adhering to the side of the pycnometer, then a stopper with a hole was used to stopper the pycnometer. This procedure was repeated for the other four pycnometers. The five pycnometers were then placed in an agitating water bath and hooked up to a vacuum apparatus to de-air the samples. After the water bath reached ~40 ºC, the vacuum was applied for at least 1 h.

Air was removed by placing de-ionized in a 1000 mL volumetric flask and boiling it on a hot plate for at 1 h. After the pycnometers had been in the vacuum apparatus 1 h they were removed, and then filled to the calibration mark using the de-aired de-ionized water. This was done using a piece of small diameter flexible tubing with its outlet kept just below the surface of the slurry in the pycnometer. The tubing was used so that air was not reintroduced into the pycnometer. The pycnometers were then placed in an incubator along with a bottle of de-aired water and a beaker of water containing a thermometer. The pycnometers remained in the incubator for 24 h.

After 24 h, the pycnometers were removed. If the water was below the calibration mark, the water that was in the incubator was added using the tubing, as before. The pycnometers were placed on a piece of Styrofoam and the mass of the pycnometer, soil and water was recorded to the nearest 0.01 g. The temperature of the slurry/soil-water was measured to the nearest 0.1 ºC using the same thermometer and method as used in calibration.

Soil Permeability

In July 2004 (April 2005 for Upslope), soil penetrometer measurements were taken at each plot using an ELE Pocket Penetrometer Model 29-3729. Five measurements were taken for the twenty-six plots in order to calculate an average value for the unconfined compression strength of surface soil. The penetrometer indicator sleeve was placed at zero before starting. The penetrometer was then placed perpendicular to the soil and the piston needle was pushed, at a steady rate of penetration, until the foot was flush with the soil. The index sleeve was placed back to zero and the process was repeated 4 more times for a total of five measurements. If the soil penetrometer reached its maximum of 4.6 kg/cm^2 without full penetration then that was recorded as 4.6 MAX.

Bulk Density

In July 2004 (April 2005 for Upslope), bulk density was tested in each plot by hammering a known volume of PVC pipe into the ground until the top was flush with the soil surface. The pipe was carefully removed and all the soil within was placed into a plastic bag and labeled. The samples were weighed wet and then dried in a drying oven at 110 °C. After 24 h each sample was weighed and the dry weight was recorded, then the samples were placed back into the oven for another 24 h. The samples were reweighed then transferred into sealed plastic bags and stored at –20 °C. The dry weight was divided by the volume of the cylinder to obtain the bulk density value for each plot.

Soil Color

The soil color for each plot was performed on the April 2004 samples (April 2005 for Upslope). A small portion of each sample was placed on a white piece of paper under a Sunshine in a Box incandescent bulb (Verilux), which contains all wavelengths of visible light to mimic sunlight. A Munsell Soil Color Chart was then used to visually determine the color of each plot sample.

Loss on Ignition and Cation Exchange Capacity

The amount of organic matter was determined on the remaining sieve analysis samples. Twenty-four porcelain crucibles were weighed and 20 g of soil from each plot was placed into the appropriate crucible. The crucibles were placed in a drying oven at 105 °C for five hours then cooled to room temperature and reweighed to determine if any water was lost.

Once the soil samples were dry, the crucibles were then placed in a muffle furnace at 450 °C for 5 h to burn off the organic matter. The crucibles were removed from the furnace, cooled to room temperature and reweighed. The weight of the crucible, Wc, was subtracted from the weight of the crucible and inorganic mineral, Wcs, to give the weight of organic matter lost. The percent of organic matter was then calculated by dividing the weight of organic matter lost by the weight of soil after drying and multiplied by 100. The percent organic matter was then used to calculate the Cation Exchange Capacity (CEC).

Cation exchange capacity was calculated using the following formula, taken from Hausenbuiller (1985):

$$(2)\ \text{CEC (meq/100g)} = [(\%\text{Clay} \times 30 \text{ meq/100g}) + (\% \text{ Organic Matter} \times 200 \text{ meq/100g})] \times 0.01$$

The percent clay numbers were taken from the hydrometer results. Clays have a diameter size of ≤ 0.0002 mm, so the percent of soil particles finer than that diameter was used for the percent of clay in that sample. The percent organic matter was taken from the loss on ignition results. The constants, 30 and 200, correspond to the average cation exchange capacity for illite, the primary clay mineral in this soil, and the average cation exchange capacity for organic matter respectively (Hausenbuiller 1985).

Soil Cation Concentrations

Flame atomic emission spectroscopy (Perkin Elmer AAnalyst 200 Atomic Absorption Spectrophotometer) was used to measure the amount of sodium and potassium in each soil extraction. Samples were prepared by pipetting 10 mL of sample and 10 mL of 2% nitric acid into a 50 mL acid-washed conical tube. This made a 1:2 dilution to overcome matrix modification. Standards for both sodium and potassium were prepared in 2% nitric acid in a laminar flowhood to prevent contamination.

Flame atomic absorption spectroscopy (Perkin Elmer AAnalyst 200 Atomic Absorption Spectrophotometer) was used to measure the amount of calcium and magnesium in each soil extraction. Samples were prepared by pipetting 10 mL of sample, 9.8 mL of 2% hydrochloric acid and 0.2 mL of 5% lanthanum chloride into a 50 mL acid-washed conical tube. This resulted in a 1:2 dilution, and the lanthanum was used to combat matrix effects. Standards for both calcium and magnesium were prepared in 2% hydrochloric acid with the added matrix modifier lanthanum.

Standard calibration curves were made prior to running the samples. All samples were analyzed, and a known sample containing SLRS-4 (a standard water sample for quality control analysis) was run to test the atomic absorption/emission spectrophotometer accuracy. For any result greater than 10% of the known value for the SLRS-4 sample, we rejected the cation results and reran the calibration curve and samples. Detection limits of the spectrophotometer were calculated for all cations by running the blank five times and calculating the standard error. All samples were multiplied by a dilution factor of two and converted from part-per-million (ppm) to mg/kg (a standard way of expressing concentration in soils).

Chloride Concentrations

Eight liquid soil extractions were filtered through a 0.2 micron syringe filter, and then measured using a Thermo Orion Ion Selective Chloride Electrode Model 9217. The measurements were below 2 ppm, which was under the detection limit of the electrode. Therefore, all extracts were syringe-filtered through a sterile 0.2 micron pore size filter, and chloride concentrations were measured using ion chromatography.

RESULTS

The soils were dramatically different in physical and chemical composition among sites. The Cascade Lakes Roadside soil samples were primarily composed of sands, and were very lacking in silts and clays in comparison to the Upslope, Farside, and Chapel Pond Roadside sites (Table 4-1-1). As a consequence of being composed of different materials than the other sites, the Cascade Lakes Roadside soil samples were much lighter in color (Table 4-1-2). Cascade Lakes Roadside soils were much more dense (as indicated by bulk density and specific gravity), and the surface was less penetrable (Table 4-1-3). Organic content was far lower at the Cascade Lakes Roadside plots than elsewhere (as indicated by loss on ignition, Table 4-1-3). As a consequence of having a very low content of clays and organic matter, Cascade Lakes Roadside soils had a much lower cation exchange capacity than other sites (Table 4-1-3). The soil composition also resulted in much less water being retained within the sampled soil at the Cascade Lakes roadside than other locations (Table 4-1-3). In general, the Chapel Pond Roadside samples were more variable than other sites, and the Chapel Pond Roadside was intermediate in most physical properties between the Cascade Lakes Roadside and Cascade Lakes Upslope & Farside samples.

The Cascade Lakes and Chapel Pond Roadside soils had somewhat higher concentrations of chloride than the Upslope and Farside sites (Table 4-1-4). Sodium concentrations were much higher in the Cascade Lakes and Chapel Pond Roadside soil samples than the Upslope and Farside sites (Table 4-1-4). The nutritive cations calcium, magnesium, and potassium were in much lower concentrations in the Roadside soil samples than the Upslope and Farside sites (Table 4-1-4). The Cascade Lakes Roadside soil was more depauperate in these cations than the Chapel Pond Roadside soil (Table 4-1-4). The pH was lower (i.e. the soil was more acidic) at the Upslope and Farside sites than the Roadside soils (Table 4-1-4). There were no clear seasonal trends in the concentration of sodium or chloride (Fig. 4-1-2), nor for the nutritive cations. There was a greater than 1: 1 ratio of sodium to chloride ions in the soil, as expected given their different mobility caused by their charge properties (Fig. 4-1-3).

Chloride and sodium concentrations appear to be little changed between 1985-1986 and 2003-2005, whereas the concentrations of the nutritive cations calcium, magnesium, and potassium appear to have declined dramatically (Table 4-1-3). These comparisons must be treated cautiously, however, because of differences in procedures between the Fleck et al. (1988) study and ours (See DISCUSSION, below). The pH of the Cascade Lakes Roadside soils appear to be unchanged, whereas the Upslope soils are more acidic. This latter result may be a consequence of greater abundance of conifer at present than two decades ago (See Section 4-2).

Table 4-1-1. Soil grain fractionation, as indicated by the percent dry mass retained in sieves of progressively finer mesh.

		Cascade Lakes			**Chapel Pond**
Soil Type	**Sieve**	**Roadside**	**Upslope**	**Farside**	**Roadside**
very coarse sand	10	0.1	0.1	0.2	0.2
coarse sand	40	27.2	25.1	28.5	20.7
medium sand	60	28.4	18.1	17	15.3
fine sand	80	15.1	10.8	10	8.8
	100	5.9	5.4	4.5	3.9
very fine sand	170	12.6	7.6	11.5	13.9
	200	2	6.9	2.6	5.6
silt & clay	pan	8.7	25.9	25.6	31.3

Table 4-1-2. Color of soil samples, scored using a Munsell soil color chart.

		Cascade Lakes			**Chapel Pond**
Munsell Code	**Color**	**Roadside**	**Upslope**	**Farside**	**Roadside**
10 YR 5/3	Brown	9			1
10 YR 6/2	L. Brownish Gray	3			
2.5 Y 6/3	L. Yellowish Brown	2			
7.5 YR 3/2	Dark Brown		2	2	
10 YR 4/2	Dark Grayish Brown		1		
7.5 YR 3/1	Very Dark Gray		1		
7.5 YR 4/3	Brown		1		1
10 YR 3/4	Dark Yellowish Brown			1	
2.5 YR 2.5/2	Very Dark Brown			1	
5 YR 2.5/1	Black			1	
5 YR 2.5/2	Dark Reddish Brown			1	
7.5 YR 4/4	Brown			1	2
10 YR 5/2	Grayish Brown				1

Table 4-1-3. Summary of physical features of the soil samples (mean ± se).

	Cascade Lakes			**Chapel Pond**
	Roadside	**Upslope**	**Farside**	**Roadside**
Bulk density	1.6 ± 0.02	0.6 ± 0.05	0.5 ± 0.11	1.1 ± 0.21
Specific gravity	2.8 ± 0.01	2.6 ± 0.09	2.5 ± 0.09	2.6 ± 0.09
Cation exchange capacity	2.8 ± 0.55	27.1 ± 6.21	40.4 ± 23.67	17.7 ± 13.69
Loss on ignition	1.7 ± 0.07	11.7 ± 3.15	18.6 ± 11.59	7.7 ± 6.39
Penetrability	4.0 ± 0.17	3.1 ± 0.88	3.7 ± 0.17	3.0 ± 0.33
Soil water retention	6.6 ± 0.24	29.5 ± 6.97	32.4 ± 1.81	20.2 ± 5.18
N	14	5	7	5

Bulk density = g / cm^3
Cation exchange capacity = meq / 100 gm
Loss on ignition = % mass loss with incineration
Penetrability = kg / cm^2
Soil water retention = % mass loss with drying

Table 4-1-4. Concentrations (mean ± se mg/kg soil) of ions and pH from soil collected at the four sites. N = total number of samples; the two Roadside and the Upslope plots were each sampled five times.

	Cascade Lakes			Chapel Pond
	Roadside	**Upslope**	**Farside**	**Roadside**
Chloride	31.5 ± 3.41	25.6 ± 1.79	27.0 ± 1.92	31.5 ± 3.36
Sodium	103.1 ± 3.24	74.7 ± 4.95	43.9 ± 1.57	136.9 ± 11.00
Calcium	4.5 ± 0.60	135.7 ± 14.25	167.5 ± 13.45	43.3 ± 9.90
Magnesium	0.2 ± 0.02	19.9 ± 2.90	23.1 ± 2.87	3.3 ± 0.88
Potassium	1.0 ± 0.10	46.2 ± 8.63	29.7 ± 3.40	7.5 ±1.50
pH	6.9 ± 0.02	5.5 ± 0.04	5.9 ± 0.16	6.3 ± 0.17
N	70	5	35	25

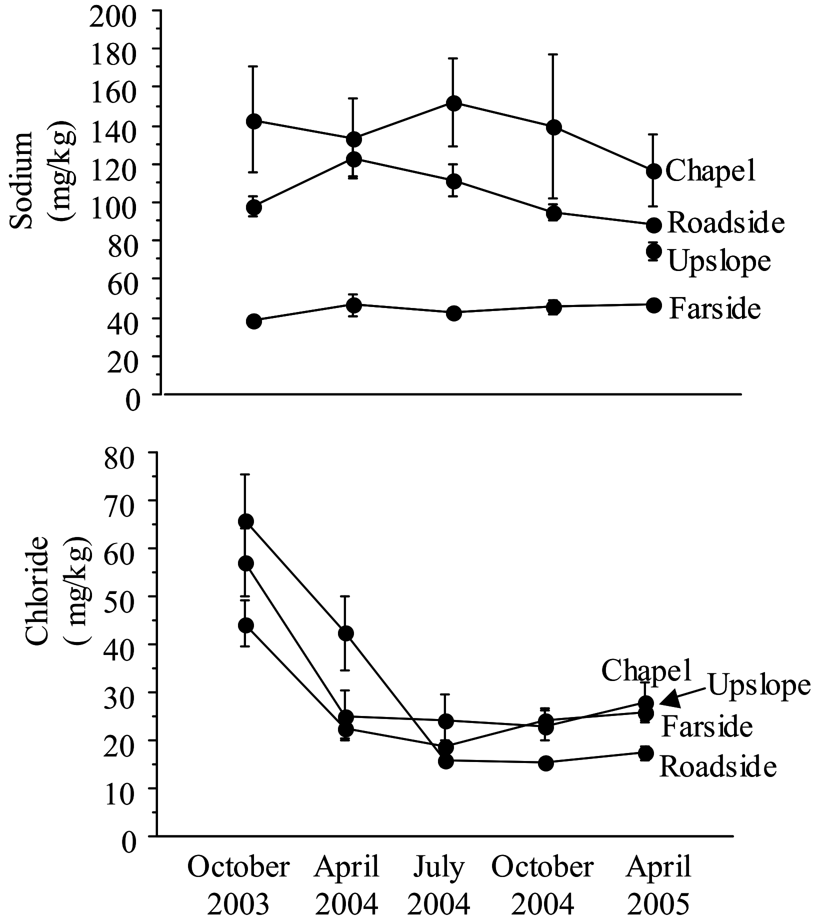


Fig. 4-1-2. Seasonal trends (mean ±se*) in sodium and chloride concentrations of soil collected at the three sites that were repeatedly measured in 2003 to 2005. Upslope, which was only sampled in April 2005, is also indicated.*

Table 4-1-5. Comparison of the concentrations (mean ±se *mg/kg soil) of ions and pH from soil collected in the same month at the Upslope and Roadside sites in 1985-1986 and 2003-2005. Data from 1985-1986 from Fleck et al. (1988).*

		Roadside		**Upslope**
		April	**October**	**April**
Chloride	**1985-1986**	79.8 ± 12.3	34.9 ± 0.89	59.8 ± 16.5
	2003-2005	29.9 ± 4.53	40.7 ± 6.75	25.6 ± 1.79
Sodium	**1985-1986**	374 ± 37.7	65.9 ± 11.3	38.5 ± 5.90
	2003-2005	105.7 ± 5.89	96.3 ± 3.26	74.7 ± 4.95
Calcium	**1985-1986**	520 ± 39.8	2670 ± 52.2	4800 ± 1064
	2003-2005	3.0 ± 0.30	7.8 ± 1.23	135.7 ± 14.25
Magnesium	**1985-1986**	33.5 ± 3.53	15.3 ± 2.60	518 ± 102
	2003-2005	0.2 ± 0.02	0.2 ± 0.04	19.9 ± 2.90
Potassium	**1985-1986**	33.3 ± 2.54	22.6 ± 3.26	278 ± 59.0
	2003-2005	0.9 ± 0.14	0.7 ± 0.49	46.2 ± 8.63
pH	**1985-1986**	8.1 ± 0.20	6.2 ± 0.04	6.2 ± 0.10
	2003-2005	6.8 ± 0.04	6.9 ± 0.04	5.5 ± 0.04

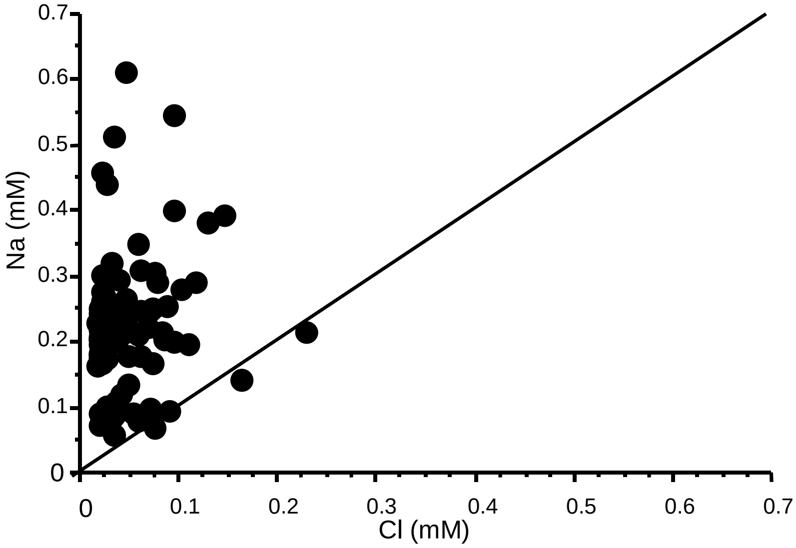


Figure 4-1-3. Scatter plot of chloride vs. sodium, with a 1:1 line for reference. A 1:1 ratio was expected due to the sodium and chloride being applied together in a molar ratio of 1 (NaCl).

DISCUSSION

The Cascade Lakes Roadside soil contained more sand, was denser, and was less permeable than the Cascade Lakes Upslope, Farside, and Chapel Pond Roadside samples. The Cascade Lakes soil was comparatively low in organic matter, and had a low cation exchange capacity. The Cascade Lakes Roadside soil was drier than other sites. The pH was higher than Upslope, Farside, and Chapel Pond Roadside; these three sites had organic matter derived from conifers, which acidifies soil.

The Cascade Lakes Roadside soil had elevated concentrations of sodium, and was depleted of magnesium, calcium and potassium. Whereas the Cascade Lakes Roadside chloride and sodium concentrations appear to differ little from 1985-1986 as reported in Fleck et al. (1988), the nutritive cations were in lower concentrations in 2005. Unfortunately, the results reported in Fleck et al. (1988) and our results are not completely comparable. Fleck et al. (1988) extracted ions using an ammonium acetate solution while we used de-ionized water. Use of ammonium acetate results in a more complete extraction than de-ionized water, but using water to extract ions should provide a better estimate of ions that are available to plants or for transport during rain events.

Taken altogether, the results indicate that the Cascade Lakes Roadside soil has been severely altered in physical structure and chemical composition due to heavy use of chemical deicers and abrasives. Soil at the Cascade Lakes Roadside is composed primarily of sand, while the Cascade Lakes Upslope and Farside soils are composed of dark, organic silts and clays. Chapel Pond Roadside has some sections with each characteristic. Use of sand as an abrasive has changed the composition of the soil, altering density and porosity and decreasing the concentration of fine clay particles, whose charge structure results in the retention of nutritive cations in forest soils. Soils with smaller particles have more cation exchange sites, which means that they can retain more cations than soils that are predominately sand. In some test bores at Cascade Lakes Roadside, the soil appeared to be nearly pure sand to a depth of at least 1.5 m. The accumulation of sand results in a more droughty and infertile soil.

Sodium concentrations are elevated at the Cascade Lakes Roadside, caused by the historically heavy applications of sodium chloride along Route 73, and nutritive cations are depleted. The decrease in calcium, magnesium and potassium could be due to direct competition with sodium for cation exchange sites, and a change in the physical soil structure. In soil systems, calcium and magnesium are more strongly attracted to the cation exchange sites of the soil since they have a higher valence than sodium. However, if the sodium concentration in the soil is high enough, as appears to be true in our study, the sodium can out-compete both calcium and magnesium for the cation exchange sites (Howard and Beck 1993, Norrström and Bergstedt 2001, Löfgren 2001; Bryson and Barker 2002). If calcium, magnesium and potassium are not bound to the soil they can be transported away by flow of groundwater.

The low concentration of organic matter in the Cascade Lakes Roadside soil is likely to be caused by the decline in vegetation cover along roadside, and this an indirect effect of winter road maintenance. Loss of organic matter, like loss of clay particles, results in a lower cation exchange capacity, and thus a more infertile soil. It also indicates a loss of nutrients that would otherwise be released by the decomposition of organic matter.

Despite historically heavy applications of sodium chloride along Route 73, chloride levels differ little among sites, and have not increased over the last two decades. This seemingly paradoxical result is due to the negative charge on the anion and the negative charge of soil particles. Chloride does not remain in the soil; it is flushed into the Cascade lakes during precipitation and snow melt events. Soil particles tend to have a negative charge, and since two negatively charged particles repel each other, chloride does not bind to the soil particles but is transported away, in this case into the Cascade Lakes. At present, the level of chloride in the soil (1.5 ± 0.2 ppm) is below the concentrations that cause direct damage to plants from root exposure (see Chapter 1).

Whereas the Cascade Lakes Roadside soils are the most dramatically altered by winter road maintenance activities, Chapel Pond Roadside soil also appears to have been altered somewhat by abrasives and chemical deicers. Chapel Pond soil is quite variable – some plots appeared to be as altered as the Cascade Lakes Roadside soil, whereas others appeared to be closer in physical and chemical structure to Cascade Lakes Farside soils. Even the Upslope soil has elevated sodium as compared to the Farside soil, which is likely to be causesed caused by aerial deposition of sodium chloride originating from road splash (see Section 4-3).

In summary, the Cascade Lakes Roadside soil has been dramatically degraded, and the Chapel Ponds Roadside soil has been somewhat degraded, due to abrasives and chemical deicers used in control of snow and ice. The Cascade Lakes Roadside soils are physically and chemically altered to the extent that it is reasonable to infer that roadside vegetation will be negatively affected.

An important unknown is how soil chemistry has been altered in other ways that measured in our study. Nitrogen and phosphorus, which are often limiting in forest systems, were not directly measured in our study. Given the chemical and physical structure of the roadside soil, it is likely that these are depleted, similar to the nutritive cations and soil organic matter. Greater uncertainty remains about heavy metals such as aluminum. Heavy metal mobilization occurs with the use of deicing agents (Armhein et al. 1992; Granato et al. 1993). This can affect the health of roadside plants and, in the case of the Cascade Lakes and Chapel Pond, result in export of heavy metals into the lakes.

The other uncertainty pertains to changes in soil microorganisms, including bacteria, protozoa and fungi. The changes in the physical and chemical properties of the roadside soils we have documented are likely to result in an altered community of microorganisms, including some that are important for nutrient recycling or forming mutualistic associations with trees and other plants.

Section 4-2. **PAPER BIRCH (*BETULA PAPYRIFERA)* AND OTHER VEGETATION**

INTRODUCTION

An apparent die-off of paper birch along New York State Route 73 has been of concern to the public since the early 1980’s (Fleck et al. 1988) and remains an issue of concern to environmental advocacy organizations (e.g. Meyers 2002, Adirondack Council 2003). Three major questions need to be addressed about the mass mortality of paper birch and decline of road cover along the highway: (1) *Is the roadside decline extraordinary, or are dead birch along the road simply indicative of a widespread natural stand decline?* (2*) If the decline is caused by highway activities such as winter road mortality, what mechanisms are causing tree mortality and a decline in vegetation cover?* (3) *What can be done, if anything, to halt and reverse the decline in vegetation cover?*

METHODS

Paper Birch Tree Abundance

In each of the 31 plots at the Cascade Lakes Roadside, Upslope, and Farside, and Chapel Pond Roadside, all *Betula papyrifera* (paper birch trees) were tagged using metallic tree tags. The diameter at breast height (dbh, measured on the trunk at 1.6 m above the ground) of each paper birch tree in all of the plots was measured in October 2003 using standard tree calipers (April 2005 for Upslope). Only birch trees with a diameter greater than 3.0 cm were measured and included in the census.

The condition of every paper birch tree was also visually assessed based on the percentage of tree crown cover in October 2003 and again in July 2004 (October 2005 for Upslope). To quantify condition, the tree crown was visually inspected, and a scored based on the percentage of the crown with healthy foliage (Table 4-2-1).

Table 4-2-1. Tree health scoring system used. Adapted from Fleck et al. (1988).

Description	Percent of Remaining Crown
Healthy or no apparent evidence of stress	100%
Slightly stressed or a loss of 1/3 of crown cover	66%
Severely stressed or a loss of 2/3 of crown cover	33%
Dead or no crown cover	0%

Tree Ring Analysis

In July 2004 (April 2005 for Upslope), a tree core was taken from one randomly-selected tree in each of the twenty-six plots in order to determine the growth rates of the trees at each location and compare them to Fleck et al. (1988). Some plots did not contain any paper birch trees, so no core could be taken. If the plot only contained standing dead trees, a core was still taken from one moribund tree. Trees were cored at 1.5 m above ground, using a Haglof tree borer, and were placed in small plastic bags. The tree cores were air-dried for one week, placed in plastic bags, and stored at room temperature.

In October 2005, the cores were sanded and imaged under magnification. High resolution digital images of each core were taken. Incremental growth was estimated by measuring inter-ring distance using J-Image software (application source: US National Institutes of Health).

Up to twenty rings were measured per tree core. Because growth is not a constant (growth decelerates with age), a regression was done using age as the independent variable and interring interval (growth increment) as the y-axis. The slope of the regression provided an estimate of the rate of deceleration in incremental growth, and the intercept provided an estimate of expected incremental growth at the time of sampling.

Birch Recruitment

In July 2004 (October 2005 for Upslope), abundance of seedlings was quantified. Either a 1.0 m^2 or 0.25 m^2 wire square frame was tossed onto a random section of the plot. All *B. papyrifera* tree seedlings under 0.5 m tall within the frame were tallied. This procedure was repeated a second time and then averaged.

RESULTS

The density of paper birch (over 3.0 cm diameter) was much higher in 1985 than 2005 at the Cascade Lakes Roadside and Upslope sites, and the average size of birch was larger in 2003-2005 than 1985 (Table 4-2-2). The size structure of birch trees shifted from 1985 to 2003-2005 to the larger size classes, and the Cascade Lakes Roadside only had large paper birch remaining in 2003-2005 (Table 4-2-3).

The density of birch on the roadside location was dramatically lower at the Cascade Lakes Roadside than the other Cascade Lake sites or Chapel Pond, and there was virtually no recruitment of sapling birch trees at the Cascade Lakes Roadside, unlike the other Cascade Lakes sites (Table 4-2-2). At all sites, a high proportion of standing birch trees was classified as dead, and many live trees were under significant stress (as indicated by Fleck et al. (1988) canopy condition index, Table 4-2-2).

In 2003, among the fourteen 20 x 5 m plots established along the Cascade Lakes Roadside, there were only two live birch trees greater than 3.0 cm diameter, one classified as ‘healthy’ and one as ‘slightly stressed’. By 2004, both birches had died.

There were no significant differences among sites in terms of recent incremental growth rate, as judged by the tree ring measurements, or in terms of the decline in incremental growth rate with age (Table 4-2-4).

Table 4-2-1. Summary of the birch tree abundance and health among the four sites.

	Cascade Lakes					Chapel Pond
	Roadside		Upslope		Farside	Roadside
	1985	2003 - 2005	1985	2003 - 2005	2003 - 2005	2003 - 2005
Tree Density	816	7 ± 4.8	945	378 ± 87.0	227 ± 48.4	209 ± 79.3
Sapling Density		0 ± 0.0		1.8 ± 0.91	12.6 ± 3.92	
DBH	14.4 ± 1.47	29.6 ± 8.17	13.8 ± 0.71	17.6 ± 1.10	21.4 ± 0.93	19.3 ± 1.17
% Alive	45	66	90	50	68	66
% Healthy	62	50	62	62	72	100
N	10 / 220	14 / 3	6 / 124	5 / 39	7 / 46	5 / 33

Tree Density = paper birch ≥ 3.0 cm diameter per ha
Sampling Density = stems per m^2
DBH = Stem diameter at breast height (cm)
% Alive = Percent of standing trees not deceased
% Healthy = Percent of birch crowns without sign of stress.
N = number of plots sampled / number of trees measured

Table 4-2-3. The proportional size distribution of paper birch at the four sites studied in 2003-2005 and two sites in 1985. Size estimated by diameter at breast height (cm).

			Tree Diameter				
	Site	Year	3.8 - 8.8	8.9 - 13.9	14.0 - 19.0	19.1 - 23.9	> 24.0
Cascade Lakes	**Roadside**	**1985**	0.29	0.24	0.33	0.10	0.05
		2003	0.00	0.00	0.00	0.67	0.33
	Upslope	**1985**	0.18	0.32	0.27	0.14	0.09
		2005	0.11	0.16	0.37	0.29	0.08
	Farside	**2003**	0.00	0.07	0.39	0.17	0.37
Chapel	**Roadside**	**2003**	0.06	0.12	0.33	0.24	0.24

Table 4-2-4. Incremental growth (mm) as estimated by tree rings. The slope represents the change in growth from the present to the past (younger trees grow faster), and the intercept is an estimate of the expected present incremental growth

		Slope	Intercept	*N*
Cascade Lakes	**Roadside**	0.015 ± 0.016	0.547 ± 0.160	5
	Upslope	0.021 ± 0.012	0.408 ± 0.070	5
	Farside	- 0.003 ± 0.018	0.731 ± 0.181	7
Chapel	**Roadside**	0.010 ± 0.010	0.594 ± 0.134	6

DISCUSSION

The results indicate that paper birch have been generally in decline in the Cascade Lakes region. This is not surprising, since based on the stand age estimates in Fleck et al. (1988), the Upslope paper birch stands (and presumably all of the valley) established around 90 years ago. Paper birch is a relatively short-lived pioneer species, and is replaced after a century by other species. At the Cascade Lakes, paper birch trees have been damaged by ice storms, rock-slides, and fungal and insect pathogens (Fleck et al. 1988, personal observations). We suspect that the apparent high mortality of paper birch in the Cascade Lakes region and Chapel Pond are the consequence of naturally high mortality occurring in a senescent stand situated in a tough environment. We observe that throughout the watershed of the Cascade Lakes and at Chapel Pond, paper birch is being replaced by boreal conifer species.

Nevertheless, a disproportionately higher loss of paper birch has occurred along the Cascade Lakes Roadside than elsewhere. Remaining are only a few large, old trees. Moreover, unlike other sites, there is virtually no recruitment of paper birch or other tree species on the Cascade Lakes Roadside.

What is the cause of the decline paper birch along Route 73 in the region of the Cascade Lakes? Sodium and chloride levels appear to be too low to be directly toxic via root uptake alone. The sodium and chloride levels found in the soil (5.3 ± 0.2 ppm sodium and 1.7 ± 0.2 ppm chloride) were below the EC_{25}, effective concentration to damage 25% of the vegetation, which is 67.5-300 ppm sodium and 215-1500 ppm chloride (Canadian Environmental Protection Act 1999).

We have inferred three causes of decline, besides natural stand senescence or toxicity from root uptake:

- Direct damage from road construction and maintenance activities (e.g. installation of rock walls, pushing rock debris off the road surface).
- Salt accumulation in plant tissues and disruption of water balance.
  - Accumulation of aerially transported road salt onto branches other plant surfaces.
- Changes in soil chemistry and structure.
  - Nutritive cations having been leached from the soil, replaced by sodium.

- Changes in soil structure reducing water retention and thus potentially increasing water stress.

Activities to stabilize the steep Route 73 roadside bank appear to have resulted in damage to some trees (personal observation). It seems likely that some trees have been injured by fallen rocks that have been removed from the road and discarded. Judging from the remaining standing dead birch trees along the highway, however, it does not appear likely that direct damage was a principal cause of the roadside birch decline.

Salt damage caused by aerial deposition from road spray is likely to be a major source of physiological stress to roadside birch trees, and may affect trees at some distance from the road (see Section 4-3). The EC_{25} threshold limits for aerial exposure to sodium (200-16100 ppm) and chloride (800-70700 ppm) plausibly may be surpassed along the highway. Although soil concentrations of sodium and chloride appear to be too low to be toxic, aerial exposure and root uptake may act synergistically in causing stress to the trees. Noxious health effects of sodium and chloride uptake through the roots and by aerial deposition include reduced leaf size, reduced incremental growth, leaf chlorosis, desiccation, and tissue death, including leaf death and bud death (Canadian Environmental Protection Act 1999).

A likely major contributing factor to mature tree mortality and recruitment failure is the decline in soil fertility caused by changes in soil structure and chemistry due to winter road maintenance. Birch trees require a nutrient rich soil with high organic content. Throughout the Cascade Lakes Roadside, and some plots along Chapel Pond Roadside, the soil organic content and concentrations of nutritive cations was so low that tree health was probably impaired and seed germination and seedling growth impaired. Seed germination and seedling growth may also have been reduced by the lower penetrability of the soil surface and lower water availability than found at less impacted stands of birch (e.g. Cascade Lakes Farside).

In conclusion, there is likely to be no single cause of the die-back of birch and of other vegetation along Route 73. Changes in the soil structure and chemistry, and direct toxic affects of sodium and chloride may act synergistically to weaken trees and prevent regeneration. This is likely to be particularly apparent where natural conditions are stressful, as they are in the Cascade Lakes region.

Section 4-3. **AERIAL TRANSPORT OF DEICING ROAD SALT**

INTRODUCTION

Aerial transport of deicing road salt may result in significant deposition on vegetation and soil a distance from a road surface. This can result in stress and damage to plants at distances of over 50 m (Lumis et al 1973, Kelsey and Hootman 1992, Cain et al 2001, Paludan-Müller 2002; see Chapter 1).

Plowing and road splash can result in significant chemical deicer accumulation in roadside snow banks, which are released at snowmelt (Oberts 1991). This can result in a very high acute exposure to chemical deicer during the episodes of snowmelt, when the accumulated burden of contaminant is released over a short time period (Labadia and Buttle 1996, Environment Canada and Health Canada 1999).

As a preliminary estimate of aerial transport of deicing road salt and accumulated salt burden in roadside snow piles, we measured chloride levels in snow at four transects at the Cascade Lakes, sampling at points that correspond to those sampled for the soil and vegetation studies.

METHODS

On 19 January 2005, snow was collected by Michael Twiss and Joseph Osso Jr. at four sites along each of four parallel transects perpendicular to the Cascade Lakes gorge (Fig. 4-3-1). The georeferenced sampling sites along the four transects included an Upslope (Pitchoff Mountain side) location, North and South Roadside locations, and a Farside (Cascade Mountain side) location. The Upslope and Farside locations were selected to be approximately 60-150 m from the roadway and 1 m from the lakeshore, respectively (the exception was the most southerly transect, where the Farside sample location was near the trailhead for Cascade Mountain; Fig. 4-3-1, A). The roadside locations were selected to be 2-3 m from the roadway.

At each site, 250 mL of snow was collected from the surface to a depth of approximately 3 cm. Snow was collected in tared polypropylene sample cups; the mass of snow was 20-40 g. Snow pack varied from light snow in the forested Upslope sites to a dense drifted snow on the Farside (lake shore) sites. The snow was allowed to melt upon return to the laboratory, and the chloride content was measured using an ion selective electrode (see Chapter 5-1 for methodology). Chloride content is expressed as mg Cl per kg of snow (ppm).

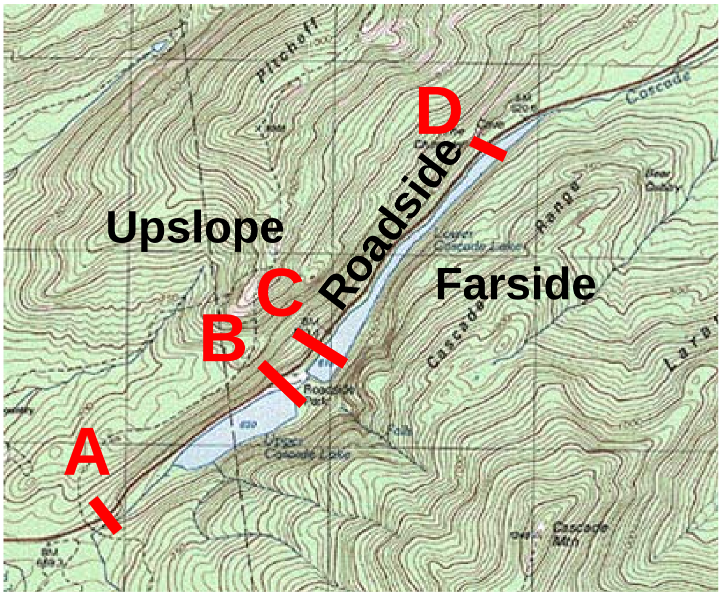


Figure 4-3-1. Location of the four snow-sampling transects at the Cascade Lakes.

RESULTS

Significant aerial deposition of sodium chloride was detected on plant parts at distances from the road (Fig. 4-3-2). Appreciable chloride concentrations were measured at all sampled sites, indicating that there is measurable aerial transport of sodium chloride away from the road to the forests upslope and across the lake (Table 4-3-1). The roadside samples, as expected, had extremely high concentrations of chloride (Table 4-3-1).

Table 4-3-1. Chloride levels (ppm) from snow grab-samples collected on 19 January 2005. Transects are as indicated in Fig. 4-3-1.

		Roadside		
Transect	**Upslope**	**North**	**South**	**Farside**
A	70	2370	475	34
B	43	930	35300	623
C	4	633	907	492
D	310	3840	2140	125

Appreciable salt deposition on trees was observed visually on this date (Fig. 4-3-2). The location of this sampling site (Transect C, Upslope) strongly indicates that salt can be transported at a great distance away from the roadway surface

DISCUSSION

These results indicate that aerial transport of chemical deicers may result in exposure of vegetation to road salt throughout the Cascade Lakes gorge. Acute exposure to sodium chloride may be high away from Route 73 during the snowmelt period in spring. At the roadsides, acute exposure of plants and soil organisms may be very high – the highest measured roadside snow grab (35,000 ppm chloride) was twice the level of seawater (15,000 ppm chloride), and surpassed the range of measured values for snow dumps (2000 – 10,000 mg/L) presented in Environment Canada and Health Canada (1999). Wet deposition from road splash as briny as was recorded in our study is likely to result in tissue death to roadside vegetation (Cain et al. 2001).

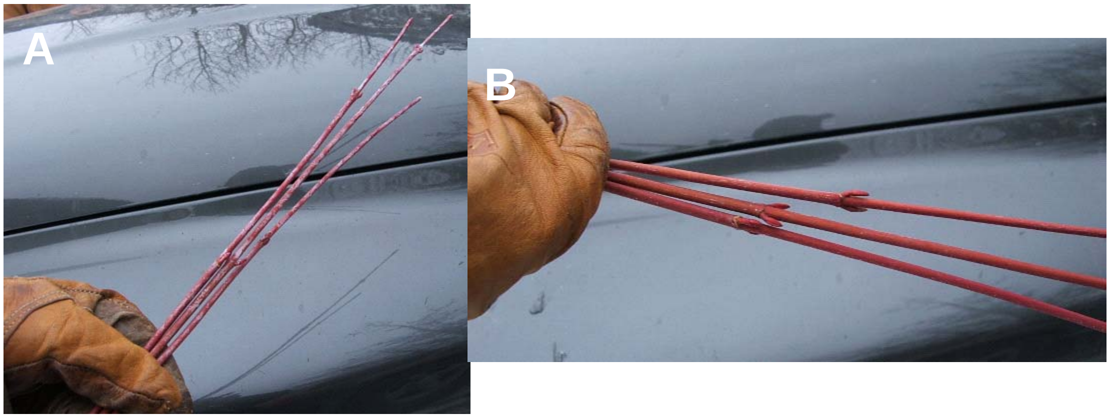


Fig. 4-3-2. Branches of red-osier dogwood (Cornus sericea) collected on 19 January 2005 on the Upslope section of Upper Cascade Lake, about 20m elevation above and 100m distance from the road surface. Image A shows the side facing the road surface (white discoloration is salt deposition), Image B shows the same branches from the side facing away (the northeastern exposure). Photos by Michael Twiss.

CHAPTER 5

LIMNOLOGICAL INVESTIGATION OF THE CASCADE LAKES AND CHAPEL POND

2003-2005

Rough early-winter conditions on Upper Cascade Lake, November 2003. Photo by Michael Twiss.

Section 5-1. **PALEOLIMNOLOGICAL INVESTIGATIONS OF THE CASCADE LAKES AND CHAPEL POND**

Report prepared by J. Curt Stager.

INTRODUCTION

Sediment cores were collected from deep-water sites in Upper Cascade Lake, Lower Cascade Lake, and Chapel Pond (Adirondack Park, NY) in order to reconstruct long-term records of the possible effects of road salt and sand contamination in the lakes. The siliceous valves of diatoms (single-celled algae) in the cores were used to infer past water chemistry conditions with the aid of ecological transfer functions, and organic content of the sediments was examined in order to detect inputs of inorganic materials associated with road construction and management. This study was conducted in order to extend the observational records of the lakes back to the 19th century, in conjunction with ongoing investigations of current water chemistry.

METHODS

Gravity cores were collected through the ice in the deep basins of each lake during the winter of 2004. Each core was extruded vertically in 1 centimeter increments in the laboratory, and subsamples were stored under refrigeration in WhirlPak^{TM} bags.

Radioisotopic chronologies for the three cores were determined by subcontractor Brian Cumming (Queen's University, Ontario) and MYCORE Scientific (Ontario), using both ^{210}Pb and ^{137}Cs concentrations versus depth. For the ^{210}Pb chronology, the Constant Rate of Supply method was used to calculate sediment ages (Binford 1990).

Organic content of sediments was estimated by combustion of dried samples at 500 degrees C (%LOI = percent weight loss on ignition). Reductions in %LOI were considered to be likely indicators of dilution of organic lake muds by inorganic terrestrial materials.

Diatom assemblages were analyzed after digestion in H_2O_2 and mounting on slides with Permount medium. Identifications were made to the species or variety level, and 350-450 valves were counted per slide at 1000X under oil immersion, using standard taxonomic references. Transfer functions were applied to the diatom data by subcontractor Brian Cumming in order to reconstruct epilimnetic chloride, using weighted averaging regressions on limnological and diatom data from the EMAP database, which includes 240 lakes in the Adirondacks and similar habitats in eastern North America (Dixit and Smol, 1994). Samples of sediments from tributaries and littoral habitats were also collected to aid in the interpretation of the diatom records.

RESULTS

Geochemistry

The radioisotopic profile of the Upper Cascade Lake core (31 cm long, from 18 m lake depth) displayed a smooth decline of ^{210}Pb activity with depth and a clear peak of ^{137}Cs activity marking the ca. 1963 AD horizon (global maximum of atmospheric nuclear testing; Figure 1). The bottom of the core (31 cm depth) was beyond the range of ^{210}Pb -dating, and was estimated to have been deposited in the early 1700's AD. Water content and organic content (%LOI: Figure 1) increased sharply above the 26 cm interval (ca. 1770's AD); the low values in gritty, light-colored deposits at the base of the core may indicate inputs from a landslide or flood event in this lake during the 18th century. The abundance of siliceous remains of testate amoebae in the lower samples (plates:diatoms, Fig. 5-1-1) supports this interpretation because testate amoebae are common in moist habitats such as those found on the steep slopes and cliffs along the lake shore. LOI also decreased noticeably around the 14 cm interval as well.

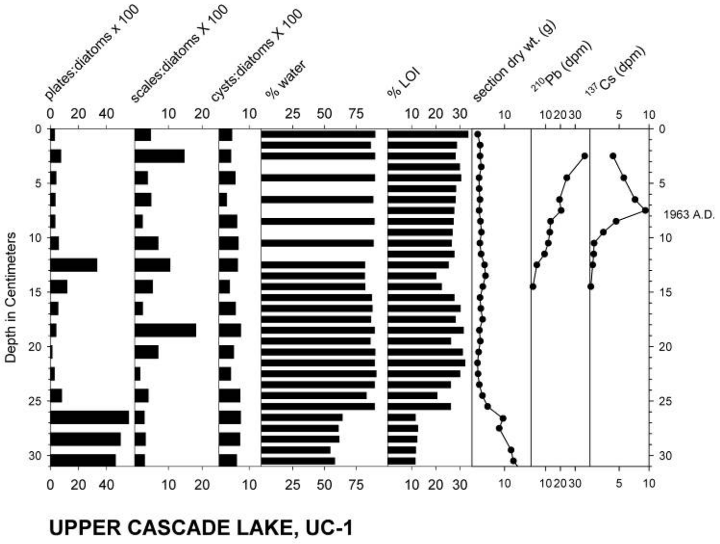


Figure 5-1-1. Microfossils, physical characteristics, and radioisotope profiles in the Upper Cascade Lake core UC-1.

In the Lower Cascade core (39 cm long, from 13 m lake depth), the ^{210}Pb profile was also relatively smooth and there was a clear peak in ^{137}Cs activity (ca. 1963 AD) centered on the 15 cm interval (Fig. 5-1-2). The bottom sediment sample, at 38-29 cm depth, was calculated to have been deposited ca.1828 AD. There was no evidence of major inorganic sediment inputs like that found in the UC-1 core, except for a large number of amoeba plates in the bottom sample. This finding contradicts local reports that the two Cascade Lakes were separated by a landslide during the mid-1800's; if such an event occurred, then Lower Cascade Lake should have registered a major drop in LOI in the same time interval that Upper Cascade did. There was, however, a moderate decrease in %LOI in the LCL-1 core between the 33 cm and 24 cm intervals (1870's-1930's AD).

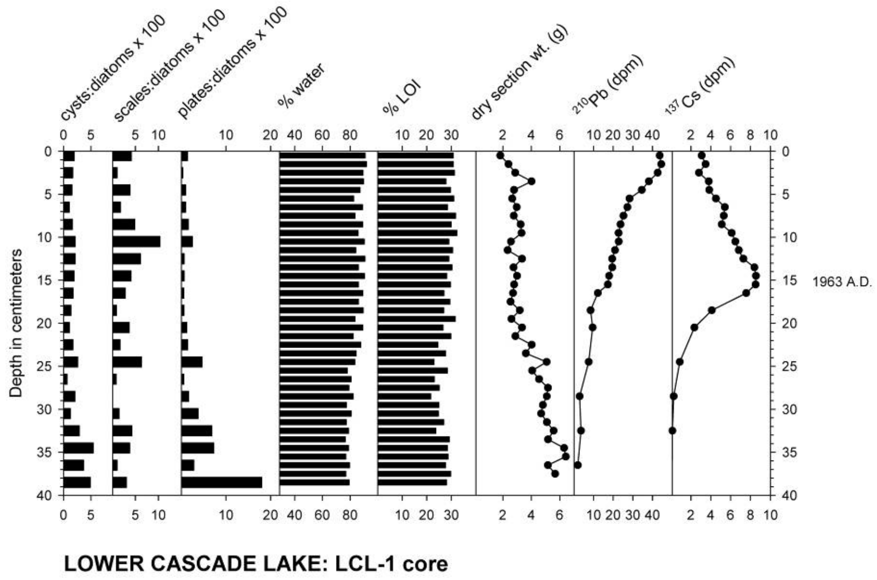


Figure 5-1-2. Microfossils, physical characteristics, and radioisotope profiles in the Lower Cascade Lake core LCL-1.

Dating the Chapel Pond core (21 cm long, from 7 m lake depth) was more problematic, because the ^{210}Pb profile was not consistent with the position of the ^{137}Cs peak in the CP-4 core (Fig. 5-1-3). The ^{210}Pb decrease with depth was more erratic in this core, and the ^{137}Cs peak was less distinct, possibly indicating disturbance of the sediment column. The sediment record was more compressed than those in the Cascade Lake cores, presumably because sediment accumulates more rapidly in the latter, and confidence in the accuracy of the chronology of this core is not as strong as for the other two. There was little change in water content and organic content, but there was a slight decrease in %LOI in the 13 cm sample (ca. 1800 AD), and a trend of decreasing LOI in the upper 6 cm of the core (the last ca. 50 years). This core was situated at the north end of the lake rather than from the center, in order to avoid sand deposition from sporadic flooding events from the southern inlet stream which deposits sediment and woody debris on the ice, but occasional spikes in the abundance of amoeba plates may reflect runoff from the high cliffs adjacent to the coring site. There was a marked rise in the abundance of siliceous scales of chrysophyte algae between the 13 and 5 cm intervals which represents a shift in the structure of phytoplankton communities during the 19th century and early 20th century, but the reason for that change is unclear at present.

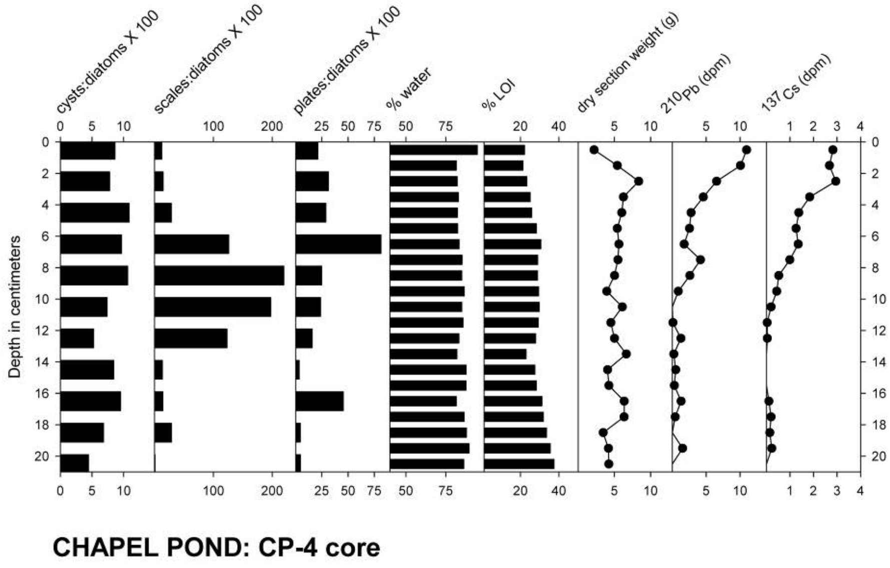


Figure 5-1-3. Microfossils, physical characteristics, and radioisotope profiles in the Chapel Pond core CP-4.

Plotting the physical characteristics of the three cores versus time yielded evidence of short-lived impacts of human activity in the watershed.

In the Upper Cascade core, there was a pronounced increase in sediment accumulation rates in the 1930's AD which probably represents inorganic inputs during construction of the lakeshore highway, and a smaller peak around 1980 AD (Fig. 5-1-4: top).

A large increase also occurred in the 1930's interval in the Lower Cascade core, with increasing deposition rates overall after ca. 1960 AD.

There was a slight increase in accumulation rates in the CP-4 core in the mid-1900's, but uncertainty in the chronology of this core does not permit definitive assignment to the 1930's interval. Accumulation rates were higher in Chapel Pond in the most recent decades as well.

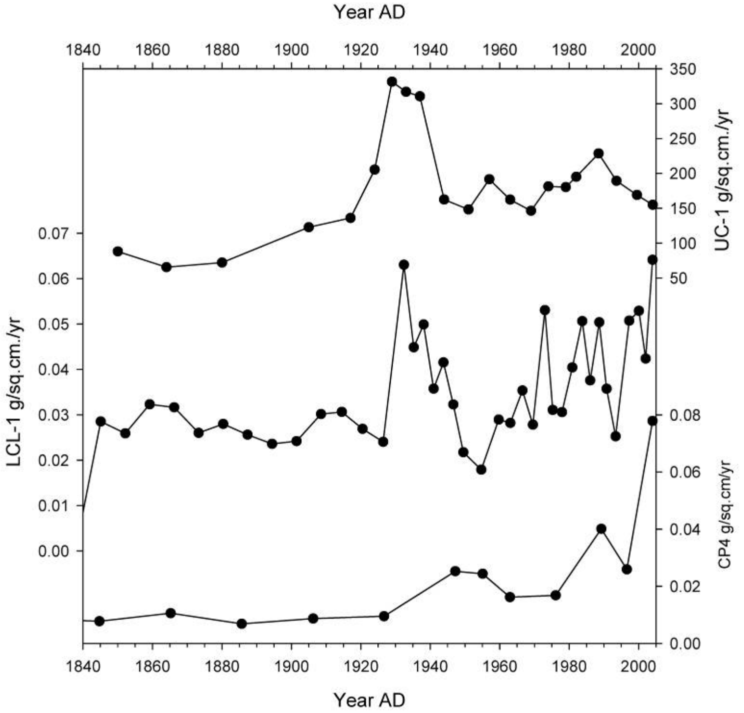


Figure 5-1-4. Sediment accumulation rates (grams/square cm/year) in the Upper Cascade, Lower Cascade, and Chapel Pond cores (UC-1, LCL-1, and CP-4, respectively).

Interpretation of the accumulation rate spikes of the 1930's as representing sand and silt inputs related to road construction is supported by the %LOI profiles as well (Fig. 5-1-5).

Aside from the apparent slide deposit of the 18th century, the lowest %LOI of the Upper Cascade record occurred in the 1930's AD interval. A smaller decline occurred around ca. 1880 AD as well. The sediment accumulation increase of the ca.1980 AD interval was accompanied by a rise in %LOI, which suggests that organic materials rather than sands account for that increase.

In the Lower Cascade core, a broad interval of reduced %LOI spanned the ca. 1880-1930's AD period. This is perhaps best explained by the construction of Cascade Lake House (built 1880 AD, operated until 1915 AD), subsequent land use activities related to it, and road construction in the 1930's. It is not surprising to find evidence of more sustained impacts of these activities in the Lower Cascade Lake sediment record than in that of Upper Cascade because the former lake lies downstream of the hotel grounds.

The Chapel Pond core registered a trend of declining %LOI in the last 40 years or so. Although small amounts of gritty sand could be felt by hand in most of the samples from the Lower Cascade core, none of the cores appear to indicate large inputs of road-derived sands to the coring sites.

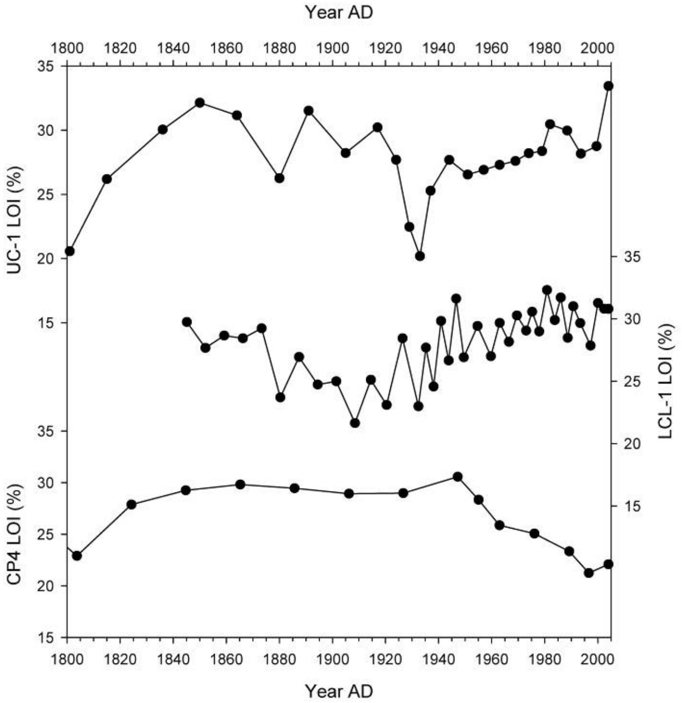


Figure 5-1-5. Organic content (%LOI) of the three cores.

Diatom Records

All three cores contained rich and diverse assemblages of both benthic and planktonic diatom taxa. Because abundant stream inputs from the adjacent cliffs were a likely source of benthic taxa, such as *Achnanthes minutissima* and *Tabellaria flocculosa* IV, that are also found in the lakes themselves, greater emphasis was placed upon the lacustrine planktonic forms here, and only planktonic forms were used in the water chemistry reconstructions. This approach may have limited the accuracy of those reconstructions somewhat.

In the Upper Cascade core, a diverse assortment of benthic diatoms (miscellaneous Pennales) included members of the genera *Achanthes, Eunotia, Navicula, Pinnularia*, and others. The dominant planktonic diatoms were *Cycotella bodanica, Asterionella formosa,* and *Tabellaria flocculosa IIIP* (Figure 5-1-6), which are common in circumneutral, moderately productive lakes in the Adirondack uplands. Above the 10 cm interval (ca. 1950 AD), *Aulacoseira subarctica, A.*

ambigua, and *Synedra* sp. became relatively more abundant, largely at the expense of *C. bodanica* percentages.

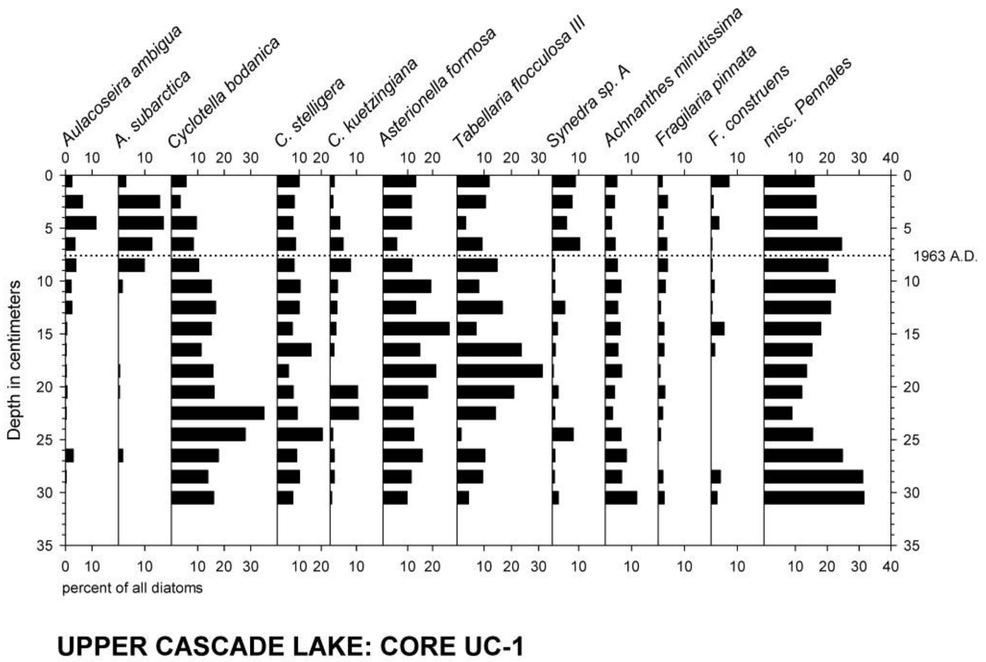


***Figure** 5-1-6. Percentages of the major diatom taxa in the Upper Cascade lake core UC-1.*

In the Lower Cascade core, benthic diatoms were common in all samples (Fig. 5-1-7). The dominant planktonic taxa in the lower part of the core (below ca. 28 cm depth; ca. 1908 AD) were *Cyclotella bodanica, C. stelligera,* and *Tabellaria flocculosa*, *Aulacoseira ambigua* and *A. subarctica* were dominant in the upper ca. 28 cm, and benthic *Fragilaria construens* became abundant above the ca. 20 cm level (ca. 1940 AD).

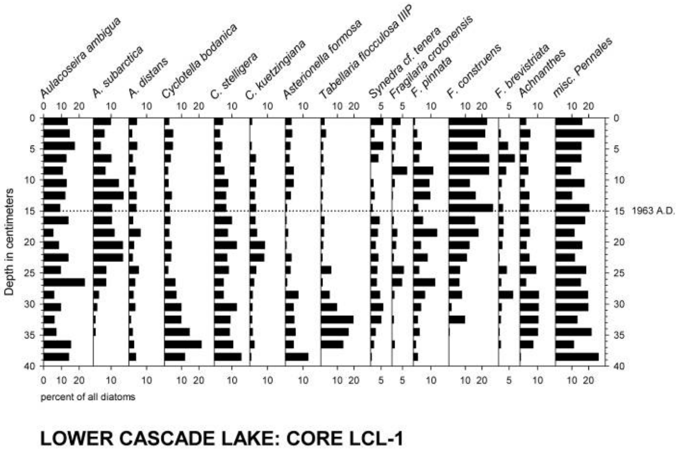


***Figure** 5-1-7. Percentages of the major diatom taxa in the Lower Cascade lake core LCL-1.*

In the Chapel Pond core, benthic diatoms were by far the most abundant forms (Fig. 5-1-8). Below ca. 5 cm depth (ca. 1960 AD?), planktonic assemblages were dominated by *Aulacoseira lirata*, and *A. distans* dominated above the 5 cm level. The diatom community of this lake appear to represent less productive conditions than are found in the Cascade lakes.

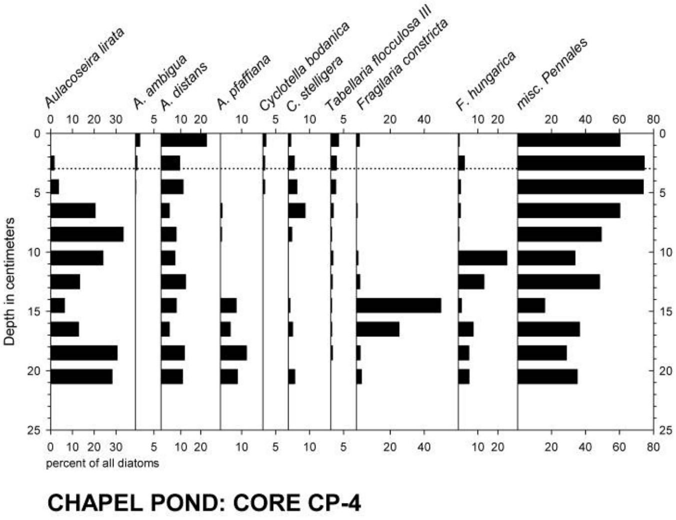


Figure 5-1-8. Percentages of the major diatom taxa in the Chapel Pond core CP-4.

Water chemistry reconstructions.

The diatom records of the three cores were converted to spring-fall epilimnetic chloride concentrations by Brian Cumming, using only the planktonic taxa in the assemblages in order to avoid errors due to mixing of lacustrine and stream diatoms in the sediments (Fig. 5-1-9, left three panels).

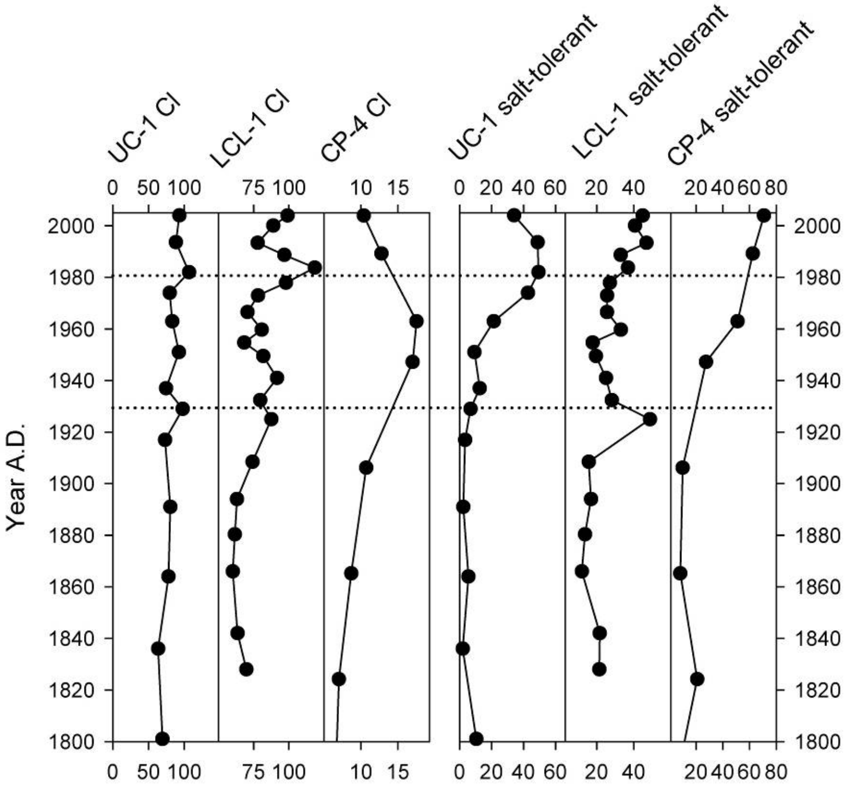


Figure 5-1-9. Inferred chloride (mg/L) and percentages of salt-tolerant planktonic diatoms in the three cores.

The inferred chloride profiles displayed moderate, generally increasing concentrations over the last 150 years. For Upper Cascade Lake, Cl values ranged from 60 to 80 mg/L for most of the record and rose to 90-120 mg/L above the ca. 1980 AD interval. In the Lower Cascade core, Cl fluctuated more, ranging from ca. 60-80 mg/L until the 1930's AD, with somewhat higher and more variable concentrations from ca. 1930 to ca. 1980 AD, and sustained higher values of 88-106 mg/L above the ca. 1980 AD interval. At Chapel Pond, Cl levels were almost an order of magnitude lower, with a slight increase from ca. 6-13 mg/L in all but the 1950's-1960's AD interval, where values rose to ca. 17 mg/L.

In addition to the quantitative reconstructions of water chemistry, percentages of the most salt-tolerant planktonic diatoms were also plotted against time (Fig. 5-1-9, right). This was done because the chloride tolerances of some of the species in the diatom assemblages (e.g. *Synedra nanana*) were not contained in the EMAP dataset, thus somewhat limiting the accuracy of the quantitative reconstructions.

In the Upper Cascade core, percentages of salt-tolerant diatoms fluctuated much more dramatically than did the inferred chloride concentrations. A slight increase occurred during the 1930's AD, and percentages increased 4-5 fold after the 1960's AD. In the Lower Cascade core, percentage trends were generally similar to those of inferred chloride, with a first peak ca. 1930 AD and sustained maximal values above the ca. 1980 AD horizon. At Chapel Pond, there was a consistent increase in percentages of salt-tolerant planktonic diatoms throughout the latter half of the 20th century.

DISCUSSION

When interpreting Fig. 5-1-9, it is important to remember that each sediment sample in the time period shown represents 8-9 years (Upper Cascade), 4-5 years (Lower Cascade), or 15-27 years (Chapel Pond) of accumulated sediment. When interpreting paleo records of this nature, which are not only averages of several years but also potentially subject to sporadic sediment disturbances by physical and biological agents, individual data points are far less useful than overall trends. The long-term trends are therefore the principal focus of the environmental interpretations given in this report.

All three of the cores from the Cascade lakes and Chapel Pond contained sediment records covering the last 150 or more years. The radioisotopic records were of variable quality, and assigned ages should be regarded with suitable caution; the accuracy of ^{210}Pb dates decreases significantly as one moves farther back in time, and in the case of the Chapel Pond core, the ^{210}Pb and ^{137}Cs dates did not match particularly well, perhaps due to cesium mobility in the sediments. However, the apparent coincidence of major changes in the sediment accumulation rates and organic content with known disturbances in the lake watersheds lends support to the chronologies presented here.

There were two time periods during which the most significant changes occurred in the physical and inferred Cl records of the cores: the 1930's AD and ca. 1980 AD.

During the 1930's, sediment accumulation rates increased and sediment organic content decreased in the deep basins of the Cascade Lakes. This is almost certainly a reflection of road construction activities on the lakeshore road, which would have mobilized inorganic materials and deposited them in the lakes. A similar disturbance also occurred at Chapel Pond, but to a lesser degree, and the chronology of the event is less precisely dated; the estimated dates put the change there about a decade after it occurred in the Cascade lakes. In each of the three cores, sediment accumulation rates increased overall after the road was improved and paved (1930's AD) and winter sand application commenced, which suggests that road management has increased sediment inputs to the lake bottoms.

In Lower Cascade Lake, increased inorganic sediment inputs were also apparently associated with the operation of the Cascade Lake House as well, producing a sustained minimum in %LOI from the early 1880's AD when the establishment opened to about 1915 AD when it closed. There was no evidence in the Cascade cores for sediment disturbances related to local forest fires ca. 1950 AD

Only in Chapel Pond did sediment organic content decline during the late 20th century, suggesting that terrigenous materials have diluted the lake deposits in recent years. The origin of these materials is not known at present, but they may be related to an increase in the frequency or sediment load of flood events in the inlet stream situated across the lake from the coring site. In the Cascade Lakes, the increase in sediment accumulation rates was accompanied by a rise in organic content, contrary to what would be expected from road sand inputs. The cause of this trend is also unknown at present, but it can reasonably be hypothesized to be the result of either increased algal productivity, the decay of dying vegetation along the roadside, or a reduction in sand and silt after the road was paved.

In the Cascade Lakes, inferred spring-fall epilimnetic chloride concentrations increased after the road was improved in the 1930's AD, and most notably after the 1980 Winter Olympics when salt application was most intense. In Chapel Pond, the inferred chloride reconstruction conflicted with the percentages of salt-tolerant diatoms, the latter suggesting a clear influence of road salt contamination as well. In spite of the aforementioned limitations of the diatom-inferred chloride model and the conflicting results for Chapel Pond, it is nonetheless reasonable to conclude from these results that road salt has significantly raised epilimnetic Cl concentrations in all three of these lakes during the ice-free months of the year when the planktonic diatoms upon which this study was based are most abundant.

Epilimnetic chloride measurements made during this project yield much higher values than those obtained by previous investigators, but it is unclear whether this reflects differences in measurement techniques or a true rise in Cl at the start of the study. The most recent chloride concentrations inferred from the diatom records of the Cascade lakes are very close to those obtained in the field during the past few years (ca. 100 mg/L), but the magnitude of the increases in the inferred trends is not nearly as great as that displayed by the water chemistry measurements. At present, it is not known whether the differences in point measurements reflect true Cl increases, but the relatively small changes in the most recent portions of the paleo records seem to suggest that the apparent increase may be somewhat exaggerated. An additional core (42 cm long) collected from the southern end of Lower Cascade Lake for this study was not analyzed in detail, but the youngest and oldest diatom assemblages from that site showed relatively little change, as did the LCL-1 core from the main basin.

The inferred record of Cl concentrations from the Chapel Pond core overestimates modern chloride measurements (ca. 10-12 vs. ca. 2 mg/L), and the trends in Cl and percentages of salt-tolerant taxa conflict. This core suffers from several unavoidable problems, including a much coarser temporal resolution than the Cascade lake cores (due to naturally slower sediment accumulation rates), inconsistency in the radioisotopic chronology mentioned earlier, and limitations in the precision of the chloride reconstruction model, as well as a shortage of water chemistry data for comparison to present conditions.

CONCLUSIONS

1. The cores collected from the Cascade lakes and Chapel Pond represent the last 150 years or more of sedimentary record. The portions of the cores representing the last 50 years are sampled at 1 cm intervals representing the combined average of ca. 8-9 years (Upper Cascade), 4-5 years (Lower Cascade), and 15-27 years (Chapel Pond) of lake history.

2. The radioisotopic chronologies of the Cascade Lake cores appear to be sound, but the Chapel Pond chronology appears to be less reliable.

3. Road improvement in the 1930's AD increased sediment accumulation rates in the Cascade lakes and probably in Chapel Pond as well.

4. There is little evidence of increased sand and silt inputs at the deep basin coring sites in the Cascade lakes, but they do appear to have increased at Chapel Pond, perhaps due to increased tributary stream inputs.

5. Inferred spring-fall epilimnetic chloride concentrations in the Cascade lakes are consistent with the most recent water chemistry measurements. They increased after the lakeshore road was improved in the 1930's and were highest after ca. 1980 AD, suggesting significant inputs of road salt. The Chapel Pond record displays lower inferred Cl levels, but they are higher than measurements made in the field during this study. A recent decline in the inferred Cl concentrations at Chapel Pond also conflicts with rising percentages of salt-tolerant planktonic diatoms in the core.

Section 5-2. LIMNOLOGICAL INVESTIGATION OF THE CASCADE LAKES AND CHAPEL POND WATER QUALITY 2003-2005

Report prepared by Michael Twiss & Joseph Osso Jr.

INTRODUCTION

The objective of the research reported in this section was to determine the extent of road salt impact on the lakes and determine if meromixis is present. Monthly sampling of each lake was carried out over a two year period in Upper Cascade Lake (UCL) and Lower Cascade Lake (LCL), whereas Chapel Pond (CP) was sampled on a seasonal frequency. Each lake was characterized on the basis of profiles of temperature, Cl^-, Na, K, Ca, Mg, chlorophyll-a, and dissolved oxygen, in addition to measurements of zooplankton community composition and abundance. By comparing the data that was collected from the current study and the data recorded by the ALSC in the early 1980s and from other sources, a characterization on the effect of nearly twenty years of road salt application to SR-73 could be made, and the data could also be used in development of a mass – balance model of chloride for the Cascade Lakes.

METHODS

Site Description

The characterization of the study lakes requires an understanding of the bathymetry of each lake and its surrounding area. The shape of each lake provides for distinctly different characteristics despite their proximity to one another. In order to create a more recent dataset of the bathymetry of UCL and LCL, a geographic positioning system (GPSMAP 76S, Garmin International, Olathe, KS) was used in conjunction with sonar depth finder (SeaFinder 320DF, Eagle, Catoosa, OK) to georeference (latitude and longitude) lake depths. The data was converted from the measured WGS 84 coordinate system into a projected UTM WGS Zone 18N coordinate system in ArcTool. This conversion allowed a compilation in ArcMap with the resulting bathymetry maps being produced (Figs. 5-2-1 and 5-2-2, see Chapter 6-1).

CP is located approximately 32 km east of the Cascade Lakes in Keene Valley, NY. This body of water serves as a good control lake because it not only has a comparable shape and depth to UCL, but is also adjacent to highway that is maintained by NYSDOT. The New York State Department of Environmental Conservation (NYSDEC) has constructed its own bathymetric map of CP (Fig. 5-32-3).

The geology of this region is made up of metanorthosite and charnockite. Each of these rock groups is separated by a normal fault running parallel to the north of UCL and LCL. Similar rock types are also found near CP (Isachsen and Fisher 1970).

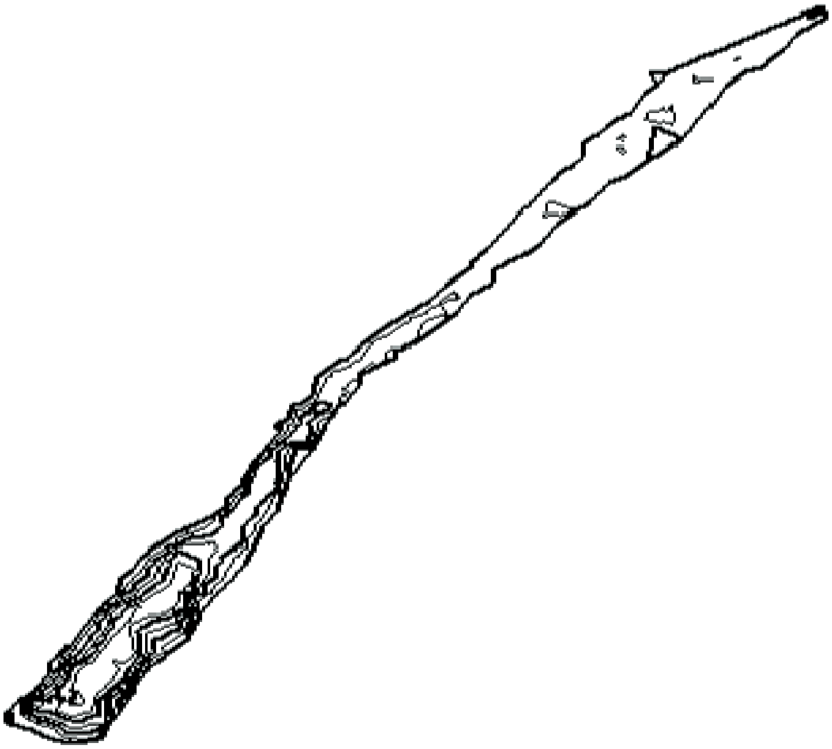


Fig.5-2-1. *Bathymetric map of Lower Cascade Lake depicting lake shape and depth. Each contour interval represents 2.5 m. (see Chapter 6-1).*

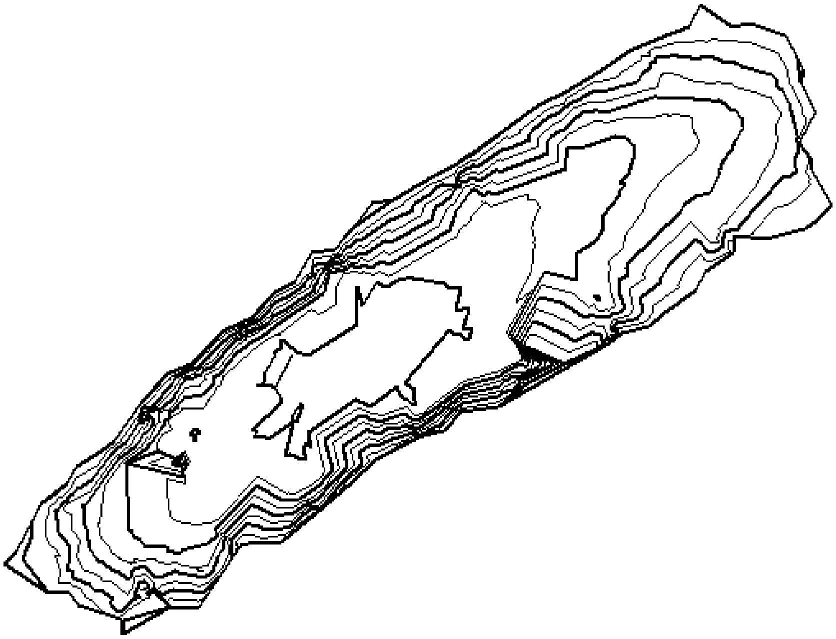


***Fig. 5-2-2**. Bathymetric map of Upper Cascade Lake depicting lake shape and depth. Each contour interval represents 2.5 m. (see Chapter 6-1)*

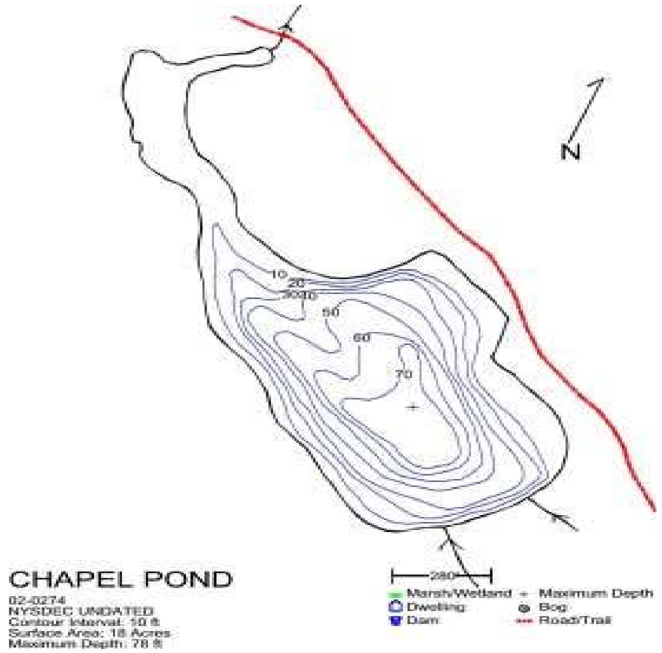


Fig.5- 2-3. Bathymetric map of CP depicting lake shape, depth, and its proximity to SR-73. Each contour interval represents 3.48 m. (data from Adirondack Lake Survey Corporation, see Chapter 3-5).

Water Sampling

The master station at each lake (Table 5-2-1) was located above the deepest point in each lake. This specific point was chosen in order to get a representative sampling of the entire lake from the surface to the bottom. At the master station, a water quality sonde was used to obtain a profile of the lake. This profile included temperature and dissolved oxygen concentration. Based on the thermal stratification, the depth from which two samples from each of the epilimnion, metalimnion, and hypolimnion layers was selected. If there were any standout measurements in oxygen, those depths were given preference. If no thermal stratification existed, then the depths were selected evenly.

During the ice-free season, the master stations were sampled from an anchored aluminum rowboat using a 2.5 L Niskin Bottle or an 8-L Teflon-coated Go-Flo bottle (General Oceanics, Miami, FL) on a nylon line triggered with a Teflon-coated stainless steel messenger. During the winter, sampling was through a hole chopped in the ice (0.3-0.7 m thick). Water samples were collected in 1-L HDPE bottles, cleaned previously using 1% nitric acid, and rinsed exhaustively with de-ionized water.

Table 5-2-1. Table representing the latitude and longitude of the mater stations and the corresponding maximum depth at the master station for each lake.

Lake	Latitude (°N)	Longitude (°W)	Maximum Depth (m)
Upper Cascade	44.22366	73.87838	20
Lower Cascade	44.22622	73.87363	14
Chapel Pond	44.13960	73.74200	24

Sonde Calibration and Preparation

The water profiling sonde used was a model 610-DM Meter with a 600XL Multi-Parameter Water Quality Monitor (YSI Incorporated, Yellow Springs, OH). This instrument measured water temperature, conductivity, and dissolved oxygen concentration.

Prior to each sampling day, the membrane on the dissolved oxygen sensor was replaced. At this time, the exposed electrodes were cleaned using a fine grit paper and rinsed thoroughly with Millipore Ultra Pure Water (Q-water) (Millipore, Billerica, MA). After drying, the reservoir was refilled with silver nitrate solution and a new membrane was applied to the sensor.

Immediately before heading onto the lake, the instrument was calibrated for conductivity and dissolved oxygen. The calibration for conductivity measurement was performed by submerging the sonde bulkhead into a potassium chloride solution of 0.01 M. The conductivity measurements fluctuated greatly between sampling dates and have been discarded, but nevertheless were used as an in-field guide for relative conductivity when comparing each depth on that sampling date.

The dissolved oxygen calibration was accomplished by assuming that the surface water was saturated with oxygen, and basing the concentration of oxygen on the water temperature and a dissolved oxygen saturation chart from Wetzel and Likens (2000). During the winter months, when supersaturation was possible, a small container of Q-water was used and assumed to be saturated with oxygen. (Early months of the study gave dissolved oxygen values that were questionable and this led to a full analysis of the instrument, leading to the discovery of a crack within the sensor. Once replaced for the January 2004 sampling, the dissolved oxygen numbers became more reliable and similar to the values obtained from the Winkler Titrations performed on each sample: see below).

Filtration Procedure

Upon returning from the field, the samples were prepared for analysis by filtration through a Whatman Glass Microfibre TCLP, acid treated/low metal 0.7µm pore size 47 mm diameter filter (Whatman, Middlesex, UK). An electrical pump was used to provide a vacuum through a Teflon filtration rig (Savillex Corp., Minnetonka, MN). This setup was rinsed with dilute nitric acid to minimize Cl- contamination. A volume of 400 mL of the filtered sample was collected in a Teflon bottle and then transferred into two-125 mL polypropylene bottles. An aliquot of the remaining sample was used for pH measurement. One of these bottles was stored in a refrigerator, to be used for chemical analysis, and the other was stored in a freezer, to serve as an archived sample. The lab pH was determined by using a Thermo Orion model 620 pH electrode (Thermo Orion, Waltham, MA), which was calibrated using a two-point calibration curve using pH 7 and pH 4 buffer solutions.

Plankton Analysis

The filter used for each sample preparation was placed into 10 mL of 90% acetone in a centrifuge tube and extracted in the darkness at 4°C for a minimum of 18 hours, but no greater than 24 hours (Wetzel and Likens 2000). The centrifuge tube was centrifuged at 2500 rpm for 10 minutes and the chlorophyll-a content was assayed fluorometrically (Turner Designs TD-700, Sunnyvale, CA). The procedure followed for the measurement of chlorophyll-a was the non-acidification technique for extracted chlorophyll-a analysis as described by Welschmeyer (1994).

Zooplankton were collected using triplicate hauls from a depth of 10 m in each lake in October 2004 using a 0.7 m diameter 176 µm mesh net. Zooplankton were fixed in sugar-formalin fixative and enumerated using a binocular microscope.

Winkler Titrations

Dissolved oxygen was measured using the Winkler titrations on the six discrete water samples. The dissolved oxygen bottle (300 mL) was filled and allowed to overflow for approximately 20 seconds (approx. 600 mL) to ensure that minimal oxygen was introduced from air exposure. Following a procedure from Wetzel and Likens (2000), 2 mL of manganous sulfate were added followed by an addition of 2 mL of alkali azide. This combination caused a precipitate to form, and time was allowed for the settling of this precipitate . Once settling was complete, 2 mL of concentrated sulfuric acid were added to acidify the solution. This produced an orange/brown solution lacking a precipitate. In the lab, 200 mL of this orange/brown solution were measured out and titrated with 0.025N sodium thiosulfate. The titrant was added until the orange/brown solution became pale yellow in color. At this point, 5 drops of starch indicator were added to the solution, changing it to a blue color. The titrant was added drop by drop until the solution became colorless, thus indicating its endpoint. From the volume of titrant added the dissolved oxygen content was detrmined (Wetzel and Likens 2000).

Ion Specific Electrode

Chloride measurements were accomplished by using a Thermo Orion ion specific electrode on a version 290A+ voltammeter (Thermo Orion). Before making measurements, the sensor for the electrode was soaked in deionized water for 10-15 minutes and then transferred to a solution of approximately 100 ppm Cl^-. Standards for this method were made by diluting a 1000 ppm Cl^- standard (Spex CertiPrep, Metuchen, NJ) to cover a range of concentrations from 2 ppm up to 150 ppm. In order to verify the continued accuracy of the electrode during analysis, a check calibration curve was created after every 6-10 measured samples to check that the calibration slope was consistent.

The electrode was able to measure all samples from UCL and LCL including the highway inlet of UCL, connecting stream, and the outflow from LCL. However, the extremely low Cl^- concentrations of CP and the inflows originating from Cascade Mountain made measurements unreliable using this electrode. These samples were sent to the Environmental Analytical Lab at Clarkson University for analysis via ion chromatography. Measurements in the analytical lab were made by using ion chromatography with a Dionex 4000i system and were verified by standard addition methods.

Samples were prepared for analysis by diluting them with an ultra-pure filtered deionized water (Millipore Gradient), in a 1:4 (sample:total volume) dilution ratio. This dilution alleviated the need for an ionic strength adjustor to be added to all samples and standards. The dilution matched the sample matrix to that of the standards and still allowed for the concentrations to fall within the range of standards being used.

Analysis of the standards and the mV readings on the meter created the calibration curve. Once the slope was verified by previous samples analyzed by the analytical lab, the individual samples were measured. A $\log_{10}$ based calibration curve was created to determine the concentration of the samples. Taking the 1:4 dilution factor into account, the measured value was multiplied by four yielding the actual sample concentration in mg/L. Error was analyzed using a sample that was measured by ion chromatography. The errors have been reported as a percent difference from the analytically measured samples.

Flame Atomic Spectroscopy

Base metal concentrations were determined via flame atomic absorption spectroscopy (FAAS) with a Perkin-Elmer Flame AAnalyst 200 Atomic Absorption Spectrometer (Perkin-Elmer, Wellesley, MA) . Air-acetylene gases produced the flame and the spectrophotometer was set to take three measurements of each sample at three-second intervals. The instrument yielded the most accurate data sets by setting the internal computer to establish a "non-linear through zero" calibration setup.

Determination of the concentration of Na and K metals involved flame emission. Standards were made using atomic absorption stocks (Fischer, Hampton, NH); the Na standards included 2, 5, 10, 20, 50, and 100 ppm and the K standards included 1, 2, 3, and 5 ppm. The standards were made with a dilution solution of 2% HCl and the samples for the emission analysis were diluted using a 1:2 ratio with 2% HCl.

Analysis of Ca and Mg was determined via atomic absorption using Fisher Brand hollow cathode lamps (Hampton, NH). In response to a complexation in the flame, which made the absorption values lower than the actual concentration, the addition of a matrix modifier to compensate for this complexation was added to the samples and standards (Jonca and Lewandowki 2004). Lanthanum chloride ($LaCl_3$) was used for this purpose in a stock solution of 5.0%. The Ca standards included 2, 5, 10, 20, and 30 ppm and the Mg standards included 1, 2, 3, 4, and 5 ppm. The standards were made with a dilution solution of 2% HNO_3, while the samples for absorption were prepared by a 1:2 dilution composed of the sample and the $LaCl_3$ solution with 2.0% HNO_3. The final $LaCl_3$ concentration in the prepared samples was 0.08% (V/V).

A standard reference material for river water (SLRS-4; National Research Council of Canada, Ontario, Canada) was used in all analysis made by FAAS. Error in analysis was determined by running a SLRS-4 sample with each metal group and was reported by using an average percent difference from the certified content in the SLRS-4. The SLRS-4 samples were prepared in the same manner as the lake samples.

Graphical Representations and Statistical Analysis

Over the course of a twenty-four month study (July 2, 2003 to June 1, 2005), the corresponding Cl^-, metals, dissolved oxygen, and chlorophyll-a data was illustrated through the use of three-dimensional contour plots; the plots allow for a xyz dataset to be represented in a graphical manner. Sigma Plot version 8.0 was used to generate these isopleth figures By utilizing the Inverse Distance function and calculating by using the Nearest Neighbor Option, the program was able to create a smooth contour graph of the data. The Cl^-, Na, K, Ca, and Mg data were statistically analyzed by ANOVA for combined data and by ANCOVA for Cl^- analysis with StatView version 5.0.1.

RESULTS

Temperature Profiles

The measured temperatures from July 2003 through March 2005 of UCL and LCL are shown in Figs. 5-2-4. By recording the monthly temperature changes in each lake, the movement of thermal energy was measured in order to study the extent of thermal mixing throughout the lake. From the compiled thermal data, both UCL and LCL appear to exhibit similar seasonal temperature cycles. As the winter season approached, a cooling began until an isothermal state was achieved in UCL. However, LCL does not exhibit an isothermal state at the same time that UCL does. In the springtime, a new cycle began with the formation of the thermocline. This thermocline existed throughout the summer season and into the fall when the cycle began again.

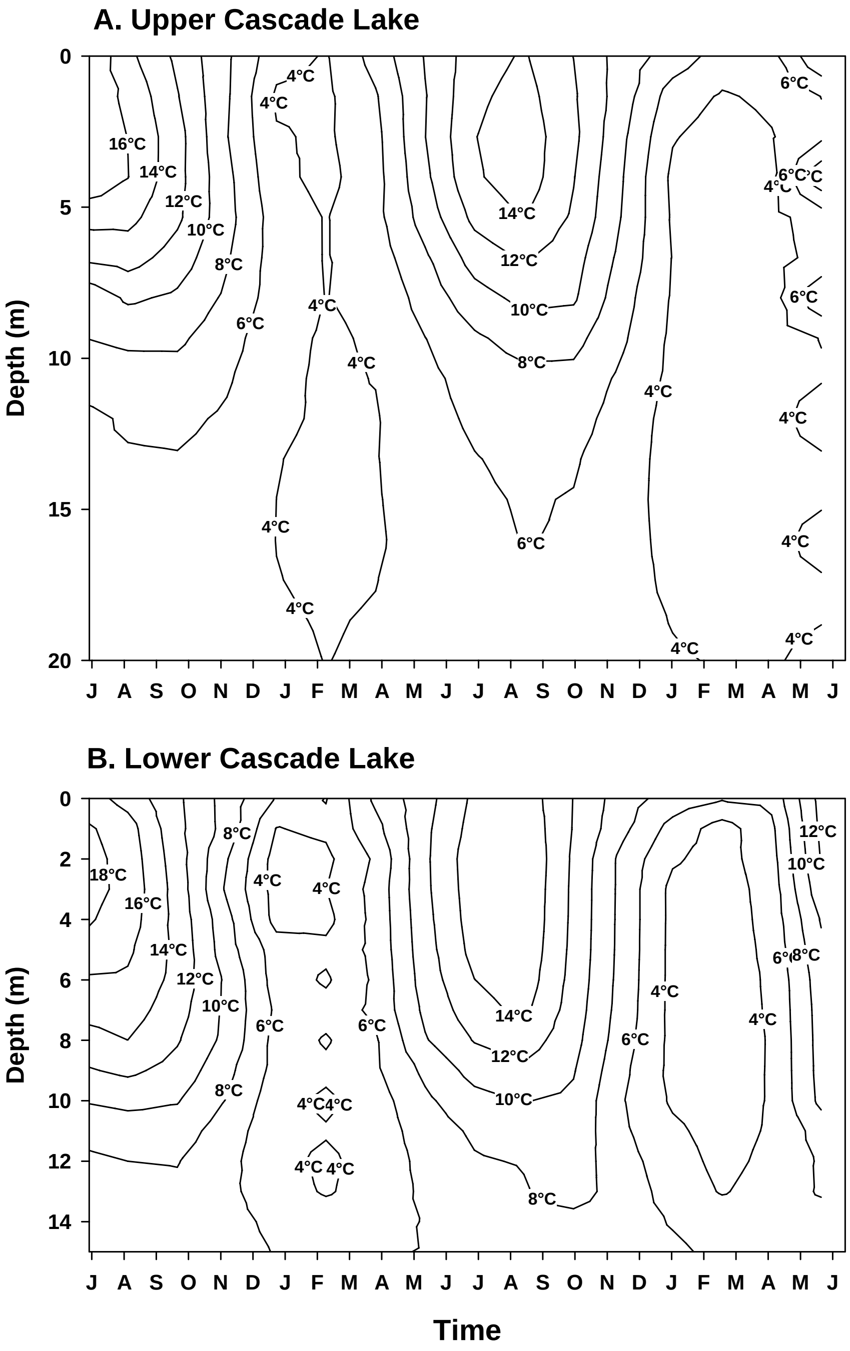
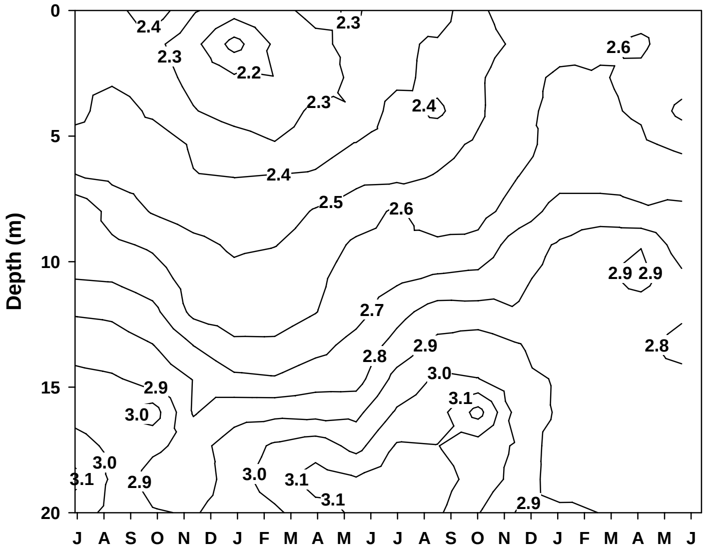


Fig. 5-2-4. Temperature profiles of Upper and Lower Cascade Lake from July 2003 through June 2005.

Anion Analysis – Chloride Profiles

The Cl$^-$ profiles of UCL and LCL show the molar Cl$^-$ concentrations of each lake (Figs. 5-2-5). In both lakes, a trend of increasing Cl$^-$ concentrations was seen with increasing depths. However, the two lakes differed in the magnitude of their Cl$^-$ concentrations. Note the apparent lack of complete mixing found in each Cl$^-$ profile.

A. Upper Cascade Lake (chloride: mmol/L)



B. Lower Cascade Lake (chloride; mmol/L)

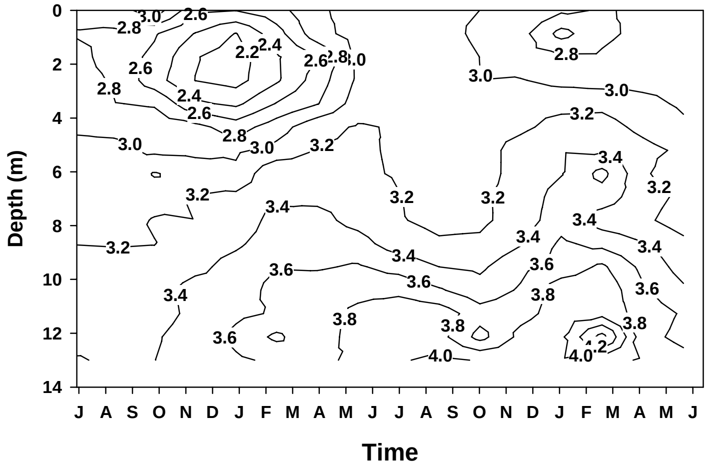
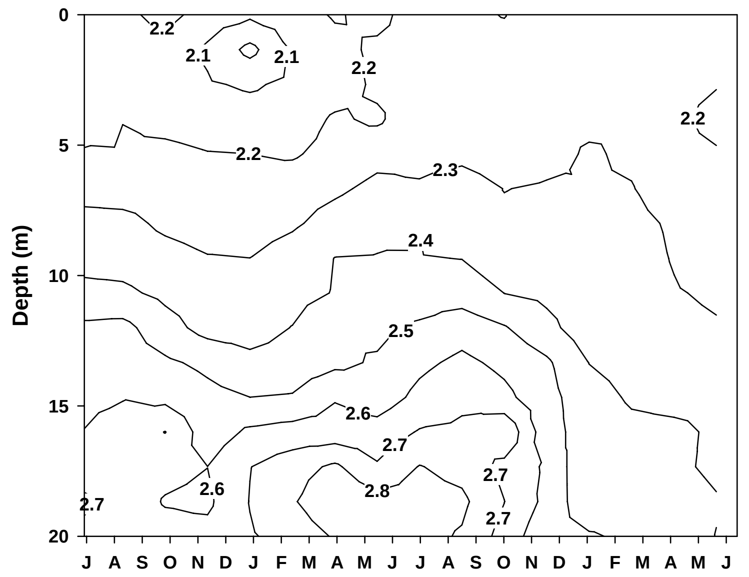


Fig. 5-2-5. Upper and Lower Cascade Lake Cl$^-$ concentration profiles (mmol/L) from July 2003 through June 2005.

Metal Analysis – Sodium, Potassium, Calcium, & Magnesium Profiles

The Na profiles for UCL and LCL (Figs. 5-2-6A, 6B) have very similar shapes and magnitudes as the Cl$^-$ profiles (Figs. 5-2-5A, 5B). In both lakes, a trend of increasing Na concentrations was seen with increasing depths. However, the two lakes differed in the magnitude of their Na concentrations.

A. Upper Cascade Lake (sodium: mmol/L)



B. Lower Cascade Lake (sodium; mmol/L)

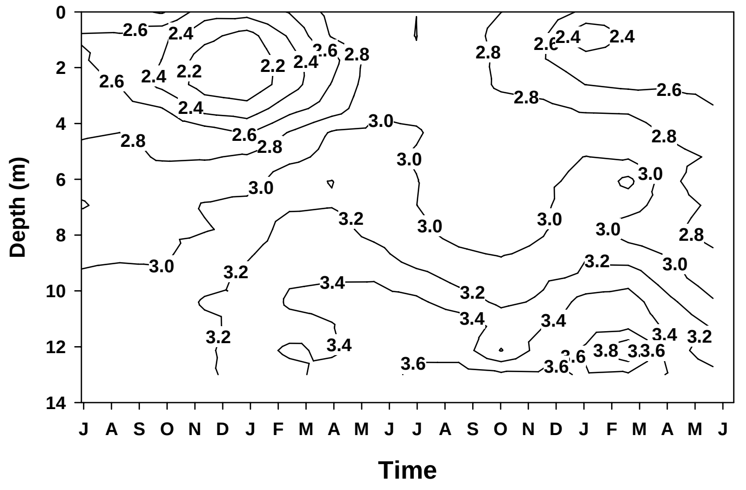
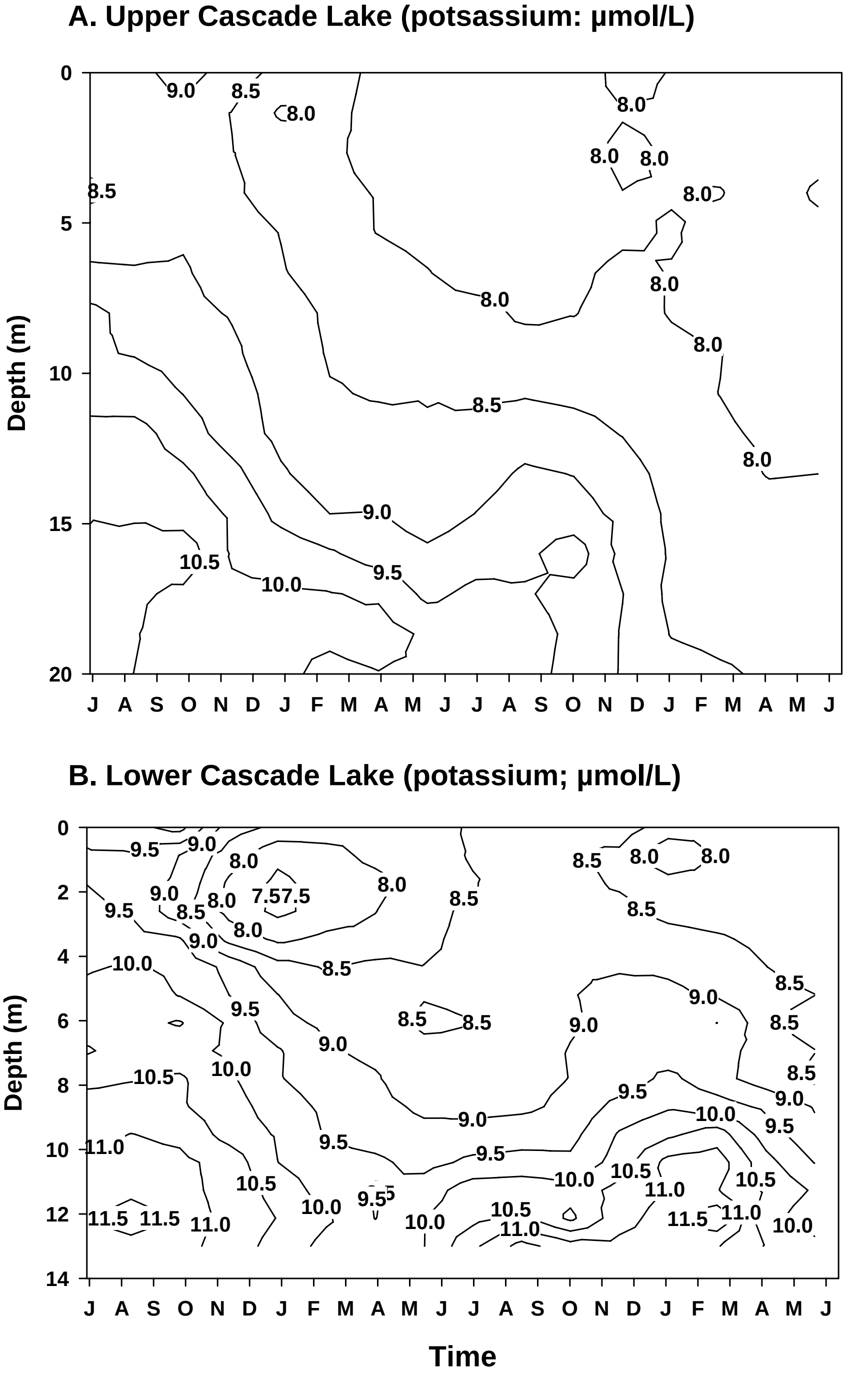


Fig. 5-2-6. Upper Cascade and Lower Lake Na concentration profiles (mmol/L) from July 2003 through June 2005.

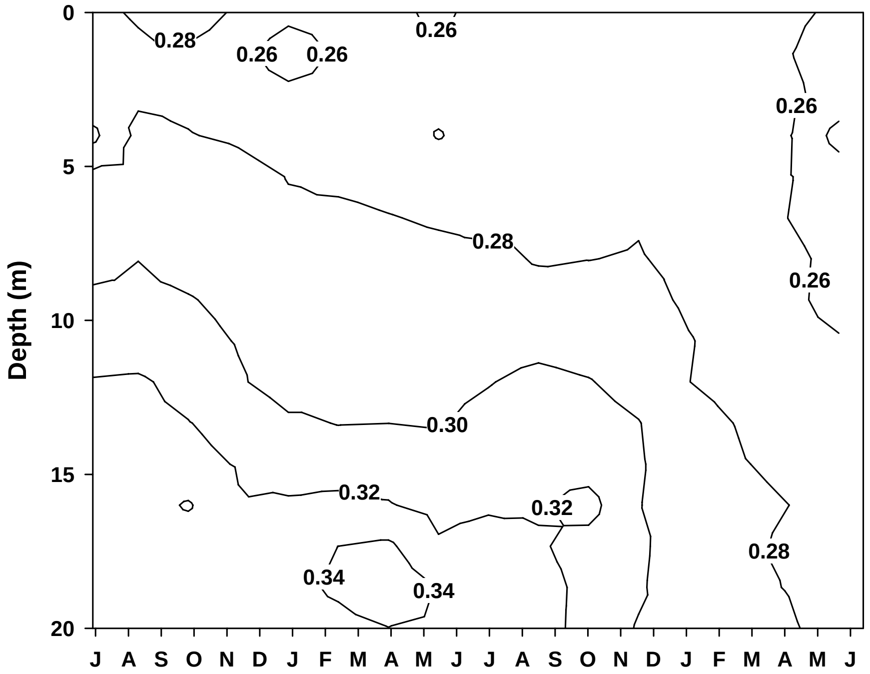
Figures 5-2-7, 5-2-8, and 5-2-9 represent the respective metal concentrations of K, Ca, and Mg in UCL and LCL. Note that K and Mg concentrations are represented in μmol/L.


A. Upper Cascade Lake (potsassium: μmol/L)

B. Lower Cascade Lake (potassium; μmol/L)

Fig. 5-2-7. Upper and Lower Cascade Lake K concentration profiles (μmol/L) from July 2003 through June 2005.

A. Upper Cascade Lake (calcium: mmol/L)



B. Lower Cascade Lake (calcium; mmol/L)

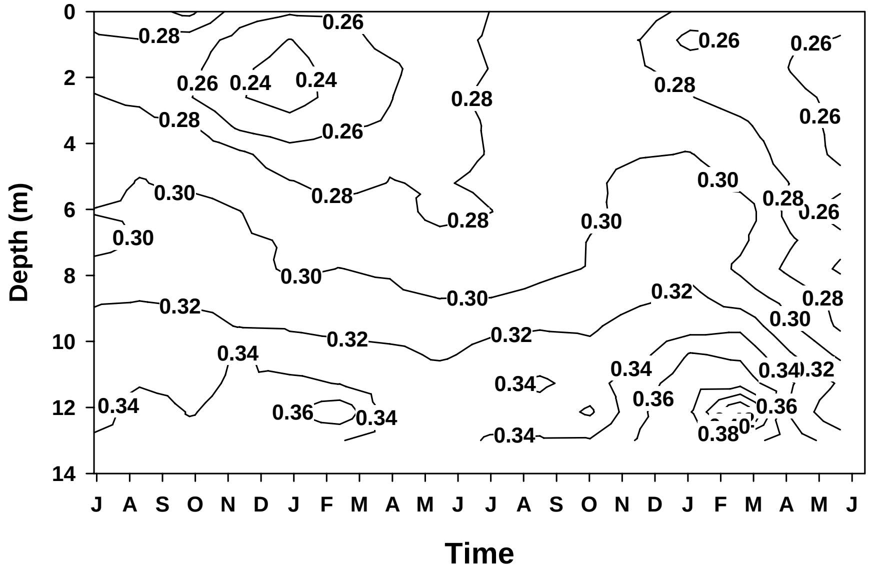
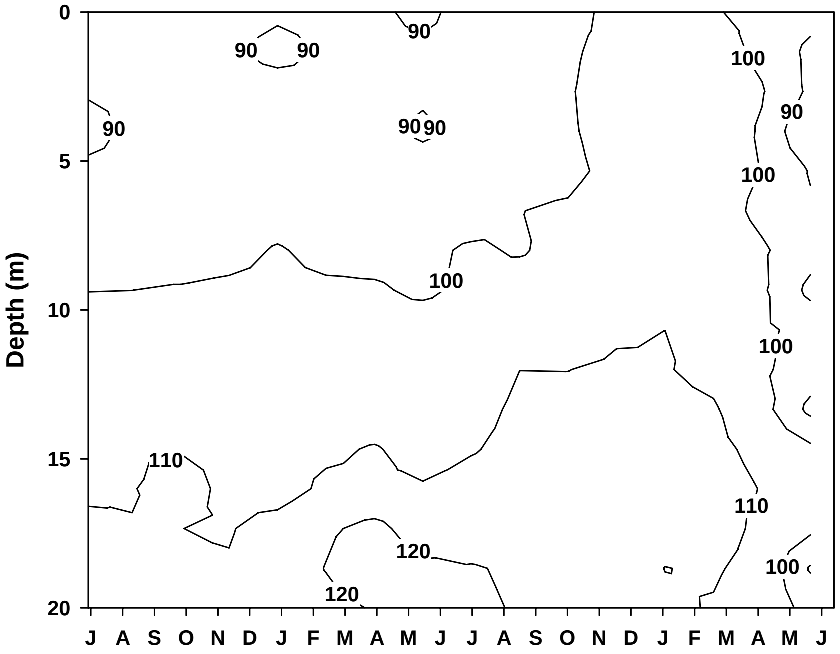


Fig. 5-2-8. Upper and Lower Cascade Lake Ca concentration profiles (µmol/L) from July 2003 through June 2005.

A. Upper Cascade Lake (magnesium: µmol/L)



B. Lower Cascade Lake (magnesium; µmol/L)

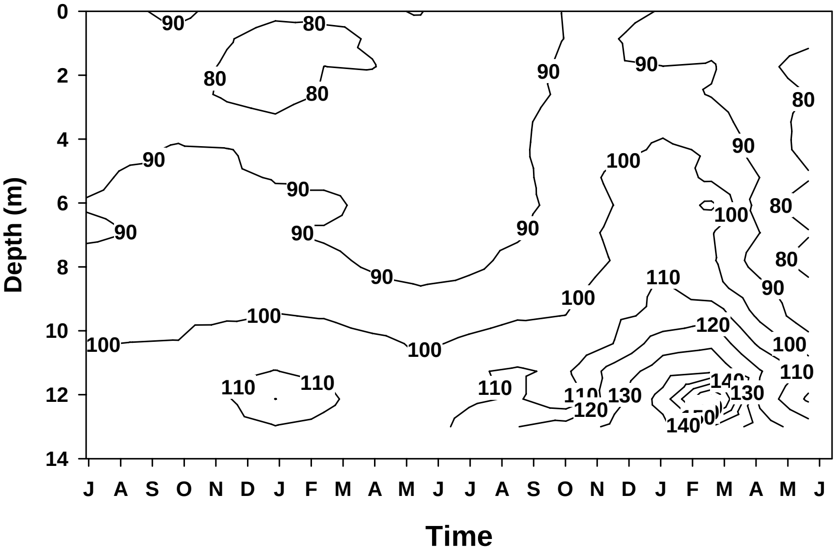


Fig.5- 2-9. Upper and Lower Cascade Lake Mg concentration profiles (µmol/L) from July 2003 through June 2005.

Milliequivalent plots of chloride versus individual base cations can revealed that Na and Cl had the highest degree of correlation in Upper Cascade (Fig. 5-2-10) and Lower Cascade Lake (Fig. 5-2-11) .

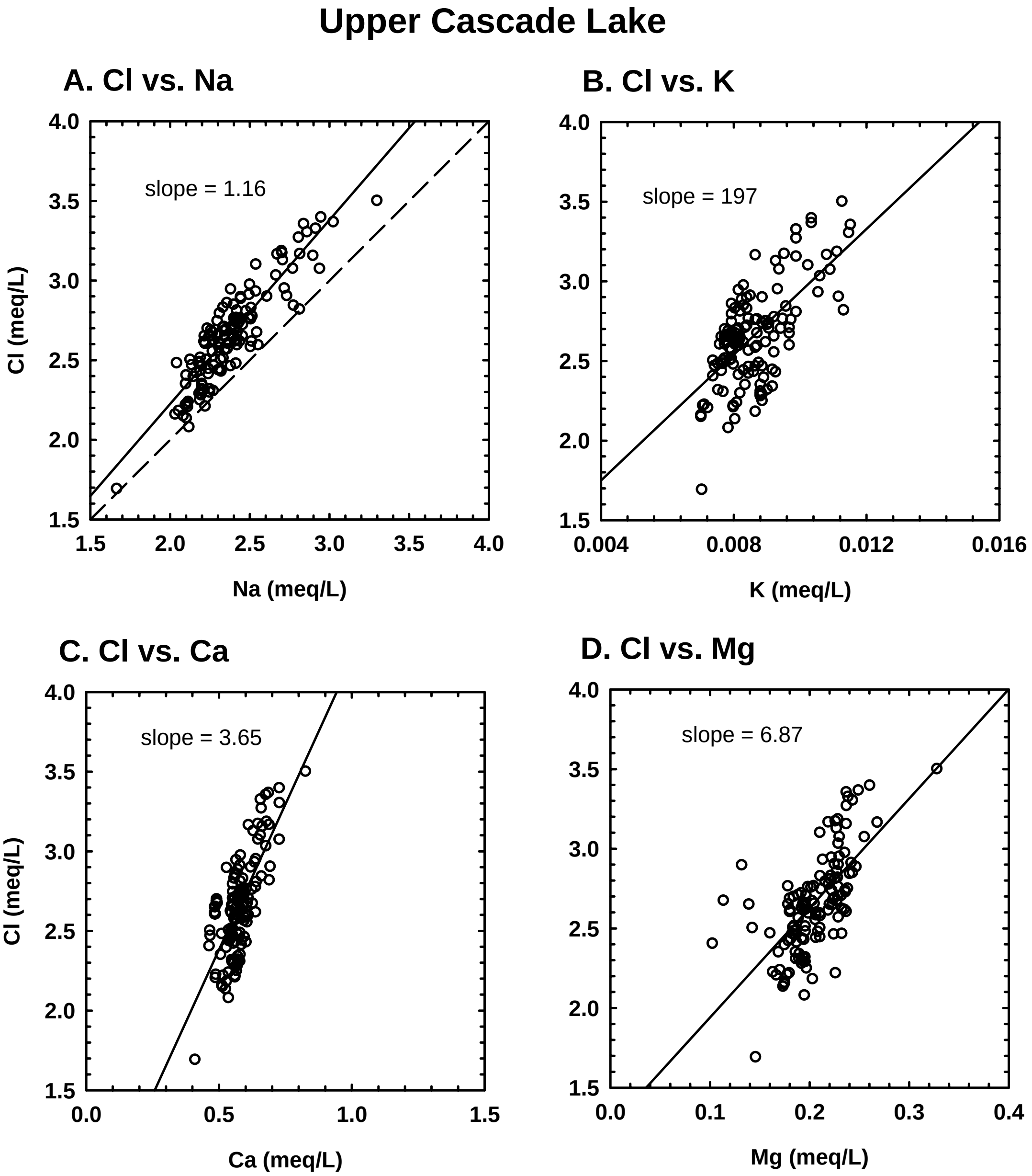


Fig. 5-2-10. Comparison of Cl⁻ with Na, K, Ca, and Mg concentrations in Upper Cascade Lake. The solid line represents a linear least square regression; the dashed line represents the one-to-one ratio of Cl⁻ to the given solute (if visible on the scale).

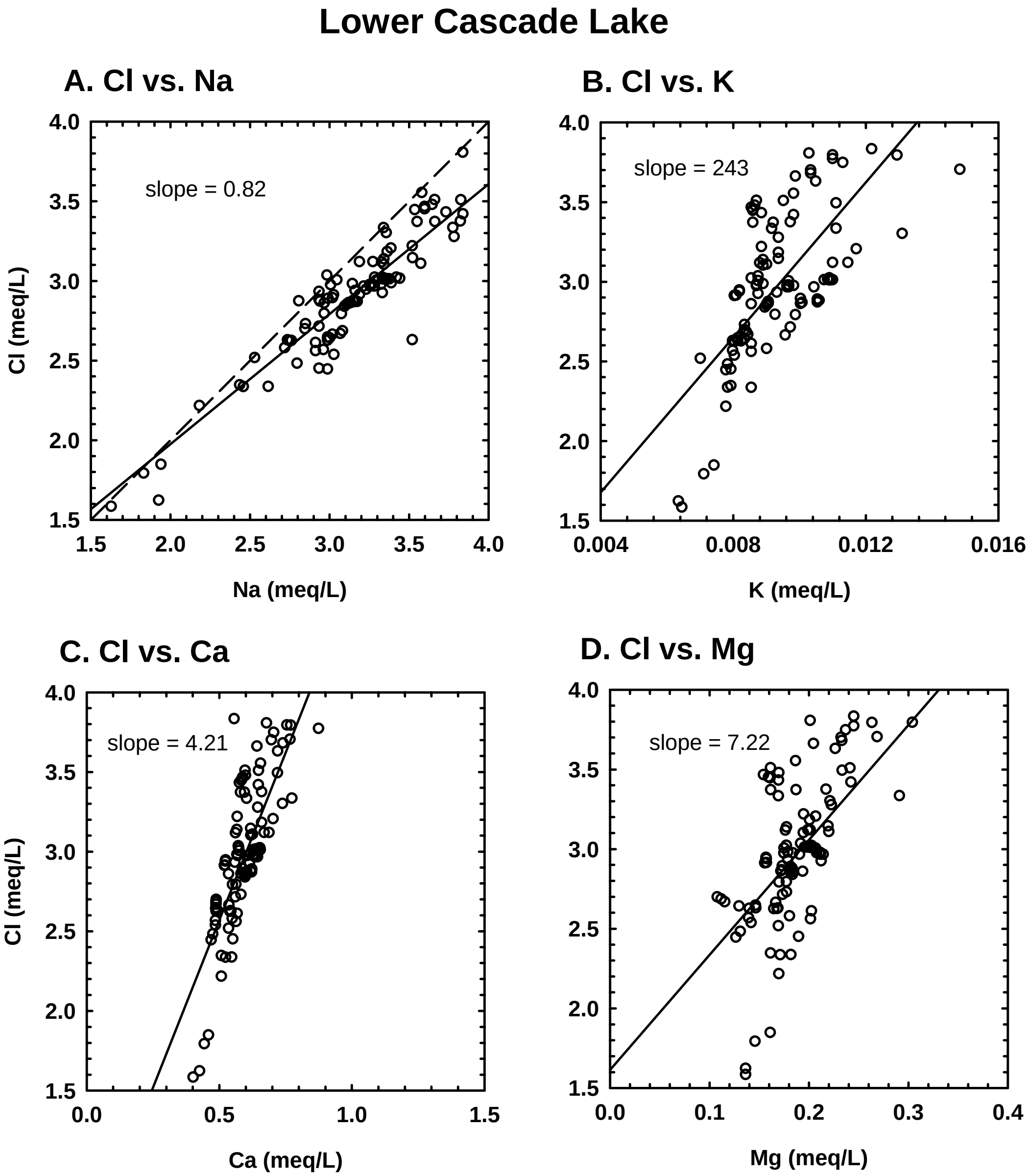


Fig. 5-2-11. Comparison of Cl⁻ with Na, K, Ca, and Mg concentrations in Lower Cascade Lake. The solid line represents a linear least square regression; the dashed line represents the one-to-one ratio of Cl⁻ to the given solute (if visible on the scale).

In both Upper and Lower Cascade Lakes, chloride is enriched relative to all other base cations, except Na (Fig. 5-2-10A, Fig. 5-2-11A).

Comparison Lake Results – Chapel Pond

The seasonal temperature profile of the control lake, CP, indicates that the water began to mix thermally during the month of October and became isothermal before January. In January 2004 and 2005, an inverse stratification was observed, indicating a complete thermal mixing. Stratification began to form again during the month of May (Fig. 5-2-12).

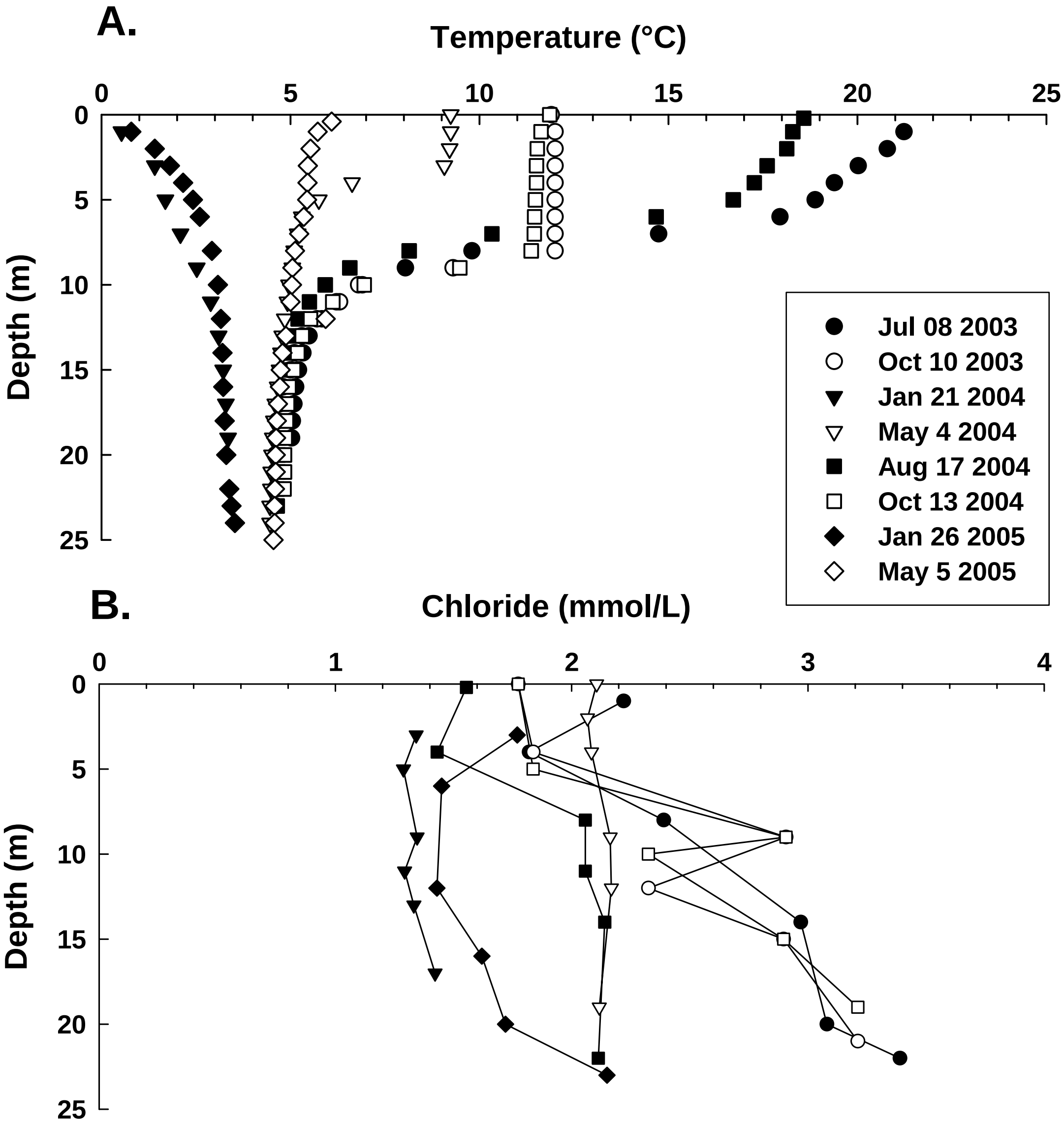


Fig. 5-2-12. Seasonal temperature profile (A) and chloride levels (B) in of Chapel Pond over the two year study period.

The Cl$^-$, Na, K, Ca, and Mg concentrations found in the seasonal sampling of CP (Fig. 5-2-13A, 13B, 13C, 13D) showed a trend where concentrations of Na and K increased with increasing depths as was seen in UCL and LCL, but the magnitude of the concentrations was much lower in CP.

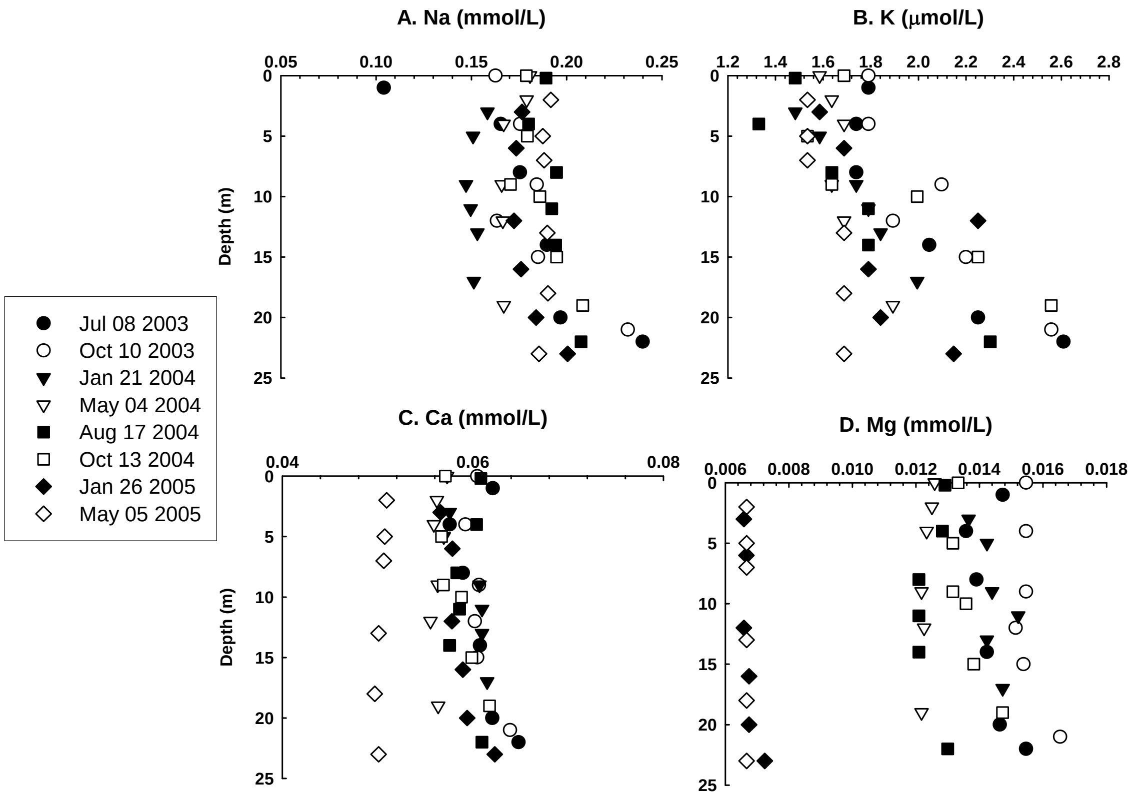


Fig. 5-2-13. Seasonal base cation concentrations for Chapel Pond (2003-2005).

Milliequivalent plots generated from water quality data from Chapel Pond (Fig 5-2-14) revealed that NaCl was an unlikely source of the majority of chloride in this lake. In contrast to Upper and Lower Cascade Lakes, the closest ratio to unity for chloride was observed for Ca and Mg (Fig. 5-2-14C, 5-2-14D). Sodium concentrations could only account for approximately 43% the chloride in Chapel Pond, suggesting that road salt impacts are not as pronounced as in the Cascade Lakes, despite the proximity of Chapel pond to SR-73.

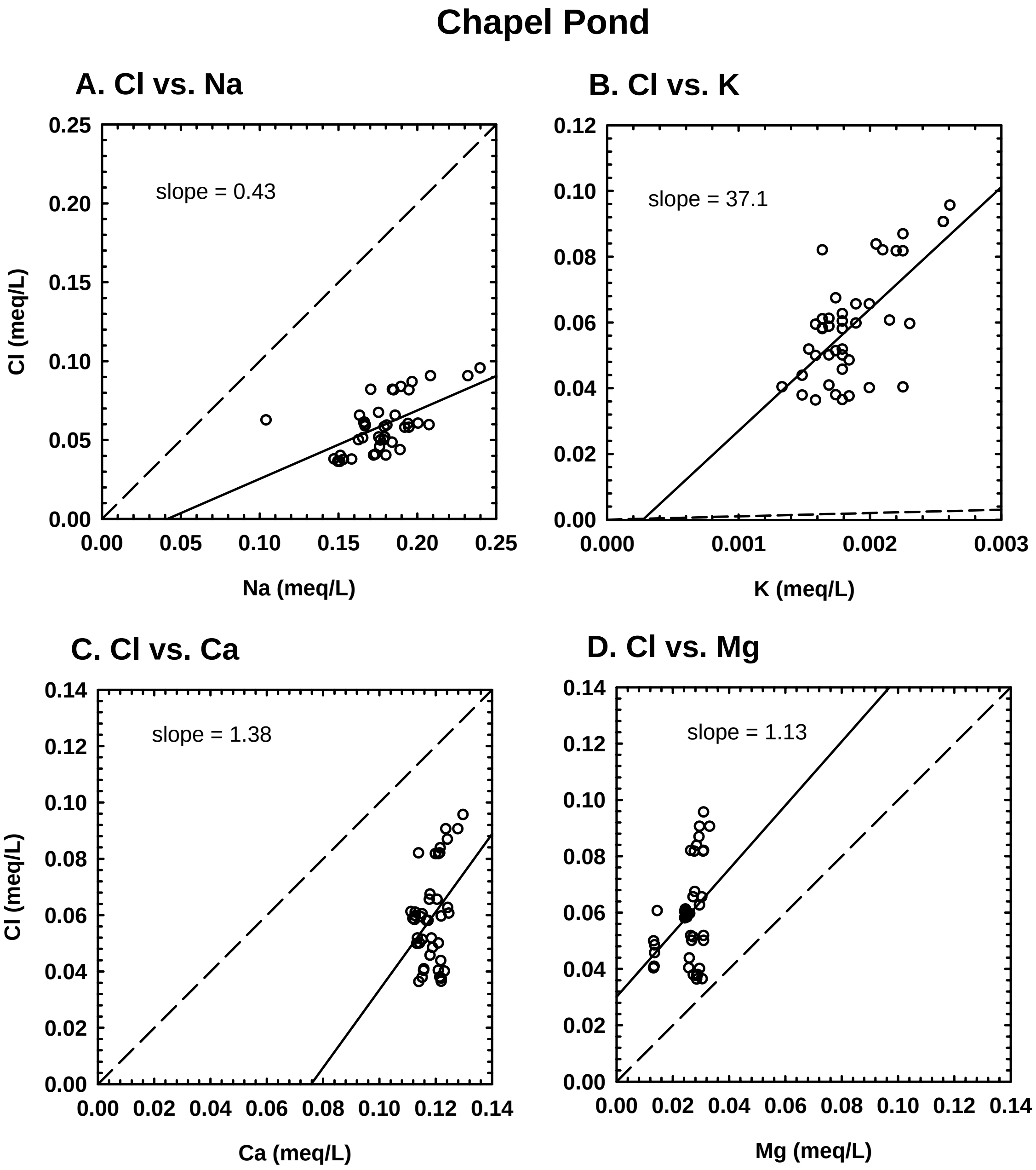


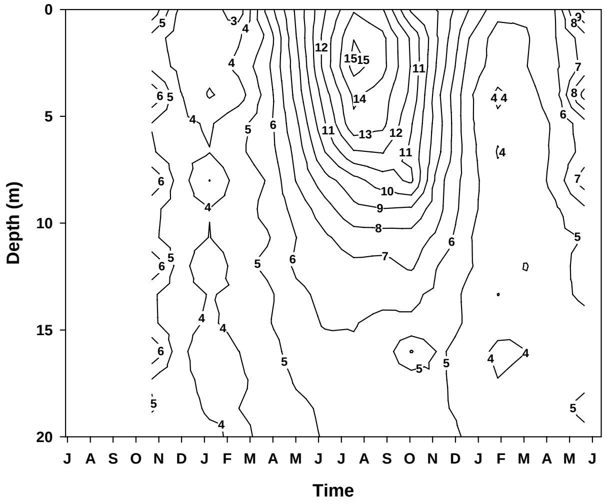
Fig. 5-2-14. Comparison of Cl⁻ with Na, K, Ca, and Mg concentrations in Chapel Pond (2003-2005). The solid line represents a linear least square regression; the dashed line represents the one-to-one ratio of Cl⁻ to the given solute.

Dissolved Oxygen Analysis

Determination of dissolved oxygen concentrations in each lake indicated that they all had comparable trends and concentrations. In UCL, LCL, and CP, the general seasonal dissolved oxygen trend was an increase in oxygen during the period when the water temperature was coldest and then the concentration decreased slightly during the stratification period (Figs. 5-2-15, 5-2-16). With respect to profiles, the lakes exhibit a heterograde oxygen profile, characterized by declining dissolved oxygen concentrations with increasing depths, except during periods of mixing (early spring and late fall). Note that the sampling period was recorded starting in January 2004 due to changes from the initial procedure.

Of the three lakes studied, Lower Cascade showed hypoxic and anoxic conditions in the deepest water over a three month period during the summer of 2004 (Fig. 5-2-17). These oxygen concentrations were well below the limit considered optimal for coldwater fish such as trout or round whitefish.

A. Upper Cascade Lake (dissolved oxygen; mg/L)



B. Lower Cascade Lake (dissolved oxygen; mg/L)

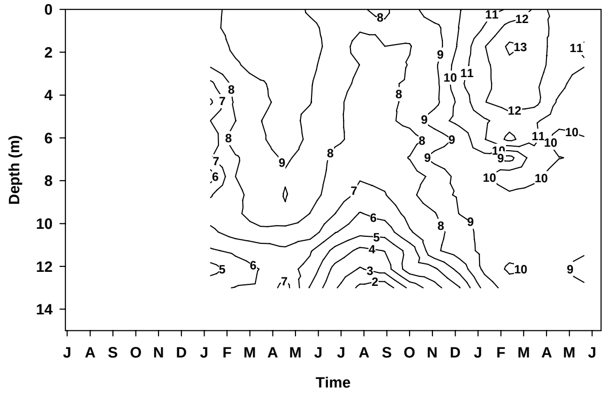


Fig. 5-2-15. Dissolved oxygen concentration (mg/L) profile for Upper and Lower Cascade Lake from January 2004 to June 2005.

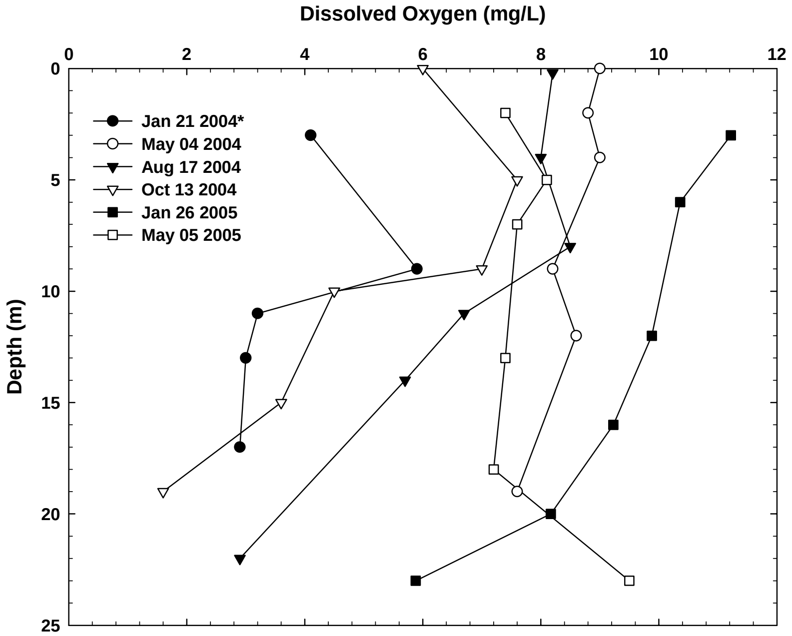


Fig. 5-2-16. Dissolved oxygen concentration profiles in Chapel Pond from January 2004 through May 2005. Note: data for Jan 2004 are considered to have been compromised by the freezing of reagents prior to fixation.

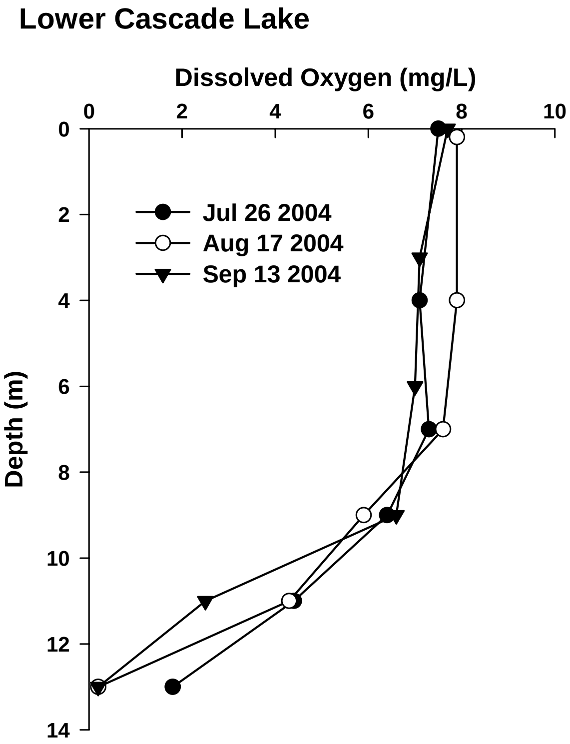


Fig. 5-2-17. Dissolved oxygen concentration profiles in Lower Cascade Lake during the summer period of thermal stratification. Water below 10 m was hypoxic.

Plankton Analysis

Chlorophyll-a (Chl-a) was analyzed in each water sample in order to characterize the biomass of photosynthetic organisms in each lake (Table 5-2-2). Based on these results, Upper and Lower Cascades Lake are considered to be mesotrophic (chl-a ≥ 1 µg/L). Chapel Pond is considered to be oligotrophic. The oligotrophic status of Chapel Pond is supported by the observation of the Cladoceran *Holopedium* sp. in this lake, but not UCL and LCL (Table 5-2-3); the presence of this genus is indicative of low nutrient systems.

Table 5-2-2. Chlorophyll-a observed over 2003-2005. Upper cascade and Lower cascade Lake were sampled at monthly intervals; Chapel Pond was sampled every 3 months.

	Chlorophyll-a concentrations (µg/L)		
	Upper Cascade	**Lower Cascade**	**Chapel Pond**
Mean	2.14	1.19	0.66
Standard deviation	4.09	0.63	0.74
Median	1.05	0.94	0.47
Minimum	0.09	0.37	0.04
Maximum	23.74	3.42	4.39
N	132	72	48

Table 5-2-3. Zooplankton densities. Water was collected on Oct 13 2004. Values are organisms per cubic meter (± standard deviation, n = 30).

	Cladocera			**Copepoda**	
	Bosmina	***Daphnia***	***Holopedium***	**Calanoid**	**Cyclopoid**
Upper Cascade	510 ± 130	1320 ± 286	0	3350 ± 54	1950 ± 390
Lower Cascade	2010 ± 610	2230 ± 650	0	2200 ± 440	2630 ± 280
Chapel Pond	139 ± 60	763 ± 54	78 ± 94	2480 ± 400	165 ± 40

pH Analysis

The results of the lab pH analysis for the three lakes show them to be circumneutral (Table 5-2-4). Increases in pH (>8) were observed in the surface water in summer and is attributed to photosynthetic activity in these poorly buffered water. Acidic conditions (e.g. pH <6.5) were only observed in the hypolimnion and are attributed to the loss of acid neutralizing capacity.

Table 5-2-4 Acidity of lake water measured over the period of 2003-2005 (Average pH was determined by averaging the H+ activity then averaging and converting to pH).

	pH		
	Upper Cascade	**Lower Cascade**	**Chapel Pond**
Mean	7.16	7.24	6.79
Median	7.38	7.48	6.95
Minimum	6.37	6.44	6.24
Maximum	9.41	9.72	8.22
N	129	108	45

Present Lakes Compared to Twenty Years Ago

The Adirondack Lake Survey Corporation (ALSC) collected water samples from UCL (1984), LCL (1984), and CP (1986) on their Cl^-, Na, K, Ca, and Mg concentrations (see section 3-5). Table 5-2-5 depicts the differences between the studies in the 1980s and the current 2003 through 2005 lake concentrations for UCL, LCL, and CP; data from some other lake studies, taken in the 1980s, have also been included in the table. Giant Washbowl (44.0838, 73.4420), Round Pond (44.0726, 73.4357), Upper Ausable Lake (44.0503, 73.5146), and Lower Ausable Lake (44.0705, 73.4930) are all located in Keene, NY. All of these lakes are in the same general area, however the only difference is that these lakes are not as influenced by a major highway as SR-73 influences UCL and LCL.

Table 5-2-5. A compilation of data collected by the Adirondack Lakes Survey Corporation, from 1984 and 1986, and the current study on Upper and Lower Cascade Lakes and Chapel Pond, from July 2003. Also included are some additional concentrations taken from a selection of nearby lakes that do not directly border SR-73.

.

ALS Data on the Cascade Lakes							
			mmol/L				
	Year	*Depth (m)*	*Cl^-*	*Na*	*K*	*Ca*	*Mg*
Upper Cascade Lake (Keene, NY)	1984	1.5	0.77574	0.4787	0.0056	0.1838	0.065
Lower Cascade Lake (Keene, NY)	1984	1.5	0.60310	0.5217	0.0056	0.1783	0.0568
Chapel Pond (Keene Valley, NY)	1986	1.5	0.02426	0.0439	0.0013	0.0646	0.0136
Upper Cascade Lake	2003	2	2.39774	2.1441	0.0089	0.2636	0.0873
Lower Cascade Lake	2003	2	3.01834	2.6644	0.0096	0.2679	0.0833
Chapel Pond	2003	1	0.06262	0.1040	0.0018	0.0621	0.0147
Adirondack Lake Selection							
Giant Washbowl (Keene, NY)	1984	1.5	0.00705	0.0261	0.0026	0.1202	0.0214
Round Pond (Keene, NY)	1984	1.5	0.00564	0.0209	0.0013	0.0683	0.0119
Upper Ausable Lake (Keene, NY)	1984	1.5	0.00564	0.0300	0.0033	0.0678	0.0177
Lower Ausable Lake (Keene, NY)	1985	1.5	0.00621	0.0261	0.0033	0.0653	0.0169

In order to characterize the mixing exhibited by UCL, LCL, and CP in this study, the Cl^- data was separated into epilimnion and hypolimnion samples as defined previously. For each date, the average epilimnion and average hypolimnion concentrations were calculated and a ratio of average epilimnion concentration/average hypolimnion concentration was made. Figure 5-2-18 displays the output for UCL and LCL and Figure 5-2-19 displays the output for CP.

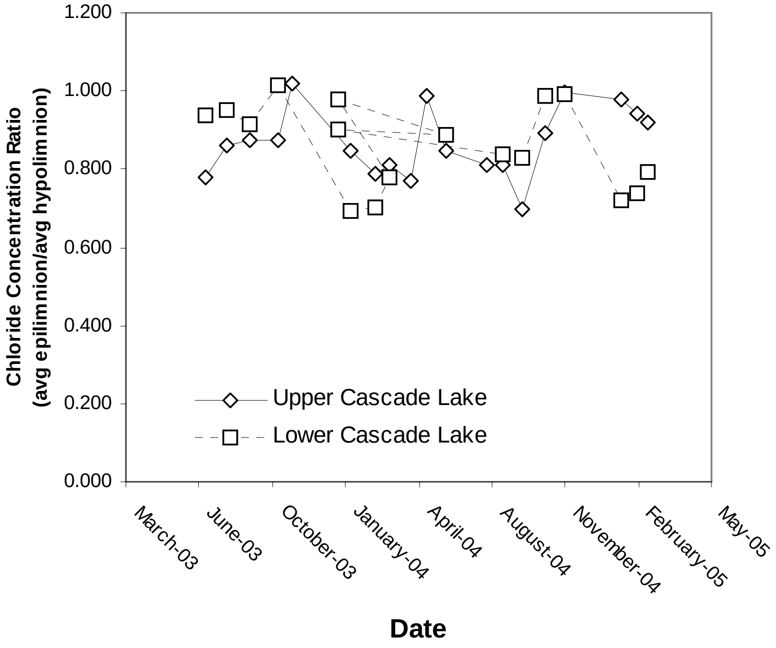


Fig. 5-2-18. Numerical comparison of the average epilimnion and hypolimnion chloride concentrations for Upper Cascade Lake and Lower Cascade Lake from 2003 to 2005.

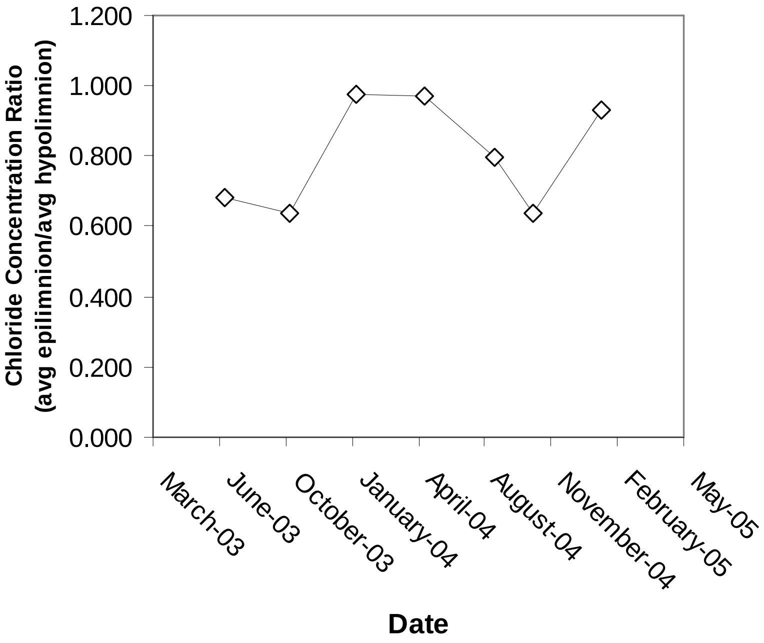


Fig. 5-2-19. Results of the numerical comparison of the average epilimnion and hypolimnion concentrations for Chapel Pond from 2003 through 2005.

Since the data from the 1980s (Sutherland 1986) did not have multiple data points, the single measured Cl^- concentrations for the epilimnion and hypolimnion were used to make a similar ratio for comparison and the results for UCL and LCL are shown in Fig. 5-2-20.

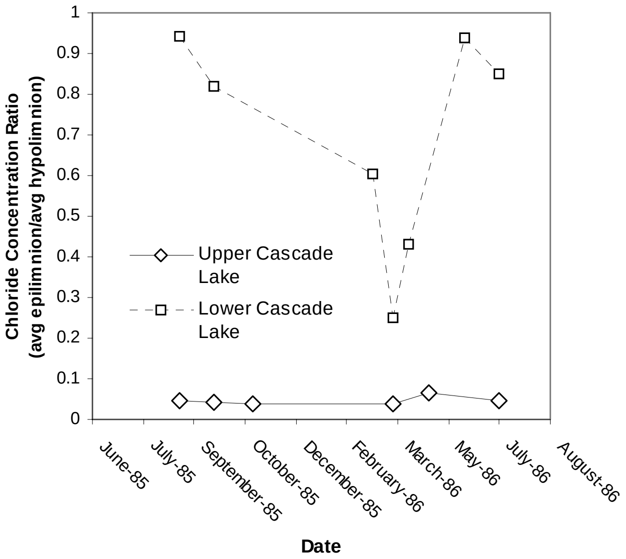


Fig. 5-2-20. Comparison of the epilimnion and hypolimnion for Upper Cascade Lake and Lower Cascade Lake from August 1985 through July 1986. Note the differences between each lake.

Error Report

The error results and detection limits for this study are displayed in Table 5-2-6.

Table 5-2-6. Error report on the chemical analysis and the detection limits for the ion selective electrode and Flame Atomic Absorption and Emission.

Element	***Avg. % Difference***	***Detection Limit (mmol/L)***
Cl-	0.0185	> 0.056
Na	-0.0149	0.00135
K	-0.0038	0.0000256
Ca	-0.0868	0.00110
Mg	-0.0274	0.0000411

Statistical Analysis

All concentrations of Cl^-, Na, K, Ca, and Mg were statistically analyzed in order to test whether the apparent trends were significant. Table 5-2-7 displays the results of a factor analysis in which Cl^- and all four metals were plotted against each other in a five dimensional plane. In addition, data from all three lakes were combined and plotted together in order to identify any inter-lake trends between the concentrations.

Table 5-2-7. Factor analysis of the basic water chemistry in Upper Cascade Lake, Lower Cascade Lake, and Chapel Pond. The concentrations of Cl^-, Na, K, Ca, and Mg all have inclusion criteria as shown under the Factor 1 Oblique Loading column.

Bartlett's Chi Square	3299.645
Degrees of Freedom	14
P-value	< .0001

Factor 1 Oblique Loading	
Cl-	0.984
Na	0.977
K	0.98
Ca	0.992
Mg	0.956

In the factor analysis described above, 96% of the variance is explained by a single factor. This indicates a strong positive association among Cl^-, Na, K, Ca, and Mg. Based on this factor analysis, all measured chemical species can be lumped together into a single factor (PC1) for a combined analysis. Table 5-2-8 depicts the results from the ANOVA using PC1. This ANOVA was performed in order to verify the sensitivity of the data regarding depth, season, and lake. The “lake*season” term analysis of the data refers to the changing conditions of the thermocline in each lake as the season changes. As each month passes, the thermocline will adjust depending on the atmospheric temperatures, this is the “lake*season” value analysis takes this into account. Season definition was as follows: October through April was defined as the winter season, while May through September was defined as the summer season. These months were chosen in order to represent the months of thermocline stability in the summer months and the lack there of during the winter months.

Table5- 2-8. Results of the combined ANOVA to verify the sensitivity of the data regarding depth, season, and lake.

	Degrees of Freedom	*F-Value*	*P-Value*
Lake	2	320.322	<0.0001
Depth	1	124.066	<0.0001
Season	1	1.566	<0.0001
*Lake*Depth*	2	31.363	<0.0001
*Lake*Depth*Season*	2	12.7	<0.0001
Residuals	243		

Based on the strong interactions in the combined ANOVA (P-values <0.0001), the null hypothesis of combining the data for statistical analysis was rejected. Each lake was then to be individually analyzed in order to avoid interpreting strong interactions observed between the chemical concentrations and the lakes, seasons, and depths.

Tables 5-2-9, 5-2-10, and 5-2-11 depict the results of the ANCOVA performed on the factored data, but for each individual lake. Table 5-2-12shows the difference in magnitudes of the mean PC1 values recorded by this statistical analysis.

Table 5-2-9. Results of the ANCOVA performed on the Upper Cascade Lake PC1 with respect to depth, season, and depth-by-season.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	100.913	<0.0001
Season	1	2.068	0.1533
*Depth*Season*	1	6.057	0.0154
Residual	110		

Table 5-2-10. Results of the ANCOVA performed on the Lower Cascade Lake PC1 with respect to depth, season, and depth-by-season.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	72.421	<0.0001
Season	1	6.973	0.0097
*Depth*Season*	1	12.19	0.0007
Residual	95		

Table 5-2-11. Results of the ANCOVA performed on the Chapel Pond PC1 with respect to depth, season, and depth-by-season.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	18.963	<0.0001
Season	1	0.588	0.4481
*Depth*Season*	1	2.424	0.12
Residual	38		

Table 5-2-12. Comparison of the mean and standard error of the calculated PC1 values from the statistical analysis.

	Upper Cascade Lake		*Lower Cascade Lake*		*Chapel Pond*	
	Summer	*Winter*	*Summer*	*Winter*	*Summer*	*Winter*
Epilimnion	0.074 (±0.029)	0.140 (± 0.026)	0.400 ± 0.030)	0.243 (± 0.077)	-2.13 (± 0.007)	-2.16 (± 0.003)
Hypolimnion	0.488 (± 0.044)	0.074 (± 0.052)	0.649 (± 0.057)	0.831 (± 0.096)	-2.09 (± 0.007)	-2.12 (± 0.004)

As shown in the above PC1 values, the hypolimnion of LCL during the winter months will have the highest concentration of ions compared to the concentrations found in UCL and CP. The lowest concentrations of ions will be found in the epilimnion of CP during the winter months. In order to verify the use of a factored chemistry for each lake, Cl^- was analyzed separately. Tables 5-2-13, 5-2-14, and 5-2-15 display the results of the specific ANCOVA analysis for each lake. Table 5-2-16 represents the mean and standard error of the Cl^- data with respect to each lake.

Table 5-2-13. Results of the ANCOVA performed on the Cl^- concentrations of Upper Cascade Lake.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	103.009	<0.0001
Season	1	0.636	0.4267
*Depth*Season*	1	6.507	0.0121
Residual	110		

Table 5-2-14. Results of the ANCOVA performed on the Cl^- concentrations of Lower Cascade Lake.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	58.386	<0.0001
Season	1	12.579	0.0006
*Depth*Season*	1	14.556	0.0002
Residual	95		

Table 5-2-15. Results of the ANCOVA performed on the Cl^- concentrations of Chapel Pond.

	Degrees of Freedom	*F-value*	*P-value*
Depth	1	58.386	<0.0001
Season	1	12.579	0.0006
*Depth*Season*	1	14.556	0.0002
Residual	95		

Table 5-2-16. Displays the mean and standard error that was calculated from the Cl⁻ concentrations (mmol/L) in each lake and split by season.

	Upper Cascade Lake		*Lower Cascade Lake*		*Chapel Pond*	
	Summer	*Winter*	*Summer*	*Winter*	*Summer*	*Winter*
Epilimnion	2.45 (± 0.031)	2.46 (± 0.045)	3.15 (± 0.047)	2.84 (± 0.108)	0.054 (± 0.001)	0.041 (± 0.001)
Hypolimnion	2.92 (± 0.047)	2.74 (± 0.054)	3.41 (± 0.073)	3.63 (± 0.108)	0.074 (± 0.002)	0.043 (± 0.002

As shown by the Cl^- specific ANCOVA analysis, the similarities in P-values indicate that the relationships were the same as found in the factored ANCOVA analysis. Note the magnitude differences between each lake as seen in Table 5-2-16. Overall, LCL is expected to have higher Cl^- concentrations when compared to UCL and CP. The highest concentrations of Cl^- will be found during the winter months in the hypolimnion of LCL and the lowest concentrations of Cl^- will be found in the epilimnion of CP during the winter months.

The statistical analysis indicated that each lake was significantly different from the others, that the Cl^-, Na, K, Ca, and Mg concentrations can be factored together due to similarities in trends, and that season and depth produced the most significant interactions in UCL and LCL, while only the depth had the most significant interaction for CP. For UCL and LCL, the data analysis showed that there were seasonal influences on the concentrations in addition to the depth of the lake being measured.

DISCUSSION AND FUTURE RESEARCH

Temperature

The temperature profile of UCL shows that UCL had a turnover period occurring during the month of November. At this point, the thermocline completely broke down due to the cooler air temperatures from the fall and winter seasons. The stable stratification that once existed between the warmer epilimnion and the cooler hypolimnion, as seen during the summer months, became absent. When temperature throughout the lake became isothermal, a complete thermal mixing period had taken place in UCL. In LCL, the stable stratification began to destabilize in October each year, rather than in November as in UCL. There was no evidence from the monthly temperature depth profiles to suggest that these dimictic lakes were subject to meromixis.

Chloride Concentrations in UCL and LCL

Both UCL and LCL showed the trend that Cl^- concentrations increased with increasing depths. Two possible scenarios could be used to explain this occurrence. The first scenario involves a body of freshwater with an ionic compound, such as sodium chloride, dissolved in it. This water would then have a higher density compared to pure water. In this situation, the density difference between the waters would draw the higher Cl^- and Na^+ concentrations to the bottom of the lake due to gravity. The second possible explanation could be that a groundwater flow may be seeping into the lake via the basin slopes of the lake. Ordinarily, groundwater does not contain such elevated levels of Na and Cl^-, but since a large quantity of the anti-icing agent is applied to SR-73, it is possible that there may be an accumulation of the agent on the edge of the highway. The anti-icing agent load may then be carried into the soil via precipitation and meltwater. This seepage would then have the potential to transport high concentrations of Na and Cl^- into the lake during the spring and summer months since the brines are frozen and are not being transported during the winter months.

A study conducted by Environment Canada and Health Canada (2001) indicated that Cl^- was readily transported into groundwater systems by physical processes and was then drained into nearby surface waters. In addition, a study by Rosenberry et al. (1999) indicated that Cl^- was transported from a highway into the groundwater system neighboring Mirror Lake, New Hampshire. Results from their studies show that the inflow stream carried the majority of the Cl^- load from the neighboring highway, but some groundwater contamination along the highway was also present. In the specific case of UCL and LCL, the shoreline between the lake and SR-73 was narrower than that found in the study conducted by Rosenberry et al. (1999). It can thus be concluded that UCL and LCL may have a more significant groundwater contribution of Cl^- during the spring and summer months due to the groundwater flow seeping into each lake.

The magnitudes of the Cl^- concentrations between UCL and LCL show distinct differences. LCL has a higher magnitude of Cl^- concentration than does UCL. LCL is a shallow lake with its deepest point, a small “bowl,” at the west-most end of the lake. The morphology and bathymetry of the lake itself may be the explanation for the higher magnitude of Cl^- concentration. A different morphology, possibly a wind factor (Blomqvist and Johansson 1999), or even splash from vehicles (McBean and Al-Nassri 1987) where aerosolized road salt may be carried into the lake and its corresponding watershed are all viable explanations for the observed higher Cl^- concentrations.

As the seasons changed, so did Cl^- concentrations in the lakes. Throughout the spring and summer months, there was a lower concentration of Cl^- in the epilimnetic waters of each lake and a higher concentration in the hypolimnetic waters. During the turnover period, a dilution of the hypolimnetic water was evident, but there was a lack of complete mixing, as a chemocline still existed between the old stratification that was present during the summer months. The formation of a chemocline, which exists year round, could be the initial stages of meromictic conditions. This may be especially true for LCL since the lake seems to have a much higher Cl^- concentration within the lake itself. During the turnover period, a dilution was evident in LCL, but it was not as pronounced as it was in UCL. Also shown was an increase in the hypolimnetic Cl^- concentrations during the winter months. These observations may indicate that LCL is presently at a greater risk to reach meromixis than UCL.

The national toxicological criterion for Cl^- (four day average value) in freshwater is set at 6.4 mmol/L (USEPA 1988). This being the standard, aquatic organisms in these lakes are not likely to be stressed at the concentrations of Cl^- found in UCL, LCL, and CP.

Base Metal Concentrations in UCL and LCL

The base metal profiles of UCL suggest that there was a lack of complete chemical mixing. In each profile of Na, K, Ca, and Mg, there was a layer that did not completely mix or dilute during turnover. A similar trend was also seen in LCL. This year-round chemocline suggests the initial stages of meromixis may be occurring, especially in LCL, however, it may be due to increased flux from sediment porewaters (since thermal profiles do not indicate any resistance to mixing occurs in these lakes).

In order to verify that the main input of Cl^- to UCL and LCL was from the application of road salt and not from a different source, base metal analysis was used to help identify the source of the contamination. Godwin et al. (2002) used milliequivalent plots to classify the contamination of the Mohawk River Basin as that of a halite load. Similar milliequivalent plots have been generated from the UCL and LCL data comparing Cl^- to each measured base metal.

Near one-to-one linear relationships and strong regression values were seen between the concentrations of Cl^- and Na, in milliequivalents, in both UCL and LCL. The correlations between Cl^- and the other base metals (K, Ca, and Mg) in both lakes did not have as strong of values. This strong correlation with Na may be supporting evidence that the loading into UCL and LCL is the result of a NaCl compound. This correlates to the fact that NYSDOT primarily applies NaCl to the highway. This also supports the notion that geological and other natural inputs are a minimal source of Cl^- and Na in this lake system.

The elevated concentrations of K, Ca, and Mg are also of special interest. These observations may be explained by the possible leaching of these metals from the soil into the lake system. The NaCl that is plowed to the edge of the highway potentially flows though the soil via meltwater and precipitation. This NaCl could be inducing ion exchange in the soils by which K, Ca, and Mg move out of the soil and into the lake system (Löfgren 2001, Mason et al. 1999, see section 4-1).

Chapel Pond Chemistry

The temperature profile of CP indicates that the lake underwent complete thermal mixing. As shown by the January sampling date of both 2004 and 2005, the temperatures were nearly isothermal and somewhat inversely stratified.

The lower magnitude of Cl^- concentrations was indicative that this lake was less impacted by anti-icing agents than was UCL and LCL. Since the bedrock of this region is very similar to the geology of UCL and LCL, this further supports the notion that it is unlikely that the high concentrations of Cl^- and Na are coming from geologic sources.

The comparison between Cl^- and the base metals indicates that anti-icing agents did not impact CP as markedly as UCL and LCL. The lower regression values of the Cl^- concentrations compared to the Na concentrations and the nonconformity with a one-to-one ratio all indicate that CP is impacted to a lesser extent than a highly impacted-lake, such as UCL and LCL.

Comparison of Current Conditions to Conditions Twenty Years Ago & Other Impacted Lakes

CP shows a much closer relationship to its 1986 counterpart than do either UCL or LCL. The may be due to the intense weather events that occur within the mountain pass where UCL and LCL are located. This pass creates blowing snow and intense squalls that requires NYSDOT to apply anti-icing agents frequently. After twenty years of this constant winter application, the Cl^- concentrations were expected to be elevated compared to concentrations found in the 1980s. Since CP's altitude is lower, the watershed is larger, and NYSDOT does not need to apply anti-icing agents as often, the Cl^- concentrations are expected to be much lower in this lake.

Chloride levels in lakes surrounding Chapel Pond (Giant Washbowl, Round Pond, and Upper and Lower Ausable Lakes) are low. The four additional lakes are found in the same region as UCL, LCL, and CP, but these lakes are not located along a major highway. Each of these lakes had lower Cl^- concentrations than those found in CP twenty years ago. This further indicates that a geologic input of Cl^- and Na is minimal and that the neighboring highway to UCL, LCL, and even that of CP has made the Cl^- and Na additions to each lake.

The comparisons of epilimnion Cl^- concentrations to hypolimnion Cl^- concentrations draw further conclusions about the state of each lake. CP showed very low Cl^- concentrations, yet the ratio of epilimnion Cl^- concentrations to hypolimnion Cl^- concentrations yield values that are approximately 0.6 during the spring/summer months and 1.0 during the winter months. This is indicative that during periods of stratification, the ratio will be lower. As a lake reaches isothermal conditions in the wintertime, the water column becomes more uniform in Cl^- concentrations and the ratio approaches a value of 1.0.

UCL displayed ratio values between 0.75 and 1.0 depending on the season. This indicates that the lake mixes and stratifies throughout the year. LCL showed a range between 0.75 and 1.0 as well, however, LCL reached its lowest ratio value much later in the year than UCL or CP. In addition, LCL did not display the similar annual cycle; the annual pattern for the ratio values was much more erratic. This indicates that the lake may be reaching a meromictic state.

In comparison to the ratio values from twenty years ago, LCL displayed a much wider range of values approximately from 0.25 to 0.95. This indicates that LCL may have undergone some physical changes over the past twenty years. UCL displayed a much lower annual ratio twenty years ago. The values remain fairly consistent between 0 and 0.1. This indicates that UCL may have also been impacted by the Cl^- loading since the relative difference is great. However, differences in sampling procedure and data collection may have influenced the Cl^- concentration ratio comparisons between the current study and twenty years prior.

In an assessment by Environment Canada and Health Canada (2001), Canadian lakes were measured for Cl^- concentration in order to evaluate the effects of sodium chloride on the environment at a national scale. The study defined that non-impacted lakes had Cl^- concentrations of less than 0.56 mmol/L. Elevated Cl^- concentrations for rural lakes impacted by road salt were defined as being between 4.23 and 8.46 mmol/L. Lakes such as Second Chain Lake, Halifax (4.79 mmol/L of Cl^-) and Chocolate Lake, Nova Scotia (5.85 mmol/L) are elevated in their Cl^- concentrations (Environment Canada and Health Canada 2001). UCL has concentrations that are on the border of being elevated in Cl^- and LCL has Cl^- concentrations that have already reached an elevated state. CP, however, is un-impacted based on the system defined by Environment Canada and Health Canada (2001). Based on these definitions, CP is the only lake of the three that has remained in its corresponding Cl^- concentration range for the past twenty years. This may be an indication that physical changes in UCL and LCL will most likely occur in the future should these increasing Cl^- concentrations continue to increase in magnitude.

Dissolved Oxygen Levels

The range of concentrations fell between 4 and 10 mg/L of dissolved oxygen in UCL, 2 and 8 mg/L in LCL, and approximately 2 and 11.5 mg/L in CP. In comparison to the national criteria for dissolved oxygen in freshwater, UCL has dissolved oxygen levels consistently above 4 mg/L. CP and LCL have times during the year where the dissolved oxygen concentrations drop below this national criteria, which indicate that aquatic organisms in these lakes may experience some stress (United States Environmental Protection Agency 1986). There was a slight decrease in the concentration with increasing depth, and it was observed that the highest concentrations were not necessarily found during the fall and winter months. In UCL, a similar trend indicated that there was also year-round oxygen being transported throughout the lake. CP and LCL appeared to contain less oxygen than UCL as their range of concentrations, from the surface to the maximum depth, was smaller and had a lower magnitude than that of UCL. The lower oxygen levels in LCL could be explained by a high biological oxygen demand due to the smaller hypolimnion volume.

Plankton Analysis

A large bloom of photosynthetic organisms may have occurred in the initial summer of this study. Each lake had their highest concentrations of Chl-a at the approximate depth of its thermocline. This was most likely due to the high concentration of nutrients present in the cooler hypolimnetic waters. The summer months had the highest concentrations of Chl-a, while the lowest concentrations were observed in the winter months. Phytoplankton experience stress from Cl^- concentrations at or above 49.2 mmol/L (Environment Canada and Health Canada 2001). The data currently suggests that the elevated concentrations of Cl^- and Na have not yet impacted the photosynthetic organisms in these lakes.

Lab pH Analysis

In UCL, LCL, and CP, the pH ranged from 6.5 to 8.5, with a general trend of decreasing pH with increasing depth. The trend of increasing Cl^- concentrations with increasing depth may be influencing the pH range of each lake, but other explanations such as biota and season appear to influence the pH more. The winter months appear to have a lower epilimnion pH while the spring and summer have a slightly increased epilimnion pH. This can be explained by the biota and season; as the sun warms the lake and photosynthetic activity increases during the summer months, CO2 is taken from the system by plants, resulting in a slightly basic water (Kalff 2002). This results in higher pH measurements in the mesotrophic lakes during the summer months. In comparison to the pH measurements made by the ALSC for CP (see section 3-5), the water was measured at pH 6.9. The seasonal average between July 2003 and January 2005 was pH 6.7. For UCL and LCL, the ALSC (1984) pH measurements were measured at pH 7.33 for each lake. The monthly average between July 2003 and March 2005 was measured at pH 7.1 for UCL and pH 7.2 for LCL. Since each lake has a circum-neutral pH (pH 6 - pH 8) and the pH values are similar to those measured in the 1980s, there does not presently appear to be an effect on pH caused by the elevated Cl^- concentrations.

Section 5-3. **QUANTITATIVE SAMPLING OF BENTHIC MACROINVERTEBRATES**

Report prepared by Tom A. Langen.

INTRODUCTION

Benthic macroinvertebrates (bottom-inhabiting insects, snails, crustaceans, annelid worms etc.) have been validated as sensitive indicators of water quality. There is a large body of research on macroinvertebrates as indicators of environmental health in flowing waters (rivers and streams) of northern temperate regions, and a smaller amount of research applied toward lakes. In environmental assessments, macroinvertebrates are worthy of attention not only because of their value at indicating water quality, but also because of their importance as food for animals higher on the food chain, particularly fish.

Assuming that samples are collected using a valid technique, there are numerous indices that can be used to compare samples to a standard reference water body or compare among water bodies in a sample. These metrics include general ones such as species number, abundance, and 'diversity' (as measured by a standard diversity index). They also include taxa – based metrics that use the taxonomic composition of samples, and which incorporate information on taxon-specific sensitivity to degradation of water quality. Lakes with good water quality will have a diversity of organisms (high species richness, high diversity index values), and have a good representation of sensitive insect groups such as *Ephemeroptera*, *Trichoptera*, *Plecoptera*, and *Odonata*. Lakes with impaired quality will be dominated by a few species, especially *Chironomidae*. Useful introductions to the use of benthic macroinvertebrates to assess water quality are found in Bode et al. (1991, 2003), Barbour et al. (1999), Rosenberg and Resh (1993), Lewis et al. (2001), and Mandeville (2002).

In this study, we compared the benthic macroinvertebrate communities of Upper Cascade Lake, Lower Cascade Lake, and Chapel Pond, sampled using a standard technique (Hester-Dendy samplers), and compared the lakes using various standard metrics.

METHODS

Ten standard (USEPA approved) Hester-Dendy benthic samplers (vendor: Wildco Inc.) were placed it each of the three study lakes. A Hester-Dendy sampler is comprised of fourteen thin, round plates (75 mm diameter) fastened to a bolt and separated by washers. When properly deployed, macroinvertebrates colonize the artificial substrate as a refuge and foraging surface. Hester – Dendy samplers have been criticized as attracting a biased sample of a water body's biota, but have been validated for use in making quantitative comparisons of abundance and species composition (Barbour et al. 1993).

At each of the three lakes, the samplers were deployed in chest – deep (1.5 m) water. Each sampler was hung vertically on a metal tomato stake, with the bottom of the sampler just touching the lake bottom. The samplers were deployed in a line transect, spaced 10 m apart, located along the shore adjacent to State Highway 78. Thus the three transects were located at the sites within lakes most likely to be impacted by salt runoff, sedimentation, and other impacts of roads. The samplers were in place for 54 days, from 17 June to 10 August 2004. At the end of the period, each sampler was carefully placed in a sealed bag, placed in a cooler, and transported to Clarkson University. Eight or nine samplers were recovered from each site.

Within 3 days of collection, each sampler was disassembled. The contents of the collection bag and sampler plates were placed in a sorting pan and carefully searched for animals. The plates were cleaned with a soft brush. All macroinvertebrates were placed in labeled vials containing 10% formalin for later identification.

Macroinvertebrates were examined under a dissecting microscope and identified. Identification was done by an expert, Dr. Andrew Casper (Laval University), who is very familiar with New York State macroinvertebrates. Thorp and Covich (1991) was also used as a reference key. All samples were labeled and stored as vouchers. The following were excluded from analysis: *Diptera* pupae, *Copepoda*, *Cladocera*, and unidentifiable insect parts.

To compare lakes, we used four metrics, based on recommendations in Lewis et al. (2001), and Mandeville (2002. (1) *Total taxa* = the number of differentiable types. Taxonomic resolution varied depending on the organism. (2) *Abundance* = the number of individuals. (3) *Inverse Simpson's index of diversity*. The larger the index value, the more diverse the community. (4) *EPT / C* = the abundance of *Ephemoptera* + *Plecoptera* + *Tricoptera /*

abundance of *Chironimidae.* Good water quality is indicated by a 1:1 ratio, whereas poor water quality is indicated by a disproportionately high abundance of Chironimidae.

RESULTS

There was little difference among the three lakes in terms of total taxa (Table 5-3-1, Table 5-3-2; Kruskal-Wallis Test: $H = 1.8$, $df = 2$, $p = 0.4$). However, the abundance of macroinvertebrates varied markedly among the lakes, with Upper Cascade Lake having a five-fold greater abundance than Chapel Pond (Table 4-3c-1, Table 3c-2; Kruskal-Wallis Test: $H = 13.3$, $df = 2$, $p = 0.001$). Lower Cascade Lake had a markedly lower EPT/C index, a lower Simpson's index, and higher weighted average tolerance value of taxa, all indicating that it had poorer water quality than Upper Cascade Lake and Chapel Pond (Table 5-3-1).

Table 5-3-1. Summary metrics comparing macroinvertebrate communities of the three lakes. N = the number of recovered Hester-Dendy samplers.

Metric	Upper Cascade	Lower Cascade	Chapel Pond
Abundance per sample	30.5 ± 5.74	19.2 ± 2.21	6.8 ± 1.53
Taxa per sample	4.5 ± 0.63	4.4 ± 0.41	3.6 ± 0.46
Total taxa	11	15	12
Total abundance	244	173	52
Simpson's Index	3.1	2.6	5.8
EPT / C	1.06	0.13	1.00
Weighted average tolerance	5.20	5.82	5.62
N	8	9	8

Table 5-3-2. Composition and abundance of macroinvertebrate taxa collected at each of the three lakes using the Hester – Dendy samplers.

Taxon			Upper Cascade	Lower Cascade	Chapel Pond	Tolerance
Ephemoptera	Heptegeniidae	Stenonema	100	14	12	4
Diptera	Chironomidae	Chironominae	92	104	14	6
Annelida	Naiadae	Styleria	24	18	0	6
Diptera	Chironomidae	Tanypodinae	12	0	6	7
Ephemoptera	Baetidae		4	0	1	6
Ephemoptera	Siphlonuridae	Siphlonurus	3	0	6	4
Annelida	Oligochaeta		3	0	0	8
Turbellaria			2	7	1	6
Tricoptera	Hydropsychidae		2	0	1	5
Odonata	Aeshnidae	Aeshna	1	3	1	5
Ephemoptera	Leptophebiidae	Paraleptophlebia	1	0	0	1
Diptera	Chironomidae	Orthocladiinae	0	10	0	5
Gastropoda	Valvatadae	Valvata	0	4	0	8
Gastropoda	Lymnaeidae	Fossaria	0	3	1	6
Gastropoda	Planorbidae		0	3	0	7
Acarina	Hydrocarina		0	2	1	6
Isopoda			0	1	7	8
Coleoptera	Elmidae	Ancronyx	0	1	0	5
Ephemoptera	Caenidae	Caenis	0	1	0	6
Gastropoda	Bithyiidae	Bithynia	0	1	0	8
Odonata	Libellulidae	Sympetrum	0	1	0	2
Megaloptera	Sialidae	Sialis	0	0	1	4

DISCUSSION

The results of this study indicate that each of the three lakes differ in terms of the composition of macroinvertebrates. Chapel Pond has the lowest abundance of macroinvertebrates, which is not surprising given that it is the least productive of the three lakes (see previous sections of this Chapter). Both Cascade lakes have a more abundant macroinvertebrate fauna, but in terms of composition Upper Cascade Lake is more similar to Chapel Pond than Lower Cascade Lake.

Applying the evaluation schema in Hilsenhoff (1987), the weighted average tolerance (also called the Hilsenhoff biotic index) of each lake indicated that Upper Cascade Lake had good water quality (some organic pollution present) whereas Lower Cascade Lake and Chapel Pond had fair water quality (fairly significant organic pollution).

Several caveats are necessary, however. The evaluation schema for the weighted average tolerance was developed for running waters rather than lakes; development of valid indices for lakes similar to the Adirondacks are still under development (Lewis et al. 2001). Moreover, all of the indices are developed for evaluating environmental stresses caused by organic pollution, not road salt contamination, sedimentation from sand, or other road – associated stressors. Although the macroinvertebrate indices indicate that Lower Cascade Lake is the most stressed of the three lakes, and this lake has the highest road salt contamination of the three lakes, it does not necessarily follow that road salt contamination is the stressor impacting the macroinvertebrate community. Other stressors include past high inputs of decaying vegetation from the surrounding forest and road side, organic material from the road, and past and current human recreational use of the Lower Cascade Lake headwaters.

Section 5-4. **EFFECTS OF RISING SALT CONCENTRATIONS ON PRIMARY PRODUCERS – AN EXPERIMENT**

Report prepared by Matthew Williams & Michael Twiss.

INTRODUCTION

Primary producers are the base of the food chain in lakes. Two main divisions of primary producers include free floating phytoplankton and substrate dwelling periphyton; each contains many species from several taxonomic groups. Community composition can change with alterations in water quality. Any community changes may affect higher trophic levels, including fish abundance and species composition.

The purpose of this study was to experimentally test effects of elevated road salt concentrations on the phytoplankton and periphyton of Upper and Lower Cascade Lakes, two lakes that already have elevated salt concentrations, but which are expected to increase in concentration in the future. Another nearby lake, Chapel Pond, was used as a reference lake, because it has similar characteristics but a much lower salt concentration.

METHODS

Phytoplankton

Sampling

Water samples were taken from Upper and Lower Cascade Lakes June 1 (Trial 1), June 14 (Trial 2), and July 9 (Trial 3) of 2004. Six liters of water were taken over the deepest parts of each lake at 4 meters depth with a niskin bottle. The water was then poured through a mesh screen (210 μm for Trial 1, 20 μm for Trials 2 and 3) to filter out grazers.

Experimental Setup

In the lab, water from each lake was poured into 24 – 250 mL polycarbonate bottles. Phosphorous was added in the form of 1μM K_2HPO_4. The bottles were organized into groups of 3 (Trial 1, 2) and groups of 4 (Trial 3). Then stock solution made from actual road salt was added to increase the sodium chloride concentrations of the samples, with increases ranging from 0 to 300 mg/L (see Table 5-4-1).

Table 5-4-1. Concentrations of increased NaCl for each treatment for all three trials of the Phytoplankton experiments

Trial	Salt Concentration Increase						
1	0mg/L	2mg/L	5mg/L	20mg/L	50mg/L	100mg/L	200mg/L
2	0mg/L	5mg/L	20mg/L	50mg/L	100mg/L	200mg/L	300mg/L
3	0mg/L	20mg/L	50mg/L	100mg/L	200mg/L	300mg/L	

The bottles were placed on a shaker in an incubator with fluorescent lighting set on a 15 h on: 9 h off cycle. Temperature was set based on the temperature of the water that the samples were taken from (Trial 1: 13˚C, Trial 2: 16˚C, Trial 3: 18˚C).

Analysis

The biomass of the phytoplankton was calculated at Days 0, 5, and 10 for Trial 1; Days 0, 4, and 10 for Trial 2; and Days 0, 3, 6, and 10 for Trial 3. Biomass was estimated by measuring the concentrations of chlorophyll *a* in the water samples. 50 mL from each sample was filtered through a 0.2 μm (pore size) filter (In Trial 3, an additional 50 mL from each sample was filtered through a 2.0 μm filter to distinguish between the nanoplankton (>2.0 μm) and the picoplankton (<2.0 μm)). The filters were then placed in acetone and left to extract overnight in a refrigerator. The concentrations of chlorophyll *a* were then measured using a fluorometer.

Periphyton

Sampling

Modified plastic slide holders (4 per lake) were loaded with acid cleaned slides and secured on wooden poles 1 m below the surface at Lower Cascade Lake and Chapel Pond June 14 (Trial 1) and July 9 (Trial 2) of 2004. The slides were left to colonize periphyton for 2 weeks and then the holders and slides were collected and transported back to the lab in ambient water. Eight liters of additional water from each lake was taken and filtered through a 0.45 μm cartridge filter using a peristaltic pump.

Experimental Setup

Twenty slides from each lake were divided into 4 groups of 5. Two groups were chosen to be placed in filtered water from the lake of their origin. One of these groups received an additional 100 mg/L (Trial 1) or 200 mg/L (Trial 2) of road salt. The other two groups were chosen to be placed in the filtered water from the other lake. One of these groups also received an additional 100 mg/L (Trial 1) or 200 mg/L (Trial 2) of salt. Each slide was placed in a 250 mL polycarbonate bottle filled with its respective water. Phosphorous was added in the form of 1μM K_2HPO_4. The bottles were then placed on a shaker in an incubator and received fluorescent light on a 15 h: 9 h cycle.

Analysis

The biomass of the periphyton was calculated at Days 0 and 7 of the experiment for Trial 1 and at Days 0 and 4 for Trial 2. Biomass was estimated by calculating the total chlorophyll *a* in the water samples. The slides from each bottle were placed in 50 mL plastic tubes containing 5 mL of 90% acetone. The slides were then crushed with a steel rod and an additional 5 to 10 mL of acetone was added to the tubes. The water in the bottles was filtered through 0.2 μm (Trial 1) or 0.4 μm (Trial 2) filters and the filters were soaked in 90% acetone overnight for extraction. The chlorophyll *a* concentrations were then measured using a fluorometer. Total chlorophyll *a* for each sample was calculated by adding together the contributions from the slides and the water that the slides were left in.

RESULTS

Phytoplankton

The chlorophyll *a* concentrations for Upper and Lower Cascade phytoplankton for Trial 1 are displayed in Figs. 5-4-1, 5-5-2. The corresponding data for Trial 2 and Trial 3 are displayed in Figs. 5-4-3, 5-4-4 and Figs. 5-4-5, 5-4-6 respectively. The biomass of the nanoplankton and picoplankton in Trial 3 is displayed in Fig. 5-4-7 (Upper Cascade) and Fig. 5-4-8 (Lower Cascade).

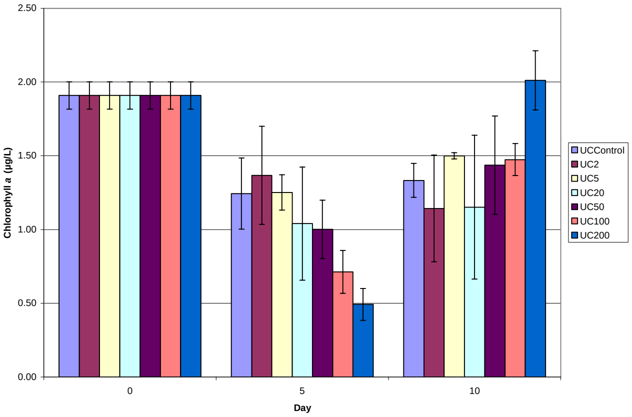


Figure 5-4-1. Trial 1. Upper Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Grazers <210 µm are still in the samples. Data are averages of 3 replicates. Error bars are 1 standard deviation away from the mean.

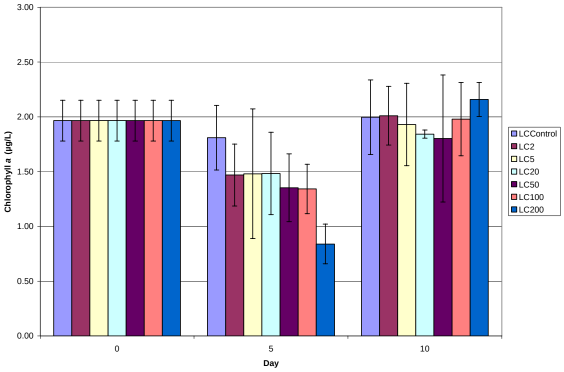


Figure 5-4-2. Trial 1. Lower Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Grazers <210 µm are still in the samples. Data are averages of 3 replicates. Error bars are 1 standard deviation away from the mean.

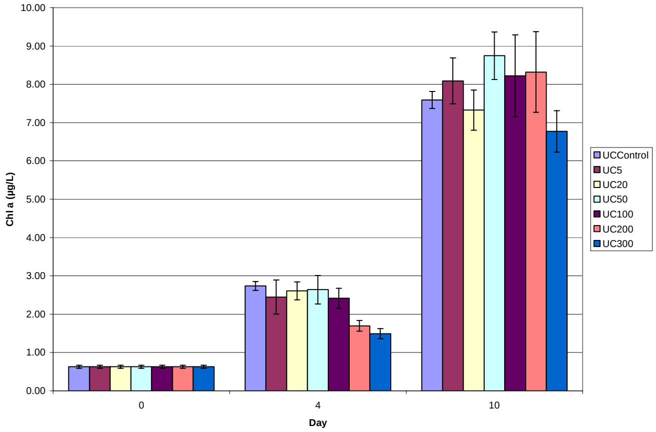


Figure 5-4-3. Trial 2. Upper Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Data are averages of 3 replicates. Error bars are 1 standard deviation away from the mean.

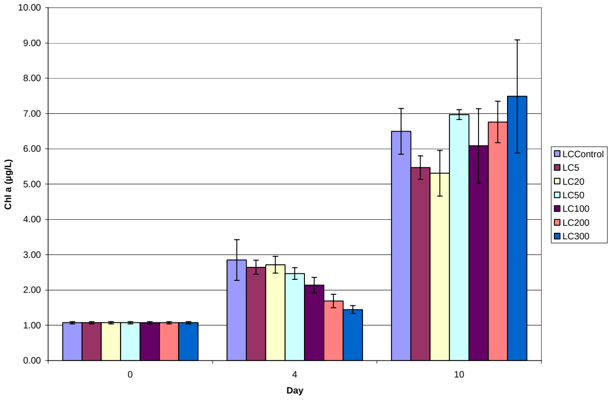


Figure 5-4-4. Trial 2. Lower Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Data are averages of 3 replicates. Error bars are 1 standard deviation away from the mean.

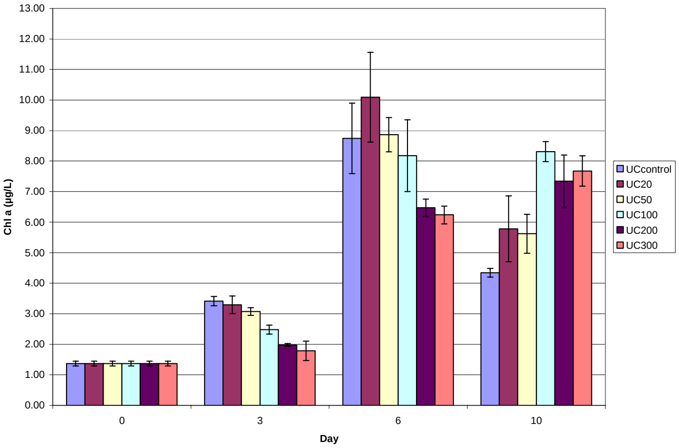


Figure 5-4-5. Trial 3. Upper Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Data are averages of 4 replicates. Error bars are 1 standard deviation away from the mean.

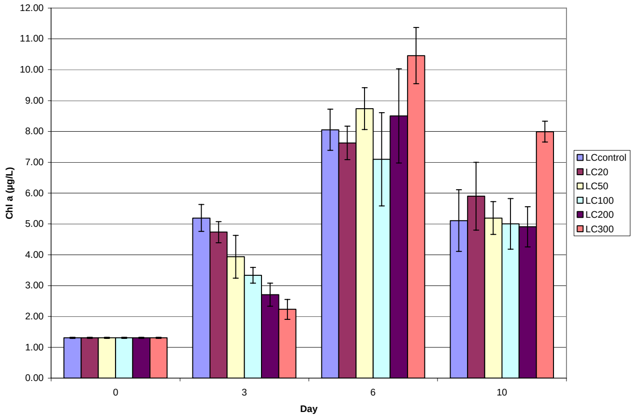


Figure 5-4-6. Trial 3. Lower Cascade Phytoplankton biomass expressed as concentration of chlorophyll a. Data are averages of 4 replicates. Error bars are 1 standard deviation away from the mean.

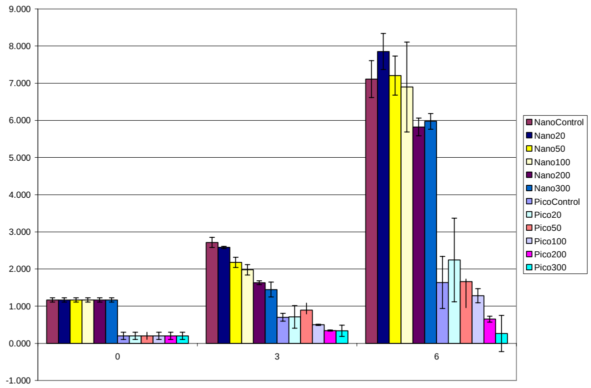


Figure 5-4-7. Trial 3. Upper Cascade Nanoplankton and Picoplankton biomass expressed as chlorophyll a. Data are averages of 4 replicates. Error bars are 1 standard deviation away from the mean.

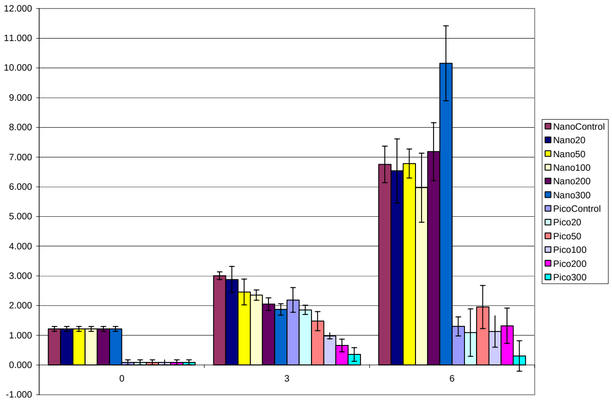


Figure 5-4-8. Trial 3. Lower Cascade Nanoplankton and Picoplankton biomass expressed as chlorophyll a. Data are averages of 4 replicates. Error bars are 1 standard deviation away from the mean.

Periphyton

The total chlorophyll *a* for the periphyton trials are displayed in Figs. 5-4-9 and 5-4-10. Separate chlorophyll *a* data for the slides and the surrounding water (not shown) have the same trends as the total chlorophyll *a* data.

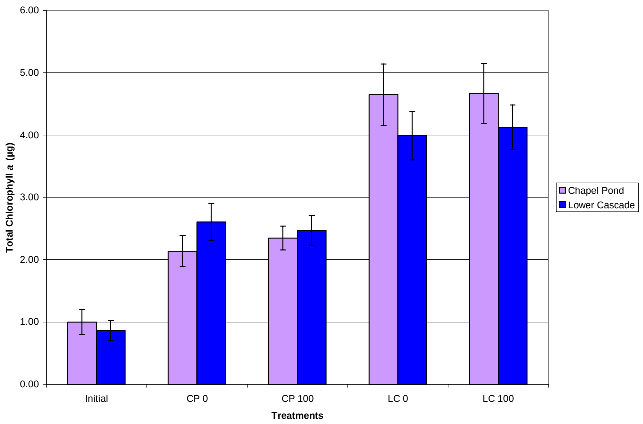


Figure 5-4-9. Trial 1. Periphyton biomass expressed as total chlorophyll a. Data are averages of 5 replicates. Error bars are 1 standard deviation away from the mean.

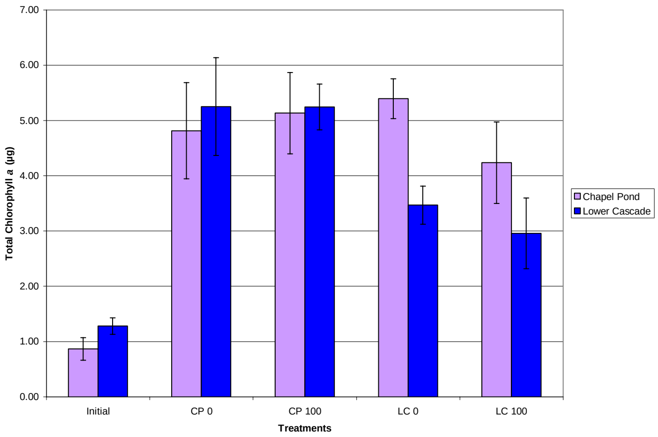


Figure 5-4-10. Trial 2. Periphyton biomass expressed as total chlorophyll a. Data are averages of 5 replicates. Error bars are 1 standard deviation away from the mean.

DISCUSSION

In the phytoplankton experiments (Figs. 5-4-1 to 5-4-8) some notable trends appear. On the second sampling of all 3 trials (Day 5 for Trial 1, Day 4 for Trial 2, and Day 3 for Trial 3), there is a decrease in biomass as salt concentration increases. At around the 100 to 200 mg/L level this difference seems to be significantly different from the control samples. This trend appears for samples from both lakes for all three trials. This suggests that the increase in NaCl concentration has some kind of negative effect on the growth of the phytoplankton within 3 to 5 days.

At the end of the experiment there was some variation. Sometimes the highest salt treatments biomasses would catch up to those of the control (Figs. 5-4-3, 5-4-4 on Day 10, and Fig. 5-4-5 on Day 6). Other times the highest salt treatments would reach higher biomasses than the control (Figs. 5-4-1, 5-4-2, 5-4-5 on Day 10, Figure 5-4-6 on Days 6 and 10). There was only one instance of the highest salt treatment samples having a significantly lower biomass than the control after 10 days (Fig. 5-4-3). In this case, however, the second highest salt treatment did overlap with the control. The similar patterns of the highest salt treatments catching up to the control and even surpassing its biomass (although more varied than the initial response at 3 to 5 Days) suggests that there is some shift or change occurring at or after the Day 3 to 5 sampling. The initial inhibition might be selecting for more salt tolerant organisms, which could lead to a higher net growth rate of these organism due to reduced competition. However, because of the general focus and constraints of this study, more information would be needed to test these hypotheses.

During Trial 3 of the phytoplankton experiments, samples were filtered through 2.0 μm filters as well as the 0.2 μm filters. With the chlorophyll *a* concentrations measured using both, chlorophyll *a* concentrations for the nanoplankton and the picoplankton could be calculated (Figures 5-4-7 and 5-4-8). The nanoplankton and picoplankton seem to follow trends similar to the total phytoplankton trends for Trial 3 (Figures 5-4-5 and 5-4-6). The picoplankton, however, seem to be more negatively impacted by the increases in NaCl. The difference between the highest salt treatment and the control is relatively much greater for the picoplankton than for the nanoplankton: 200-300% vs. 30-40%. In addition, only the nanoplankton at the highest salt treatments catch up to and surpass the control groups by Day 10. It is also important to note that regardless of treatment the nanoplankton greatly increase in biomass over the course of the experiment while the picoplankton even off or even decrease by the end of the trials. Part of this may be due to the added phosphorous which could boost nanoplankton as well as the possible grazing of the picoplankton by the nanoplankton. Overall this data suggests that the nanoplankton and the picoplankton have varying responses to increased salt concentrations but both groups are affected and do respond.

Follow up experiments and future studies of the phytoplankton could focus on the responses of specific taxa of phytoplankton and monitor the relative contributions of each taxon to the community after the addition of road salt. A closer investigation of the effects of grazing would also be important. If there is a selection for salt tolerant organisms in the phytoplankton the effects on the grazers and on the rest of the aquatic community should be considered and studied. It would also be useful to run similar experiments with samples from a relatively salt free lake such as Chapel Pond and compare the results to see if the high salt levels in the Cascade Lakes have already had an impact on the phytoplankton.

In the periphyton experiments (Figure 5-4-9) the 100 mg/L increases of NaCl did not seem to have a significant effect on biomass. The periphyton, regardless of origin, did better in Lower Cascade water than in Chapel Pond water and the salt additions did not seem to have an impact. In Trial 2 (Figure 5-4-10) the increased salt concentrations (200 mg/L) did not seem to have an impact for organisms put into Chapel Pond water. There did however seem to be a difference in the biomass between the control and 200 mg/L salt groups for Lower Cascade water. But due to the high variation, the results are not likely to be statistically significant. Follow up experiments with more replicates could improve this. The data from both trials suggests that the periphyton are not as sensitive to salt increases as the phytoplankton, which seem to initially respond at the addition of 100 mg/L and have a significant response at the addition of 200 mg/L. The periphyton begin to show a response at the addition of 200 mg/L but it is not statistically significant.

As with the phytoplankton, the periphyton experiments focused on the general response of a broad category of organisms. It would be useful to also look at specific taxa of periphyton and their individual responses to increased salt concentrations as well as the responses of grazers and higher trophic levels.

CHAPTER 6

MODELING THE TRANSPORT AND FATE OF DEICING SALT-DERIVED CHLORIDE IN THE CASCADE LAKES

View of Route 78 and south end of Lower Cascade Lake, showing salt-contaminated snow and briny melt water moving onto and under the lake ice. Photo by Michael Twiss

Chapter 6. **MODELING THE TRANSPORT AND FATE OF DEICING SALT-DERIVED CHLORIDE IN THE CASCADE LAKES**

Report prepared by Thomas Young, with contributions from Joseph Osso Jr., Nicole Reed and Lindsay Hoffman.

GENERAL INTRODUCTION

The use of a modeling approach to the assessment of the environmental fate of road salt has the advantage of allowing and extension of knowledge about the affected system beyond the bounds observed from data collected from the system. If done with realistic assumptions and accurate information, then a mass balance model can usefully permit exploration and testing of possible management actions and provide support for future management decisions. Key to usefulness, however, is the accuracy of the information about the system.

The modeling activity on this project was initiated during the first year of this project, when considerable effort was directed at acquiring modeling information of such a level of accuracy that it could not be improved upon without an unreasonable investment of resources. The main information needs satisfied during the first year, as described below, included defining the hydrologic and hydraulic characteristics of the two-lake system, developing chloride loading estimates from deicing chemical application data, and obtaining preliminary information on the behavior of chloride mass and concentration within the system.

During the second year the efforts shifted toward iterative model development, generalizing the hydrologic and loading information, calibration, identification of uncertainties, and application of the calibrated model to possible future loading scenarios. These efforts and their results and described in this chapter.

Section 6-1. **LAKE HYDROLOGY**

PROBLEM STATEMENT AND GENERAL APPROACH

Previous studies of the bathymetry of the study lakes (Adirondack Lake Survey Corporation (ALSC), methodology and data available at http://www.adirondacklakessurvey.org/; and our own unpublished reanalysis of the ALSC data) revealed considerable uncertainty over their actual surface areas and volumes. Accordingly, work was completed during the current year that sought to provide accurate estimates of these quantities with known uncertainties. This was accomplished using a combination of global positioning system (GPS) technology to collect location information, a high-precision sonar to acquire depth information, and state-of-the-art spatial analysis software (i.e., a geographical information system, GIS) to facilitate synthesis and interpretation of the collected information. Once the appropriate lake information had been developed, computations were performed of substance mass for each lake and sampling period (e.g., chloride).

OBJECTIVES

- Gather recorded x, y, and z points in space and create a database for importation into ArcGIS to develop a two-dimensional (x,y) representation of the lakes (e.g., contour maps).
- Execute interpolation procedures in order to create a realistic three-dimensional spatial model and continuous raster depth visualization of both Upper and Lower Cascade Lakes.
- Perform area and volume calculations for each lake; estimate volumes at 0.5 meter intervals using the 3D Analyst extension in ArcMap.
- Incorporate monthly temperature vs. depth and chloride concentration vs. depth measurements with segment volumes in order to provide input on chloride masses and concentrations in the lake over time as appropriate for modeling purposes.

METHODS

Data Collection

X and Y positions (North and West) were obtained using a handheld Garmin global positioning systems (GPS) device as repeated transects were rowed from shore to shore and around the perimeter of Upper and Lower Cascade Lake. In addition, GPS points were taken for a connecting stream that flows from Upper to Lower Lake. The GPS device used the WGS 84 coordinate system that measured global position in decimal degrees at an accuracy of about five meters. The depth at each point was also measured using an Eagle Sea Finder DF sonar attached to the back of a rowboat, displayed in meters. These three coordinates were obtained and recorded simultaneously by hand in a research notebook and subsequently entered into an Excel spreadsheet listing x, y, and z coordinates as they were gathered in the field and later exported into a Microsoft Access database displaying the same information.

Defining Coordinate System

To perform any calculations/analysis once a map is created, a z-factor must be found that relates x and y coordinate units to z units, assuming x and y are the same. Using the WGS 84 coordinate system, one degree latitude (y or N) differs from one degree longitude (x or W), which differs from one z unit (m). In order to correct for this problem, the entire table can be converted using ArcToolbox from the WGS 84 coordinate system into a projected UTM WGS Zone 18N coordinate system, which translates every unit into meters. This conversion eliminates the need for a z-factor and allows volume and area calculations to be performed.

Creating polylines and polygons

Polygons are useful for simply showing the shapes of the lakes. They are also needed to define the space over which the interpolation will be evaluated. In order to create a polygon or polyline, one may use an ET GeoWizards free downloaded extension or create them manually with the ArcMap editor tool. Both applications were used interchangeably when creating polygons of the lakes.

TIN creation

Once the depth points have been imported into ArcMap, a triangulated irregular network (TIN) can be generated that visually displays the three-dimensional depth surface of the lakes in two dimensions. A TIN is essentially made up of “contiguous nonoverlapping triangular elements” that connect original data points (Longley et al. 2001). This connection is linear, which is advantageous for checking accuracy but provides no realistic smoothing of the lake bottom surface. Because of the TIN capability and simplicity, the model is “only as good as its input samples”, but this is true for most methods of estimation (Longley et al. 2001). Using the 3D Analyst extension, a TIN is created from the x, y, z table. To be certain that the network only falls within the boundaries of the lake perimeter, the TIN was modified to include the lake polygon shape as a hard clip.

Surface Interpolation

Because it is impossible to sample at every location, interpolations of the lake surfaces can be executed to create a continuous raster representation of the depths using both the 3D Analyst and Spatial Analyst extensions. The basis for any interpolation is founded upon Tobler’s First Law of Geography: “Everything is related to everything else, but near things are more related than those far apart” (Longley et al. 2001). Both Inverse Distance Weighing (IDW) and Kriging were run for this purpose. An analysis mask of the lakes’ polygon shapes was chosen in the Options menu before each interpolation was performed in order to ensure that only those points within the lake were analyzed.

Inverse Distance Weighing is the method most commonly used by GIS analysts because of its “ease and simplicity” (Longley et al. 2001). Weights are given to points based on their distance and the power given to this distance. In most cases, a power of two is assigned meaning that the weights decrease by the inverse square of the distance. This method generates better results when more points are taken, and is advantageous for including known values because these points are given a weight of one and not distorted in any way. Because the weights can never be negative, the only setback to the method is that no interpolated point can be deeper than the deepest sampled point (Longley et al. 2001). Because of the amount of points taken over each lake, this was not a significant concern for this method.

The other interpolation method that utilizes both statistical and mathematical evidence to create a smooth surface is Kriging. Not only does Kriging use weights of surrounding points to determine unknown depths by distance, it also draws upon the "overall spatial arrangement among the measured points" (ESRI 2002). This is a more involved method that gives an array of options to the user. These options were not so important for the purposes of this study. Results generated by this method using default parameters were similar to those produced by the IDW technique.

Volume

Since every map unit is in meters, the z-factor is left at a default value of one. A base height of zero meters is chosen to obtain a total volume for each lake. The surface area is also evaluated simultaneously using this tool. The volume and area estimates were saved as a text file for further analysis. In order to determine the lakes' volume at 0.5 meter intervals, the base height was set at increasing 0.5 meter levels and the area/volume tool was executed. Each volume was saved to a text file and simple subtraction at each level allowed individual volumes to be determined for each lake.

Temperature Profile

A temperature vs. depth profile was created for each lake after monthly measurements were taken at one-meter intervals. From these profiles, it was determined which lakes were stratified or isothermal for each month. For stratified lakes, the thermocline was found using the graphs to find each lake's epilimnion and hypolimnion. The volumes of each layer were calculated using an Excel spreadsheet listing the depths at 0.5 meter intervals.

Chloride Mass

Chloride concentration measurements were also taken monthly at six discrete depths within each lake and were given in units of ppm, or mg/l. The sample depths varied from month to month but were usually distributed evenly and taken from each layer of the lake. The total mass of chlorides within the lake can be found from the mass in individual layers, or segments, using Equation 1:

$$Total\,Mass = \sum Layer\,Cl\,Mass = \sum LayerVolume \times Layer\,Cl\,Concentration \qquad (1)$$

Differences in concentration vs. depth were taken into account since volumes were found for individual lake segments. Depending on temperature profiles, the mass was found for either the entire lake or within its epi- and hypolimnion. Chloride mass was displayed in metric tons to make it easier to compare to annual salt load data provided by the New York Department of Transportation.

Average Chloride Concentration

It may appear as if the total chloride mass is shifting from one layer to the other with time, but the volumes for each layer may also be shifting. As a consequence of this fluctuation and also the lake segment volume changing with depth, average concentrations of Chloride were calculated for the epi- and hypolimnion using Equation 2:

$$AverageConcentration = \frac{Layer\,Cl\,Mass}{Layer\,Volume} \qquad (2)$$

RESULTS

Volume and Surface Area

Volume and surface area estimates for Upper and Lower Cascade Lakes, based on the three objective approaches to analysis of the GPS and sonar data (i.e., TIN, IDW, and Kriging), are shown in the following tables. In addition, the values are given for these measures as reported for both Upper and Lower Cascade Lake by the Adirondack Lake Survey Corporation through their on-line database, which is accessible via the ALSC website (http://www.adirondacklakessurvey.org). The agreement between kriging and inverse distance weighting, methods known to be accurate for spatial interpolation, is shown below in a small table below the summary tables.

Upper Cascade Lake Results

Method	**Volume (m^3)**	**Surface Area (m^2)**
TIN	1287274	107487
IDW	1324709	106360
Kriging	1304510	106421
ALSC [4]	1144425	97000

Percent Difference (Krige/IDW)	
Volume	1.52%
Surface Area	0.06%

Lower Cascade Lake Results

Method	**Volume (m^3)**	**Surface Area (m^2)**
TIN	542860	124085
IDW	589594	113383
Kriging	599124	113646
ALSC	345522	102000

Percent Difference (Krige/IDW)	
Total Volume	1.59%
Surface Area	0.23%

Volume Percent Difference (between IDW and Kriging)	
Upper Cascade	1.52%
Lower Cascade	1.59%

Depth Models

In addition to the total area and volume estimates given above, the depth-spatial data for the two Cascade lakes also were used to construct bathymetric maps for these systems, which are shown in Fig. 6-1-1 and Fig. 6-1-2. The maps were constructed using ArcGIS 8.2.

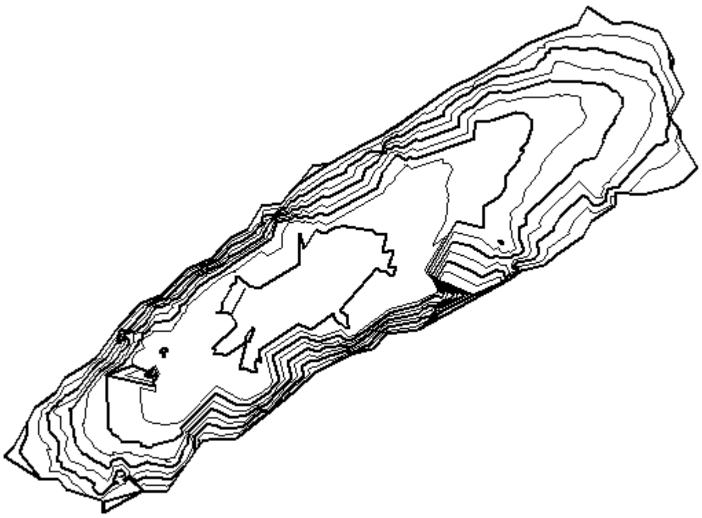


Fig. 6-1-1. Upper Cascade Lake bathymetry, contour interval = 2.5 m

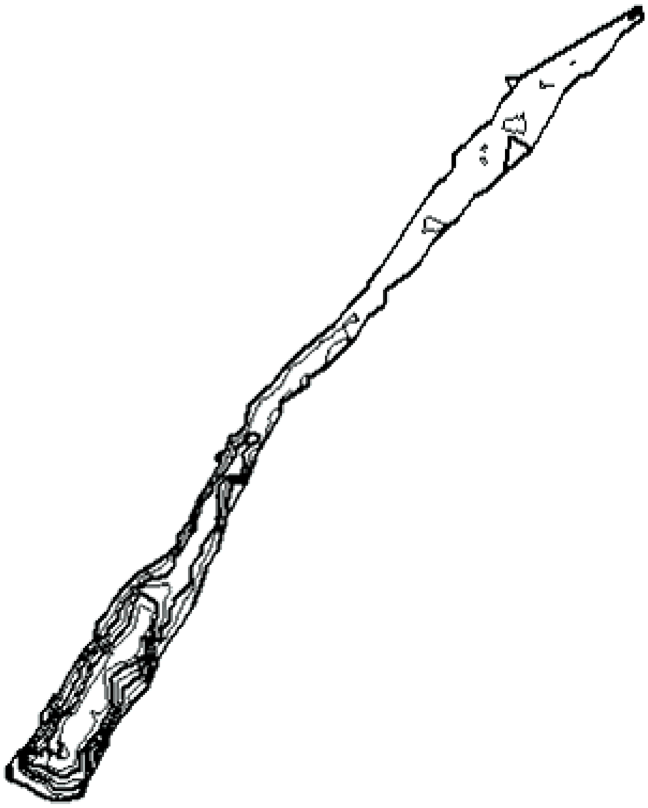


Fig. 6-1-2. Lower Cascade Lake bathymetry, contour interval = 2.5 m

The mapped data used to develop the bathymetry of the two Cascade lakes was analyzed further to yield the vertical distribution of lake volume for the two lakes. The results of these analyses are shown in in Fig. 6-1-3 and Fig. 6-1-4.

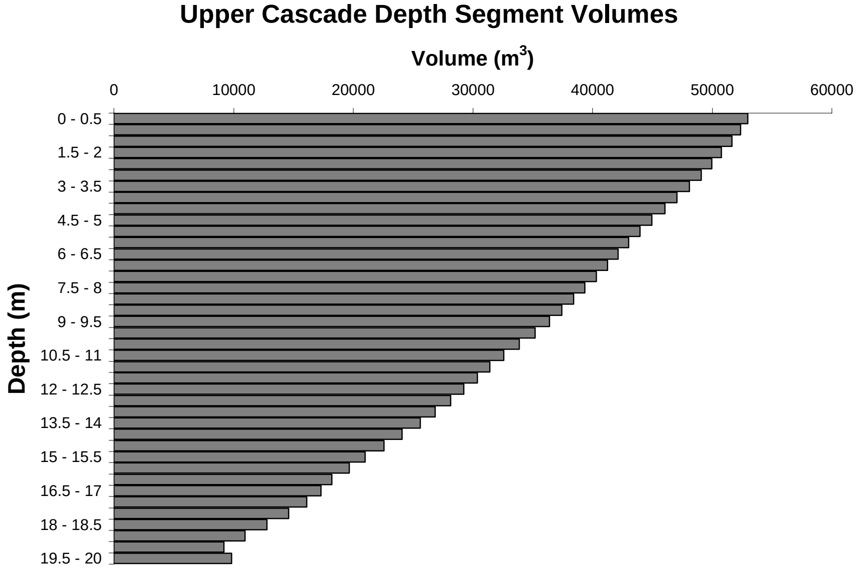


Fig. 6-1-3. Upper Cascade Lake vertical lake volume.

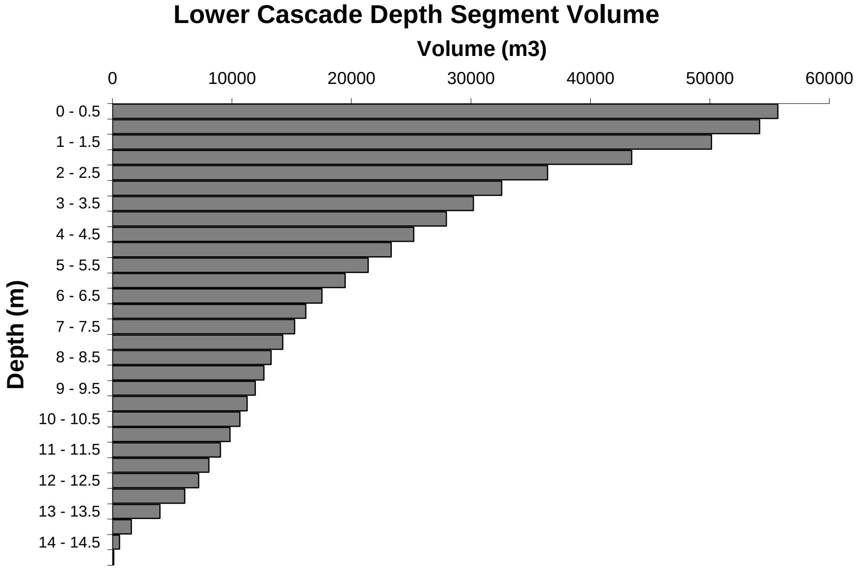


Fig. 6-1-4. Lower Cascade Lake vertical lake volume.

The data shown in Fig. 6-1-3 and Fig. 6-1-4. are particularly useful for computing estimates of the mass of substances, e.g., Cl, and heat within the water columns of the respective lakes using the relationship presented previously as Equation 1.

Estimates of whole-lake Cl mass at different points in time over the study period were obtained by applying Equation 1 to the depth data in the previous figures and the Cl concentration data for the sampling periods as reported in an accompanying section. The results of these computations are shown in Fig. 5-1-5 and Fig. 5-1-6.

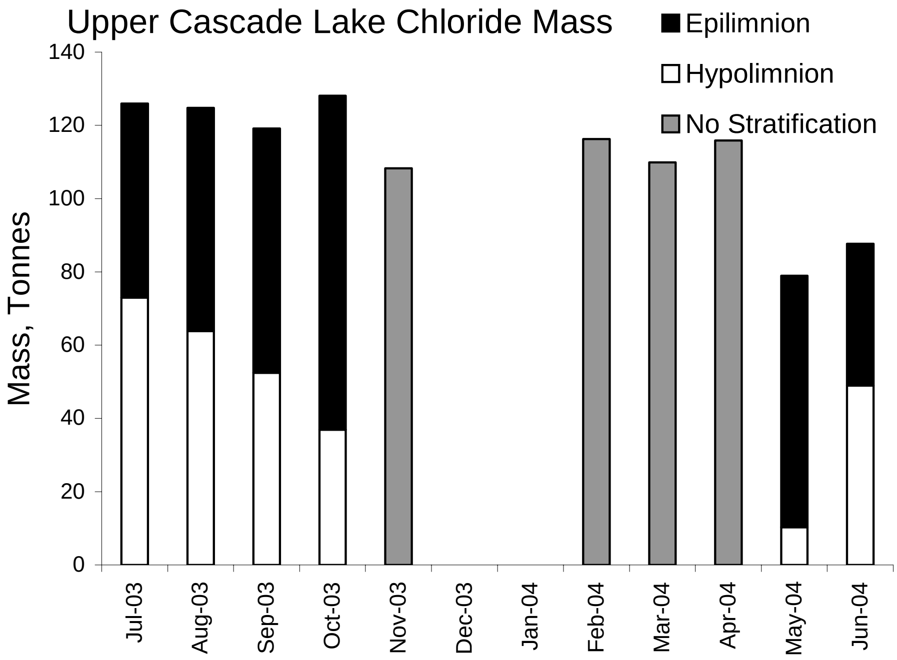


Fig. 6-1-5. Upper Cascade Lake chloride mass.

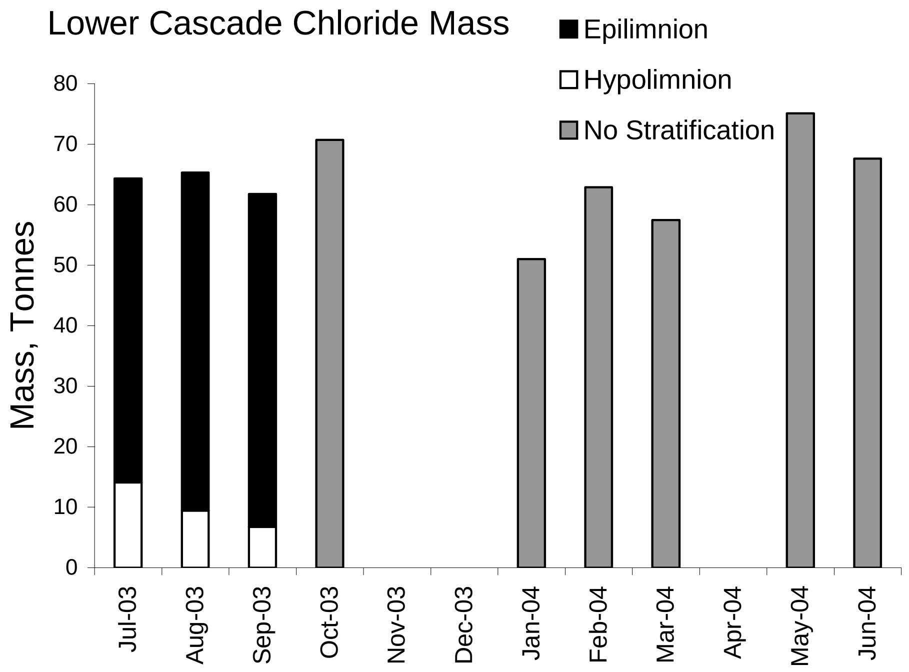


Fig. 6-1-6. Lower Cascade Lake chloride mass.

Whole-lake average Cl concentration estimates at different points in time over the study period were obtained by applying Equation 2 to the depth data in the previous figures and the Cl mass estimates for the sampling periods as estimated and reported above. The results of the average concentration computations are shown in Fig. 6-1-7, 6-1-8.

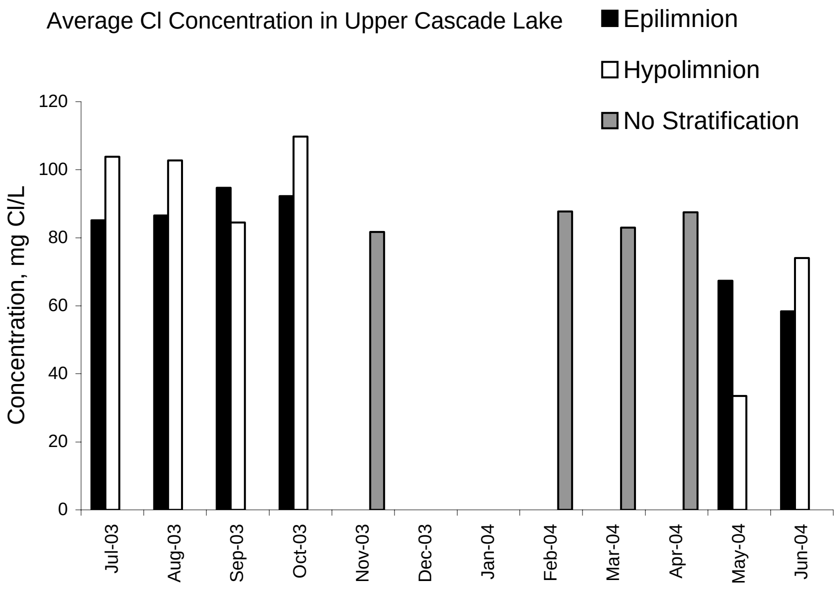


Fig. 6-1-7. Upper Cascade Lake mean chloride concentration.

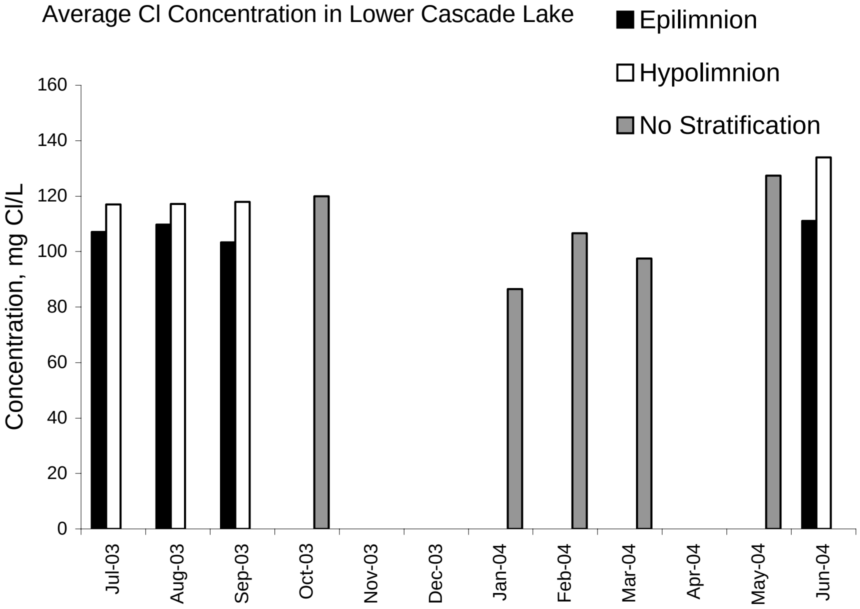


Fig. 6-1-8. Lower Cascade Lake mean chloride concentration.

DISCUSSION

The foregoing analysis of bathymetry and related information plus the application of these data to develop estimates of lake-wide chloride mass and concentration were based on a large amount of information (depth and location) for each lake. In addition, the results displayed small percent differences between methods with respect to estimates of total volume (e.g., between the Inverse Distance Weighing (IDW) and Kriging methods), which boosts confidence in the *results* obtained from the interpolation procedure (1.52% and 1.59% for Upper and Lower Cascade, respectively).

For Upper Cascade Lake, several regions were not sampled as thoroughly for depth and location information as in other regions, but this did not appear to compromise the estimations in any way. If the future provides opportunity, it may be advantageous for improving the smoothness of the interpolation to sample depth and location information with greater intensity and along two or three transects perpendicular to the initial pattern of sampling tracks. The results from doing so would yield more confidence in the model and permit production of a more detailed contour map when needed.

For both lakes, the Triangular Irregular Network (TIN) models produced volume estimates that were consistently lower than both the IDW and Kriging but this is to be expected. The TIN model is simply a linear connection of known sample points and since there is no smoothing of generated lake surfaces, one could expect a smaller total volume. For this reason, and because Inverse Distance Weighing is a simple and straightforward method that produced results similar to those of a more complicated procedure, Kriging, the IDW approach was used to determine each lake's individual segment volumes. When looking at the graph of segment volume vs. depth, one can easily see the general shape of each lake and the difference between the two. Upper Lake is a smaller, circular lake that mimics this shape on its bottom with gradual sloping boundaries, while Lower Lake has a vast majority of its volume in the first few meters and then sharply drops off in volume as depth increases. These profiles may be useful for determining the flow, accumulation, or behavior of chlorides in the lake.

The chloride mass and average concentration graphs from the first year of observation illustrated a key point, which became important to understanding some of the more obscure aspects of the overall hydraulics of the Cascade Lakes watershed. Salt applications for snow and ice control in the Cascade Lakes corridor did not appear to cause the chloride concentration or mass in the lakes to change in a way that reflected the occurrence of a dynamic episodic response during or immediately after the winter months with the onset of snowmelt and ice-out, or at least not dramatically so. Thus, it appears that the chloride load that results from road salt applications does not present itself as a shock or impulse load to the lakes. Rather, the resulting loading is delayed – the loading has a significant temporal component. This apparent retardation may not be entirely unexpected as frozen soils and later soil and vegetation may contribute as factors that act to delay the loading. Nevertheless, the magnitude of the effect suggests the functioning of a more complex loading scenario for assessing de-icing salt transport mechanisms in the Cascade Lakes environment than previously considered. The salts appear to move along a flowpath that includes a reservoir for temporary storage in transit to the lake.

Section 6-2. **WATERSHED HYDROLOGY – DIRECT AND INDIRECT DISCHARGE DETERMINATION**

STATEMENT OF PROBLEM AND APPROACH

A fundamental information requirement that must be met before it will be possible to assess and understand the response of the Cascade Lakes to de-icing salts and, furthermore, to allow development of the capability of predicting the response of the lakes to future salt loadings, is adequate knowledge of the hydrologic characteristics of those watersheds. Because the lakes are hydraulically connected (Fig. 6-2-1), continuity permits use of the outflow from the upper lake, which may be determined directly using common hydrologic monitoring methods, to estimate a component of the inflow to the lower lake. The outflow from the lower lake, in turn, may also be estimated directly using the same common hydrologic monitoring methods.

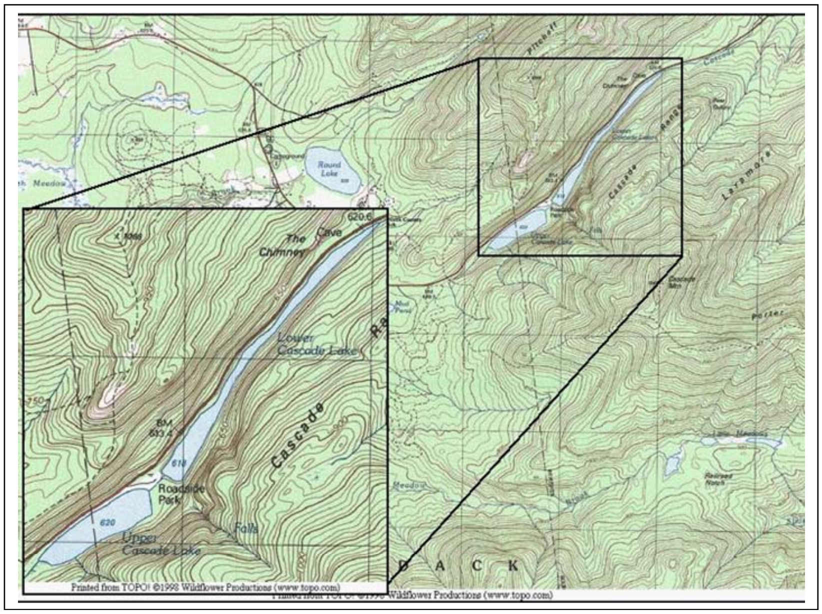


Fig.6-2-1. Topographic map of the Cascade Lakes region, showing hydrological connections.

Approximately monthly during ice-free conditions over the reporting period, the following procedure was employed in order to estimate the discharge from the Upper and Lower Cascade Lakes. The basic approach used to develop the estimates involved measuring the quantities needed to make the calculation shown in the following equations, which is a simplified expression of continuity:

$Q = v \bullet A$

Where: Q = *discharge* ($L^3 T^1$),

v = *velocity of water flow* ($L T^1$),

A = *cross-sectional area normal to the flow velocity* (L^2).

The required measurements were obtained at transverse cross-sections, which were established at points of constricted flow on the outflow streams from the Upper and Lower Cascade Lakes. The depth distribution at each cross section was measured using a steel tape and meter stick. Measurements of the velocity distribution over each cross-section were taken using a Marsh-McBirnie electromagnetic velocity meter, which previously had been calibrated by towing at constant speed in a wave tank.

MONITORING RESULTS

As the results of these discharge determinations were collected over time, two observations emerged: (1) a useful discharge rating curve could be developed for the Upper Cascade Lake outflow based on data collected on the discharge cross-section, and (2) a degree of correlation existed between the outflows of the Upper and Lower lakes.

The rating curve, using all data collected during the reporting period, is shown in Fig. 6-2-2. It should be noted that the connecting channel between the Upper and Lower Lakes is strewn with cobble and small boulders, which make depth measurements critically dependent on relocating the same position for each determination. This could not be done with complete certainty, which accounts for much of the uncertain shown in the rating curve.

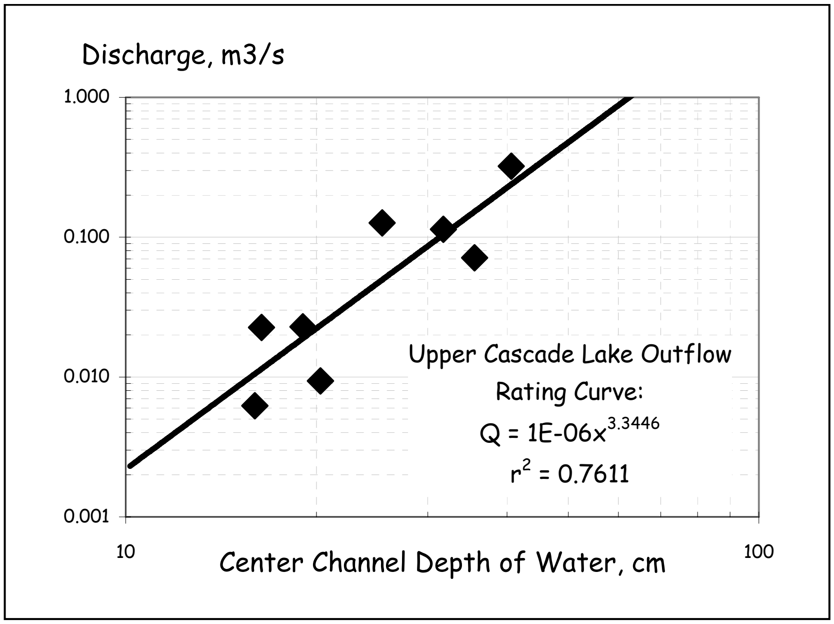


Fig.6-2-2 Outflow rating curve for Upper Cascade Lake.

This relationship between the outflows of the two lakes was found to be remarkably strong, as shown in Fig. 6-2-3. This figure is a scatter diagram that shows the discharge from the outlet of the Lower Cascade Lake as a function of the discharge from the Upper Cascade Lake, which is the inflow to the former system. It is clear from these data that the two lake discharges are highly correlated.

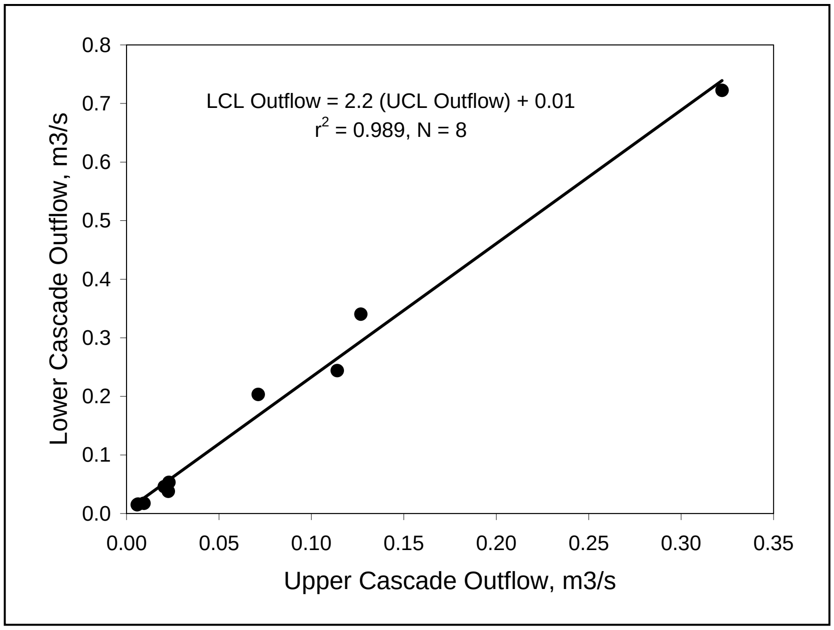


Fig.6-2-3. Association between the outflow at Upper Cascade Lake and Lower Cascade Lake.

Using the information from the rating curve and the correlation between the two lakes' outflows, it was possible to estimate flows on several dates when water sample collections were made from the two lakes, but discharges were not measured. The entire set of discharge estimates for the period of record for the two lakes is shown in Fig. 6-2-4.

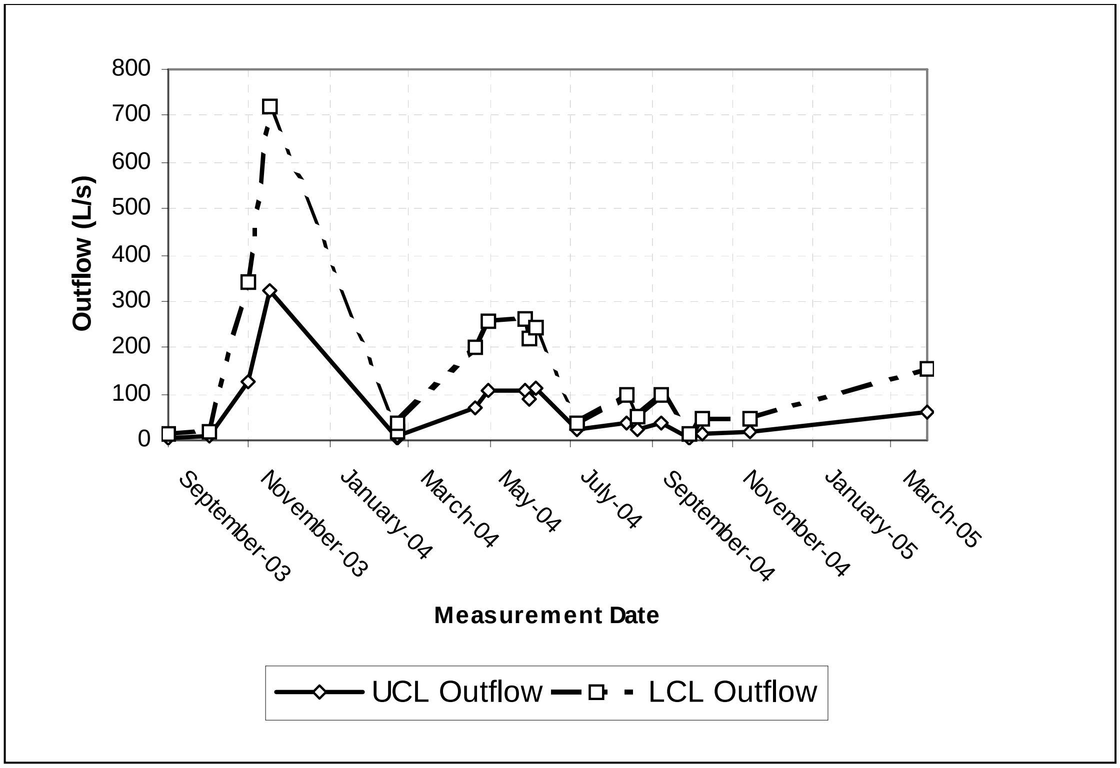


Fig.6-2-4. Discharge estimates of Upper Cascade Lake and Lower Cascade Lake during 2003 - 2006.

Section 6-3. **WATERSHED HYDROLOGY – RUNOFF ESTIMATION**

STATEMENT OF PROBLEM AND APPROACH

Beyond the directly measurable discharges for the two lakes, it was recognized that other flows represented potentially significant pathways that could be involved in the transport of de-icing salts either to or from the lakes. Examples of these would include episodic runoff flows, snow/ice melt events, evapotranspiration, direct precipitation, and groundwater. To estimate the significance of these flows in the aggregate, an analysis of the potential for landscape contributions to the hydrology of the lakes was undertaken using standard estimation methods.

Runoff estimates were obtained using two methods: the Rational Method and the SCS Curve Number Method (Soil Conservation Service). Both methods are commonly used to calculate runoff, but are based on different qualities pertaining to the areas of interest. The Rational Method depends on the drainage area of the lakes and the intensity of precipitation over the time period. The SCS Curve Number Method is based on the amount of rainfall over a period of time and the amount of water the ground is capable of holding. With both of these methods, it is possible to account for direct precipitation onto the lakes. The latter was estimated from the average annual precipitation at the closest NOAA weather station (SLK, Saranac Lake airport), 39.83 in yr^{-1}, or 1.0 m yr^{-1}, and the surface area of each lake. Using this approach, the direct rainfall contributions were 0.003112 and 0.003272 $m^3\ s^{-1}$ and for Upper and Lower Cascade Lake, respectively. It is important to note that these approaches were developed primarily for application to agricultural situations and for short-term events. Consequently, their applicability may not extend well to steep-sloped forested environments as that of the Cascade Lakes watersheds and average annual runoff estimation.

RATIONAL METHOD

The formula for the Rational Method is:

$$Q = C \bullet C_f \bullet i \bullet A$$

Where: Q = total runoff for the time period, $[L^3 T^{-1}]$
C = a coefficient based on soil classification,
C_f = a frequency factor for major storm events,
i = the intensity of rainfall over that time period, $[L T^{-1}]$
A = the drainage or watershed area, $[L^2]$

The frequency factor (C_f) was taken as 1.0 for the Cascade Lakes system, because the return rate for storms of interest was assumed to be less than 25 years. The intensity (i) was calculated by dividing the annual rainfall by one year and was found to be 0.004547in/hr. The coefficient (C) and drainage area (A) required information about the soils in the watershed, which vary widely, as noted in another section of this report. Estimates of the drainage area using ArcGIS had not yet been performed by the time the estimates presented here were developed. Instead, the required data were obtained from the ALSC (Adirondack Lakes Survey Corporation): 230.9 ha and 504.7 ha for Upper Cascade Lake and Lower Cascade Lake, respectively. Exclusive of the watershed of Upper Cascade Lake, the watershed for Lower Cascade Lake was 273.8 ha.

For the Rational Method, soils are classified into the following types:

Type A is deep sand; deep loess; silts
Type B is shallow loess; sandy loam
Type C is clay loams; shallow sandy loam; soil low in organic content; soil high in clay
Type D is heavy clay; saline soil; soil that swells significantly when wet

The type of soil in the Cascade Lakes watershed is variable, but most of the soil is Type B. Near the roadway, however, the soil is most like Type A because of all the sand applied in the past for deicing. Consequently, calculations were performed for Type A, Type B, and a combination of the two, Type A/B. The coefficients for these types are 0.11, 0.18, and 0.14 respectively (McCuen 1989).

SCS CURVE NUMBER METHOD

The SCS Curve Number Method for runoff estimation employs the following equation:

$$Q= [(P-I_a)^2]/[(P-I_a)+S]$$

Where: Q = the total runoff,
P = potential maximum runoff,
I_a = the initial abstraction,
S = storage.

Each term in the SCS equation have units of precipitation depth per year [$L\ T^{-1}$].

The potential maximum runoff (P) is equal to the amount of rain that falls annually, which was assumed to be 39.83 inches in the case of the Cascade Lakes. The initial abstraction (I_a) is the amount of rainfall that cannot go to runoff- some portion of the rainfall will always be absorbed into the ground. Initial abstraction is often assumed to be equal to 20% of the storage (S) unless other information suggests another value would be more appropriate. Storage can be from the SCS Curve Number that is selected for the system, and the Curve Number, itself, depends on the soil classification, land use, hydrologic condition, and management practice, The values of 30 and 55 were selected as representative of Type A and Type B soil types on steep sloped forested watersheds, respectively.

RESULTS AND DISCUSSION

Runoff Estimates and Correlation with Observations

The results of applying the Rational Method to the Upper Cascade Lake watershed were 0.0081 $m^3\ s^{-1}$, 0.0133 $m^3\ s^{-1}$ and 0.0104 $m^3\ s^{-1}$ for soil types A, B, and A/B, respectively. The values for the Lower Cascade Lake watershed amounted to 0.0178 $m^3\ s^{-1}$, 0.0291 $m^3\ s^{-1}$ and 0.0226 $m^3\ s^{-1}$ for soil types A, B, and A/B, respectively.

The annual precipitation runoff amounts obtained by applying the SCS Curve Number Method to both of the Cascade Lakes were 0.537 m yr^{-1} and 0.799 m yr^{-1} for soil types A and B, respectively. These results may be expressed as a volumetric runoff rate by multiplying by the surface area of each watershed, which yields 0.0387 m^3 s^{-1} and 0.0576 m^3 s^{-1} for soil types A and B for Upper Cascade Lake, and 0.0846m^3 s^{-1} and 0.1260 m^3 s^{-1} for soil types A and B for Lower Cascade Lake (Fig. 6-3-1).

A comparison of the measured outflows with the estimates from these watershed analyses is vulnerable to uncertainty from several sources. However, the data shown in the figure below suggest that the SCS Curve Number Method with a Type B soil may be suitable for long-term depiction of the watershed of the Cascade Lakes. The data plotted in this figure include the median, mean, and geometric mean of all direct and indirect (by regression) estimates of the outflow rates from the two lakes during the monitoring period. In addition, the figure displays the annual average outflow rates for the two lakes as estimated from routing direct precipitation and watershed runoff to the outlet of the two lakes where the latter estimates were obtained from the preceding analyses. As these data indicate, the SCS Curve Number Method with a Type B soil appeared to be of the correct order of magnitude for both systems. In contrast, the Rational Method appeared to severely underestimate the outflow from both lakes for all assumed soil types. It may be the case, however, that much of the precipitation that falls on the landscape is routed to the lakes and their respective outflows by way of groundwater. In that event, the estimates by the Rational Method may not be as seemingly far from the mark as suggested by the data presented here. Future analyses will be necessary to refine these conclusions.

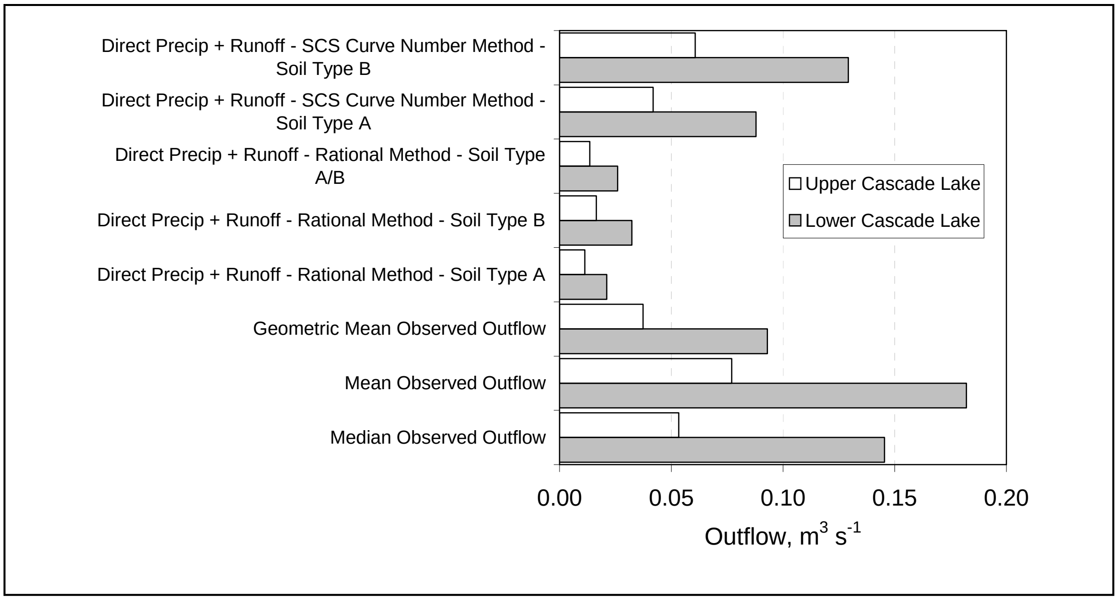


Fig. 6-3-1. Comparison of measured outflows to estimates based on runoff estimates.

Section 6-4. **MASS BALANCE MODELING**

SYSTEM DESCRIPTION AND PRELIMINARY SEGMENTATION

A simple depiction of the components of a surface water system and the mass balance approach to modeling that system is given in Fig. 6-4-1. In some respects, the situation in the Cascade Lakes is even simpler than the one depicted here because the primary contaminant of concern, chloride, is essentially unreactive. This means that the sources and sinks terms in the mass balance equation may be eliminated or greatly simplified from the approach required for more conventional contaminants, such as nutrient, hazardous substances, and toxic chemicals, all of which are reactive and can move in complex ways between air, water, and sediment.

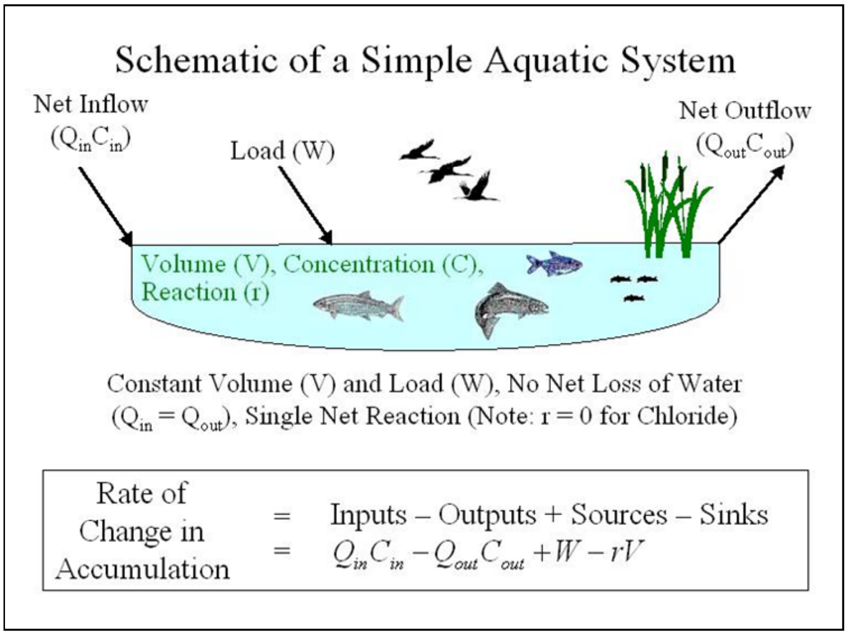


Fig.6-4-1. Schematic of the components of a simple mass balance model for chloride transport through surface waters.

The general approach to the developing a mass balance model for the Cascade Lakes system, however, will require some modification from the "one-box" approach shown above. This is true because data reported in other sections of this report clearly show the multidimensional character of the lakes, particularly with respect to vertical stratification patterns during the summer months. Because of the occurrence of this stratification, and the resulting temporal isolation of the lake water into two hydraulically distinct masses, it is possible for dissolved and particulate substances to achieve different concentrations in the two masses, and, indeed, this does occur and even for the "non-reactive" material, chloride, as is shown elsewhere in this report. Accordingly, a more detailed – but not complete – schematic of the two-lake system to illustrate the structure of the mass balance model is shown in Fig. 6-4-2. Detail at the level that accurately reflects the transport and fate processes in each lake will be provided in the next section.

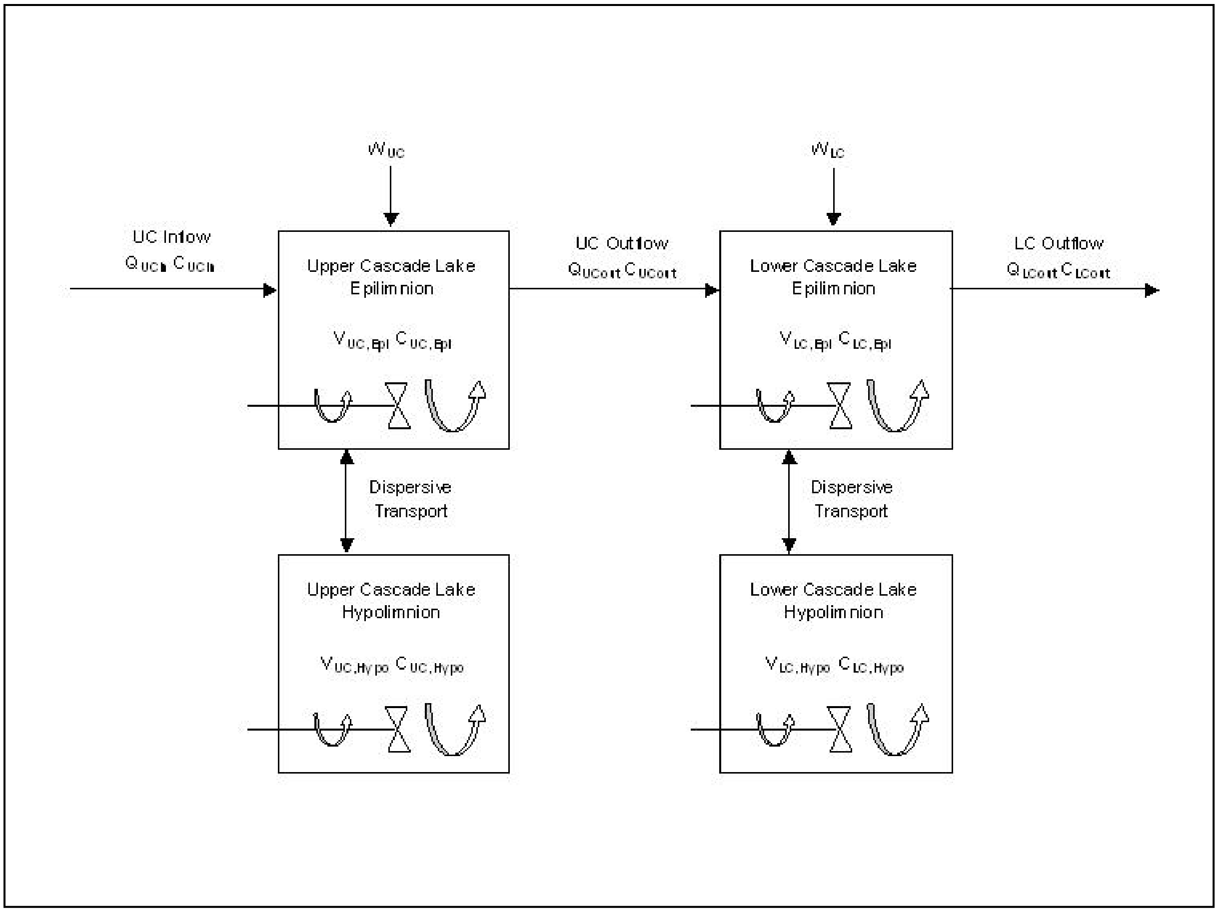


Fig.6-4-2. Mass balance model of chloride through the Cascade Lakes system.

MODEL DEVELOPMENT

Conceptualization

The model was created by considering the information known about the Cascade Lakes, as described in Chapter 5 and in the foregoing group of observations in this chapter. All temperature data, Cl^- concentrations, and other data used in calibration and as the basis for applying the Cl model to management scenarios also are presented in Chapter 5.

As depicted in the previous schematic diagram, UCL and LCL create a system of lakes in series because the outflow stream from UCL is the inflow stream into LCL. Based on this conceptual simplification as a first approximation, the model was devised to calculate simultaneously the Cl^- concentrations in the epilimnion and hypolimnion of UCL, the former of which was considered to possess a value equal the outflow concentration because of the completely-mixed segment assumption. The outflow for UCL, therefore, could be taken to equal the inflow to LCL and the solution of the model could be advanced for the Cl^- concentrations in LCL.

Although the total volume of the two lakes was considered to be a constant for modeling purposes, the relative volume of the epilimnion and hypolimnion was allowed to fluctuate in both lakes by seasonal movement of the thermocline. This was done by knowledge of the thermal structure of the lakes from temperature data collected over the twenty-one month study. A monthly resolution of thermocline position was estimated from these data, which allowed monthly estimates of the epilimnion and hypolimnion volumes of the lakes.

The primary source of Cl^- to UCL and LCL was assumed to be the application of road salt to the neighboring SR-73. Geologic sources were considered as a possibility, but the metamorphic nature of the bedrock in this area indicated that to be an unlikely source of Cl^-. Specifically, the geology of this region is composed principally of metanorthosite and charnockite (Isachsen and Fisher 1970). Any dissolution of the bedrock by water and its resulting transport into the lake system will have minimal Cl^- concentrations associated with it.

As shown in the Cl^- concentration profiles for UCL and LCL (see Chapter 5), the concentration of Cl^- increases during the spring and summer months. During the period of study, however, road salt was not applied to the highway during the spring and summer months, which indicated that the transport of road-applied Cl^- must have experienced a retardation by or retention in the soil or a groundwater system before entering the lake. The model assembled for the Cascade Lakes includes this aspect of the two lakes' behavior by utilizing a "reservoir" which permits a post-application and pre-loading accumulation of Cl^- that reflects the retarded loading to the lake system. This "reservoir" provides a mechanism to allow the model to reflect the time lag in between the road salt application to the highway increased Cl^- concentrations in the two lakes.

Revised Depiction of the Lakes and Reservoir Systems

The mass balance that was devised for UCL and LCL is displayed in Fig. 6-4-3 and Fig. 6-4-4, and several differences from the previous diagram are clear. The two diagrams, however, show similar representations of structure; however, the inflow stream for the UCL system reflects the loading from the UCL system rather than the upstream watershed.

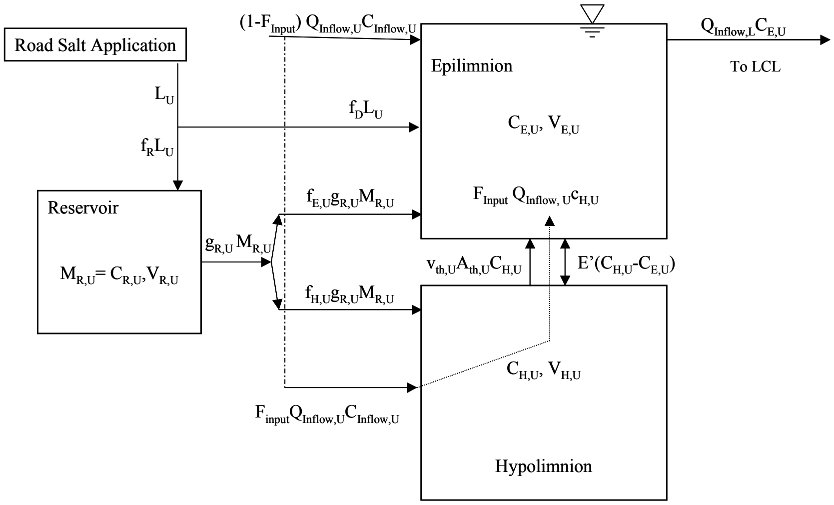


Fig. 6-4-3. Schematic diagram of Cl^- movement through the Upper Cascade Lake system for the model. Refer to variable definitions in Table 6-4-1.

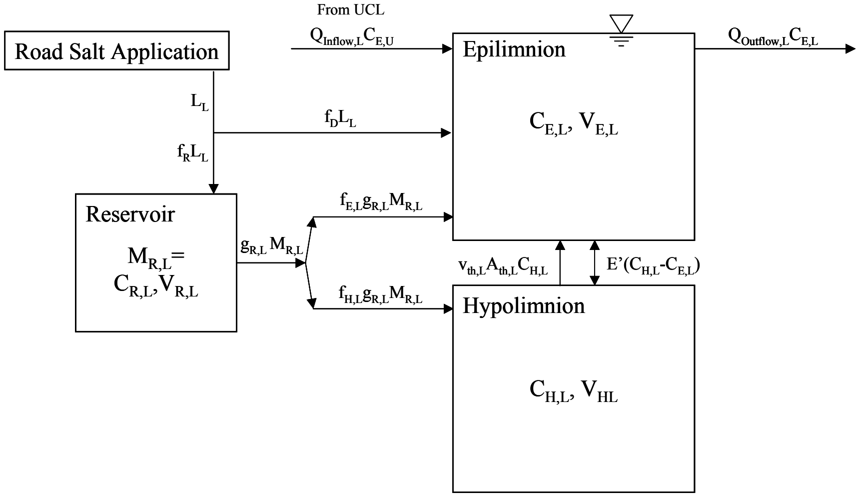


Figure 6-4-4. Schematic diagram of Cl⁻ movement through the Lower Cascade Lake system for the model. Refer to variable definitions in Table 6-4-1.

.

Table 6-4-1. Definition of terms used in flow chart with corresponding units, uses, and values.

Symbol	***Units***	***Definition***	***Values***	***Use***
L	kg/d	Cl^- load applied to SR-73	Fig 2-7	Known
f_R, f_D	0 - 1	Fraction of load entering reservoir (R) or directly into lake (L).	0-1	Calibration
M_R	kg	Mass of Cl^- in reservoir.	User Defined	Calibration
C_R	kg/m^3	Concentration of Cl^- in reservoir.	User Defined	Calibration
V_R	m^3	Volume of reservoir.	Constant	Known
g_R	0 - 1	Fraction of Cl^- leaving the reservoir.	Fig. 2-10	Calibration
f_E, f_H	0 - 1	Fraction of reservoir loading entering the epilimnion (E) and hypolimnion (H).	Fig. 2-11	Calibration
F_{Input}	0 - 1	Fraction of inlet stream load entering the hypolimnion (UCL ONLY).	0-1	Calibration
Q_{Inflow}	m^3/d	Flow rate of inlet stream to each respective lake	Fig. 2-5	Known
C_{Inflow}	kg/m^3	Cl^- concentration in inlet stream.	Vary	Known
C_E, C_H	kg/m^3	Cl^- concentration in the epilimnion (E) and hypolimnion (H) of each lake.	Vary	Model Output
V_E, V_H	m^3	Volume of the epilimnion (E) and hypolimnion (H) of each lake.	Vary	Model Output
E’	m^3/d	Bulk diffusion coefficient.	Fig. 2-9	Calibration
v_{th}	m/d	Velocity of thermocline movement through the water column.	Vary	Known
A_{th}	m^2	Surface area of the thermocline.	Vary	Known
$Q_{Outflow}$	m^3/d	Flow rate of outlet stream leaving each respective lake.	Vary	Known

In order to model the movement of Cl^- throughout UCL and LCL, it was necessary to have knowledge about mass fluxes (water and Cl) through the inlet and outlet streams, possible groundwater movement, and the application (loading) rates of anti-icing agents. It was known from rockslides in the forests within the watershed of the Cascade Lakes, and especially the region immediately surrounding and adjacent to the lakes, that geologic fracturing on a large scale had occurred. This mechanical weathering of the bedrock cliffs in this area created a large talus slope from smaller fragments of rock that remained after the fall from cliff faces. Many of these rock fragments can be seen at the shores of UCL and LCL thus indicating that the northern and southern shores of each lake may have underlying talus in part of the lake basin.

A lack of groundwater data clearly provided a source of uncertainty within the modeling analysis, yet it was a possible to include some of the uncertainty around groundwater loadings through the uncertainties associated with inputs from the hypothesized reservoir. Illustrated in Fig. 6-4-5 is a depiction of the possible movement of Cl^- through the talus slope into the water column via groundwater as the latter slowly makes its way into the lake system. This illustrates the rationale for adding the reservoir to the model framework. The soil captures the solid anti-icing agents that were plowed to the edge of the highway, in snow banks, and as the melting season eliminated the snow banks and the precipitation turned into rain, the solid anti-icing agents would dissolve and then flow through the soil with subsurface water towards the water table. In this topographic setting, the water table is represented by the surface of each lake. Depending on the time of year, the input of Cl from the reservoir is likely to vary significantly and cannot easily be measured directly. A possible approach to estimation of the concentration carried via the reservoir outflow would be to assume that the concentration resembles that of the UCL inflow stream, located along SR-73.

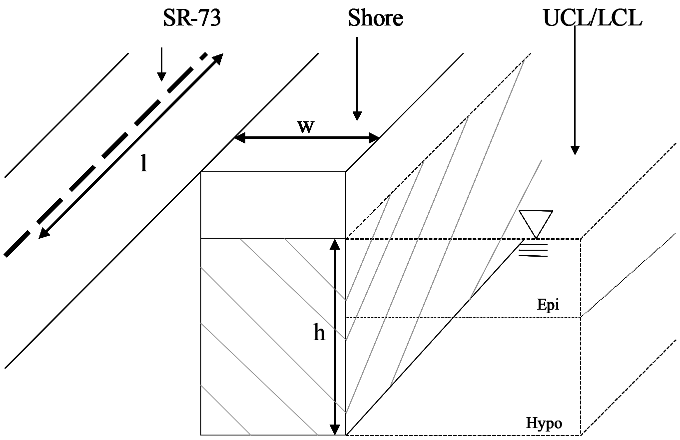


Fig. 6-4-5. Visual depiction of the method used to determine the volume parameters for the reservoir along Upper Cascade Lake and Lower Cascade Lake (not to scale). Shown is a cross-sectional view of SR-73, the shore alongside the highway, and the lake. The dashed lines represent the lakes.

The initial mass (M_R) of Cl^- within the reservoir system (i.e., $M_{R,U}$ and $M_{R,L}$ UCL and LCL, respectively) was estimated by multiplying the volume of each reservoir computed according to Equation 6 by the initial concentration. The initial concentration was estimated as the annual average value of the UCL inflow stream, as noted previously, in order to avoid being entirely arbitrary. Fig. 6-4-5 illustrates the hypothesized reservoir system for retarding the inflow to the lakes of Cl from the application of de-icing chemicals. The reservoir mass of Cl was constrained to be non-negative.

$$V_R = lwh(Porosity) \tag{Eq. 6}$$

Where:

l = Length of highway in watershed (800 m for UCL, 1,200 m for LCL)
w = Width, distance from highway to lake (5 m)
h = Height, maximum depth of lake (21 m for UCL, 14 m for LCL)
Porosity = 0.3 due to the sandy gravel soil of the shoulder (Fetter 2001).

Model Equations and Solution Approach

Beginning with the simple expression of mass conservation given previously in the schematic of the simple aquatic system, terms were added to the mass balance equations to expand the representation of transport and fate processes to include all those appropriate to knowledge of the system. These included time-dependent routing of UCL inflow to epilimnion and hypolimnion, time-dependent routing of road salt Cl to the reservoir and to the epilimnion, first-order release of Cl from reservoir with time-dependent routing to epilimnion and hypolimnion, time-dependent volume of epilimnion and hypolimnion, time-dependent advective and dispersive transport between epilimnion and hypolimnion. Separate mass balances were maintained for water volume and Cl mass.

The mass balance equations were solved for concentration and volume using the simple Euler integration, which employs a first-order finite difference approximation to the differential mass balance equations to advance the solution through each time step from user-specified initial conditions (Chapra 1997). A simple representation of this solution scheme is given by:

$$A_{t+1} = A_t + \frac{dA}{dt}\Delta t$$

where: A_{t+1} = estimated new value (concentration or volume),
A_t = current value (concentration or volume),
dA/dt = time rate of change in value (concentration or volume), and
Δt = time step.

By linking the balance calculations for volumes and concentrations, the model provided a time-variable, multi-segment, numerical simulation as the result. The platform for calculations was Microsoft Excel 2000 using a one day time-step.

The goal for the volume model was to simulate the movement of the thermocline in order to separate the epilimnetic from the hypolimnetic volumes and, in turn, concentrations. Using an assumed fixed total volume, the volume of the lake as a function of depth known from the bathymetry work, and measured thermocline migration characteristics, the actual epilimnetic and hypolimnetic volumes were computed as a function of time. For modeling purposes, the volume of the epilimnia and hypolimnia were constrained to maximum and minumum observed volumes to facilitate computational aspects of the modeling exercise. Two depths, 15 m in UCL and 11 m in LCL, were chosen as the maximum depths that the thermocline would approach.

The equations for the volume model, shown below, were based on the observed movement of the thermocline in each lake between July 2003 through March 2005. The following equations were used to describe volume for UCL's epilimnion (Eq. 7) and hypolimnion (Eq. 8) and for LCL's epilimnion (Eq. 9) and hypolimnion (Eq. 10). The terms in these equations are identified in Table 6-4-1.

$$\frac{dV_{E,U}}{dt} = (1 - F_{Input})Q_{Inflow,U} + v_{th,U}A_{th,U} - Q_{Outflow,U} + F_{Input}Q_{Inflow,U} \qquad \text{(Eq. 7)}$$

$$\frac{dV_{H,U}}{dt} = -v_{th,U}A_{th,U} + F_{Input}Q_{Inflow,U} - F_{Input}Q_{Inflow,U} \qquad \text{(Eq. 8)}$$

$$\frac{dV_{E,L}}{dt} = Q_{Inflow,L} + v_{th,L}A_{th,L} - Q_{Outflow,L} \qquad \text{(Eq. 9)}$$

$$\frac{dV_{H,L}}{dt} = -v_{th,L}A_{th,L} \qquad \text{(Eq. 10)}$$

The following equations represent the Cl mass balance model. Each layer of each lake has its own corresponding model equation. Equations were used to describe the model for UCL's epilimnion (Eq. 11) and hypolimnion (Eq. 12) and for LCL's epilimnion (Eq. 13) and hypolimnion (Eq. 14) (Chapra 1997). Note that the terms are identified in Table 6-4-1.

$$\frac{dc_{E,U}}{dt} = \frac{(1-F_{input,U})Q_{Inflow,U}c_{Inflow,U} + F_{Input,U}Q_{Inflow,U}c_{H,U} + f_{D,U}L_U + g_{R,U}M_{R,U}f_{E,U} + E'(c_{H,U} - c_{E,U}) - Q_{Outflow,U}c_{Outflow,U}}{V_{E,U}}$$

(Eq. 11)

$$\frac{dc_{H,U}}{dt} = \frac{F_{Input,U}Q_{Inflow,U}c_{Inflow,U} + g_{R,U}M_{R,U}f_{H,U} + E'(c_{E,U} - c_{H,U}) - F_{Input,U}Q_{Inflow,U}c_{H,U}}{V_{H,U}} \qquad \text{(Eq. 12)}$$

$$\frac{dc_{E,L}}{dt} = \frac{Q_{Inflow,L}c_{Inflow,L} + f_{D,L}L_L + g_{R,L}M_{R,L}f_{E,L} + E'(c_{H,L} - c_{E,L}) - Q_{Outflow,L}c_{Outflow,L}}{V_{E,L}} \qquad \text{(Eq. 13)}$$

$$\frac{dc_{H,L}}{dt} = \frac{g_{R,L}M_{R,L}f_{H,L} + E'(c_{E,L} - c_{H,L})}{V_{H,L}} \qquad \text{(Eq. 14)}$$

MODEL CALIBRATION TO MEASURED DATA

Streamflow Rates

Previously, it was discussed that measurable flow rates were determined to characterize the magnitude and variability of system flows, particularly the outflows of UCL and LCL. As shown previously, too, very high flow rates were observed in the system November 2003 when compared to November 2004. Also, flows from January 2004 through final sampling in early summer 2005 displayed flows that were much less than those observed during the late fall of 2003.

To establish a continuous flow record for simulation purposes and to avoid biasing the record with anomalous results, it was decided to use the flows observed from April 2004 through March 2005 as the full year of flow data for the model runs. The resulting hydrograph, shown with two years of flow history, is shown in Fig. 6-4-6, together with the actual hydrographic data that formed the basis for this decision. It is clear that with the exception of the unusual flow spike in fall 2003, the remaining hydrograph is well represented by the selected sequence of flow data.

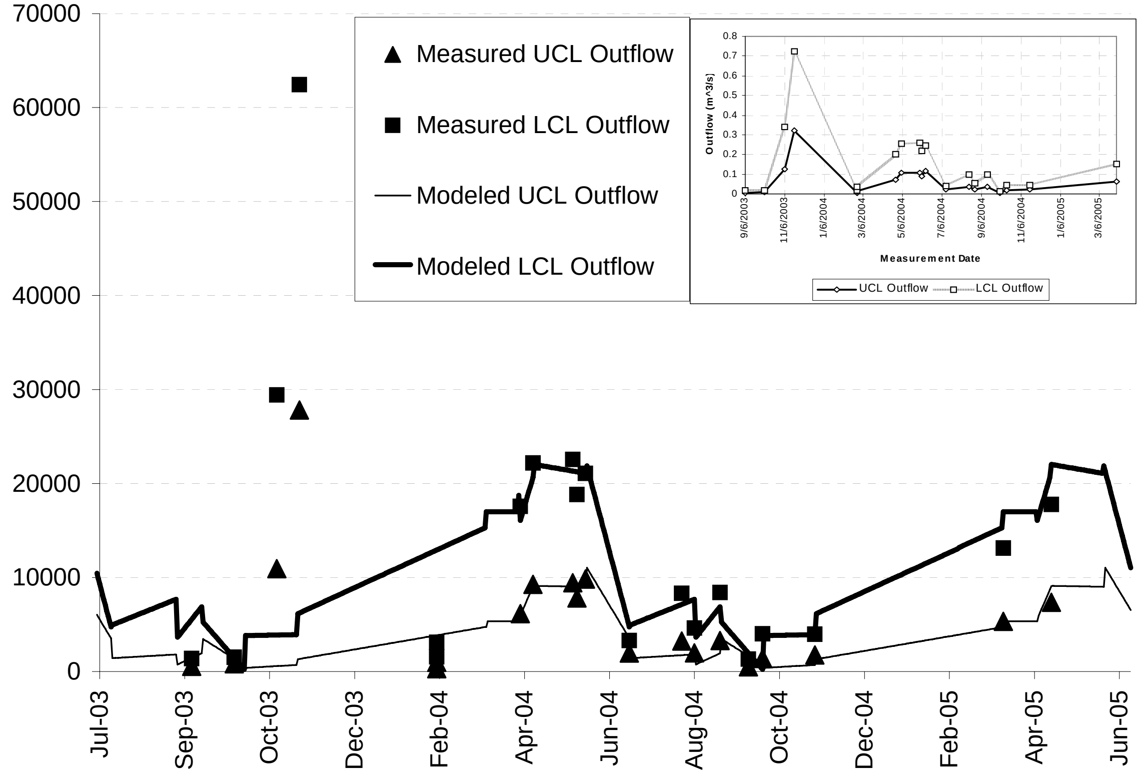


Fig. 6-4-6*. Discharge measurements for Upper Cascade Lake's outflow and Lower Cascade Lake's outflow and used for model simulations.*

As noted previously and illustrated in the figure above, the outflow from UCL (inflow to LCL) was approximately half that of the LCL outflow and the two correlated very well. This correlation allowed the estimation of the LCL outflow during times when the outflow could not be directly measured. However, the large difference between the magnitudes of the inflow and outflow of LCL (ca. 2.3-fold increase) indicated that unmonitored inflows of groundwater or surface water or both were influencing the hydraulics of the system.

The flow rates for all of the streams in each lake system seemed to be highly influenced by meteorological factors, based on field observation. Flow measurements were made from July 2003 through April 2005, which were used to find the outflow values used in the model simulation. Since the groundwater flow and surface runoff flows were unavailable, the difference between the inflow and outflow discharge would be represented as unknowns; the outflow for each system was the value used as the flow rate for this model. This created a difference in the flow rate for the UCL outflow/LCL inflow and the difference was assumed to be the difference between groundwater flow and surface runoff for the lake.

Loadings (De-Icing Chemical Application Rate)

Bonfey (2005) provided a 21 year record of de-icing chemical usage data (1984-1985 to 2004-2005) for the reach of SR-73 that extends from Keene, NY to Van Hoevenburg Park, a distance of 13.2 mi (21.2 km). The reach of SR-73 adjacent to UCL is approximately 1.45 km or 6.83 % of the total reach length, while the roadway segment adjacent to LCL is approximately 1.95 km or 9.18 % of the total reach of interest. Average total annual applied quantities of the de-icing chemicals, sodium chloride (4,905,440 lb NaCl/yr) and calcium chloride (1,190 lb $CaCl_2$/yr), were estimated from these data. The average chemical totals subsequently were converted to Cl mass loads by applying the appropriate stoichiometric coefficients, the component contributions (i.e., Cl from NaCl and $CaCl_2$) combined, and then multiplied by the corresponding exposure distances. These computations yielded average annual Cl loading estimates to each of UCL and LCL from chemicals applied to the reach of highway adjacent to each lake (Table 6-4-2).

Table 6-4-2. Estimated Average Loads of Chloride (Cl) to Cascade Lakes from Application of De-Icing Chemicals to Route 73 During the Period, 2000-2005.

Cascade Lake of Interest	Proportion of Reach Adjacent to Lake	Total Annual Load of Cl from NaCl to Lake, kg/yr	Total Annual Load of Cl from $CaCl_2$ to Lake, kg/d	Average Daily Load of Cl to Lake During Application Period, kg/d
Upper	6.83% (1.45 km or 0.901 mi)	92,132.16	17.29	504.57
Lower	9.18% (1.95 km or 1.212 mi)	123,901.87	23.26	678.56

It was assumed that the application of de-icing chemicals occurred only during periods of potential snow and ice accumulation on the roadway; this period was assumed to be one-half year or 182.6 days. Because available data did not provide time resolution of the applications, the Cl loading was assumed to occur at a constant daily rate over the application period.

The Cl amounts shown in the right-hand column of Table 6-4-2 will be referred to as the base-case loading for the model.

UCL has two inlet streams; one of which travels alongside the highway before flowing into the lake; the other flows off of Cascade Mountain into UCL. The inlet of greatest concern for UCL is the one located alongside SR-73. This inflow stream was found occasionally to have a very high concentration of Cl. Seasonal sampling of this stream indicated that the stream had a range of Cl^- concentrations between 6 mg/L and 200 mg/L depending on the season (Fig. 6-4-7). It is likely that occasionally these concentrations have a significant impact on the concentration of Cl in UCL as a whole.

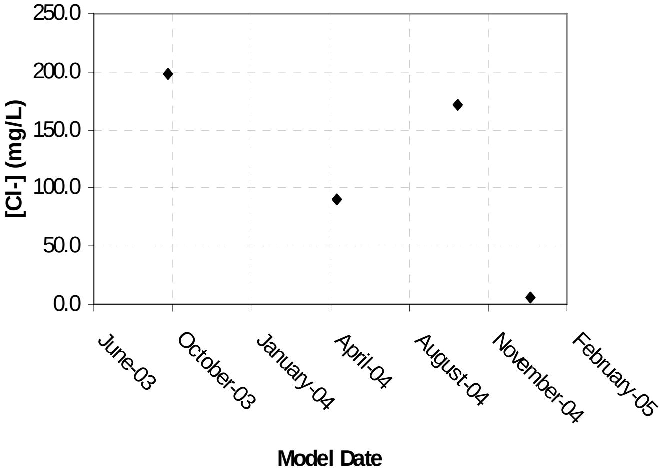


Fig. 6-4-7. The concentrations of the Upper Cascade Lake highway stream samples taken during different times of the year. A lower concentration was observed during the winter months, but higher concentrations were seen in the spring, summer, and fall seasons

As seen in the data presented in Fig. 6-4-7, the inflow stream experienced a low Cl^- concentration during the winter months compared to the summer and fall months when the Cl^- concentrations were higher. This suggests that the Cl^- loading primarily took place *after* the winter applications of anti-icing agents to the highway. Owing to the observed Cl concentration variability evident in the inflowing stream, the concentration for the model was estimated by the calibration procedure but constrained to values within the observed range of concentrations. The resulting set of concentrations as used in the model for UCL are given in the table below. This was compensated for in the model by decreasing the Cl^- concentration entering UCL from the inflow stream (Table 6-4-3).

***Table 6-4-3**. The annual chloride concentrations entering Upper Cascade Lake from the stream neighboring State Route 73 used in the model.*

	Upper Cascade Lake Inflow Concentrations (C_inflow) (mg/L)
July	45
August	45
September	45
October	50
November	45
December	45
January	45
February	45
March	45
April	45
May	45
June	45

The data suggested that the inflow stream near SR-73 not only added Cl to the UCL epilimnion, but also the hypolimnion. The model factor, F_{Input} was implemented to partition a fraction of the UCL highway inlet load for diversion into the hypolimnion as would have been appropriate due to seasonally varying thermal or chemical affects on the density of the inflow compared to water in the lake. From the observation that this stream had a cold temperature and the trial-and-error discovery that the loading to the hypolimnion was insufficient to reproduce observed concentrations in the UCL, it was assumed that some of the cold high Cl influent stream water sank to lower depths as it entered the lake. This additional load allowed for a more accurate model calculation of the hypolimnetic concentrations in UCL compared to its measured values.

The second input stream to the UCL system was located on the side of the lake opposite the highway and has its origin on Cascade Mountain.. This stream was sampled and its Cl^- concentration was comparatively low, approximately 2 mg/L. This low concentration was not a focus of the model due to the low flow rate. Since this particular inlet was not a major transport of Cl^-, it was omitted from the UCL system as a stream load of Cl, although its effects were recognized as important to the overall system hydraulic balance.

Water flowed from UCL by way of a short connecting stream into LCL. The connector stream comprised the only surface water outflow from UCL and the main surface inflow to LCL. LCL also received flow from a second stream with its origin on Cascade Mountain. The second stream inspired the name “Cascade” lakes as it provides a beautiful cascading waterfall at certain time of the year. For modeling purposes, however, LCL was considered to have one input stream – the outflow from UCL – and one outlet stream.

Due to the differing bathymetry and morphology of each lake and differences in the characteristics of their respective inflow streams, LCL was not affected by the inflow stream concentrations as UCL was. In effect, the water entering LCL from UCL was epilimnetic water during those periods when thermal stratification existed and essentially completely mixed UCL water otherwise. Consequently, the water entering LCL from UCL was nearly equal in temperature and similar in Cl to the water in LCL. Hence, the waters entering LCL encountered virtually no resistance to mixing, which obviated the need to partition the inflow seasonally to the LCL system between epilimnion and hypolimnion.

The initial Cl^- concentrations in the subsurface reservoir were unknown and were constrained only by the need to ensure the reservoir would not become completely depleted of Cl in the first two years of simulation, in effect for the period of observation. For UCL the initial reservoir concentration was set to 500 mg/L, while the value for LCL was set to 4750 mg/L.

Bulk Diffusion Coefficient

The bulk diffusion coefficient was used as defined by Chapra (1997) via Eq. 15:

$$E' = \frac{EA_c}{\ell} \qquad \text{(Eq. 15)}$$

Where: E' = *Bulk diffusion* ($m^3 d^{-1}$)

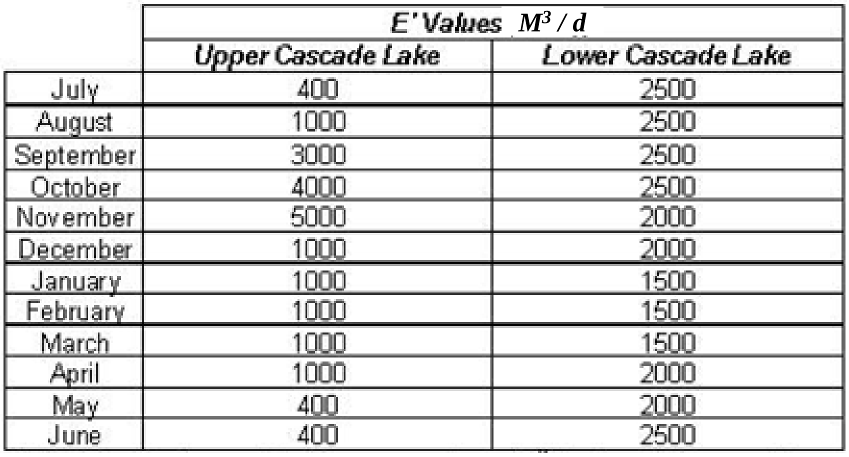
E = *Diffusion coefficient* ($m^2 d^{-1}$)

A_c = *Cross sectional area of the interface between the two layers* (m^2)

l = *Mixing length (m)*

By using Chapra’s (1997) definition of E’, the turbulent transfer of mass between the epilimnion and hypolimnion of UCL and LCL could be represented in the model. Since this value involved measurements that were not included in the study, this parameter served as an additional calibration variable for the model. Table 6-4-4 represents the values for E’ in the first cycle of calculations used in the model. These values were repeated for the entirety of the model.

Table 6-4-4. Annual dispersion values, represented by the symbol E’, used for Upper and Lower Cascade Lakes.



	E' Values M^3 / d	
	Upper Cascade Lake	**Lower Cascade Lake**
July	400	2500
August	1000	2500
September	3000	2500
October	4000	2500
November	5000	2000
December	1000	2000
January	1000	1500
February	1000	1500
March	1000	1500
April	1000	2000
May	400	2000
June	400	2500

Reservoir Loading Rates and Layer Distribution

For modeling both UCL and LCL, 50% of the applied anti-icing agents were diverted to the reservoir while the remaining 50% was allowed to enter directly the epilimnion of each lake. This fractionation was chosen based on the decreased loading from the reservoir during the winter months and the need to distribute the Cl^- load from the highway to the each lake over the course of the entire year. The result was half being diverted directly to the lakes and half sent to the reservoir for future distribution.

Control of the released of Cl from the reservoir to each lake was exerted by the reservoir distribution factor (g_R). Table 6-4-5 represents the g_R values used for both UCL and LCL in this model. The factor, g_R, was a hypotesized parameter that was estimated by by model calibration , and theredore not directly measured.

Table 6-4-5 also represents the values used for the reservoir distribution factor (g_R) for each month. These values are repeated for each year of the model.

Table 6-4-5. Monthly reservoir distribution factors for Upper and Lower Cascade Lake.

	Upper Cascade Lake (g_R,U)	**Lower Cascade Lake (g_R,L)**
July	0	0.001
August	0	0.001
September	0	0
October	0	0
November	0	0
December	0	0
January	0	0
February	0	0
March	0	0
April	0	0
May	0.01	0.001
June	0.01	0.001

The mass of Cl^- that enters each lake from the reservoir then is separated into the loading of the epilimnion and hypolimnion layers of UCL and LCL. These values of the control parameters $f_{E,U}$, $f_{H,U}$, $f_{E,L}$, and $f_{H,L}$ were determined by trial-and-error during calibration and are a fraction of the total mass entering the lakes from the reservoir. Hence, for each lake, the two fractions were constrained to sum to 1.0.

The values of these parameters for UCL and LCL are represented in Table 6-4-6 and Table 6-4-7.

Table 6-4-6. Represents the distribution of the reservoir loading to the Upper Cascade Lake epilimnion ($f_{E,U}$) and hypolimnion ($f_{H,U}$).

	Upper Cascade Lake Reservoir Load Distribution	
	f_E,U	f_H,U
July	0.4	0.6
August	0.4	0.6
September	0.4	0.6
October	0.4	0.6
November	0.4	0.6
December	0.9	0.1
January	0.9	0.1
February	0.9	0.1
March	0.9	0.1
April	0.9	0.1
May	0.4	0.6
June	0.4	0.6

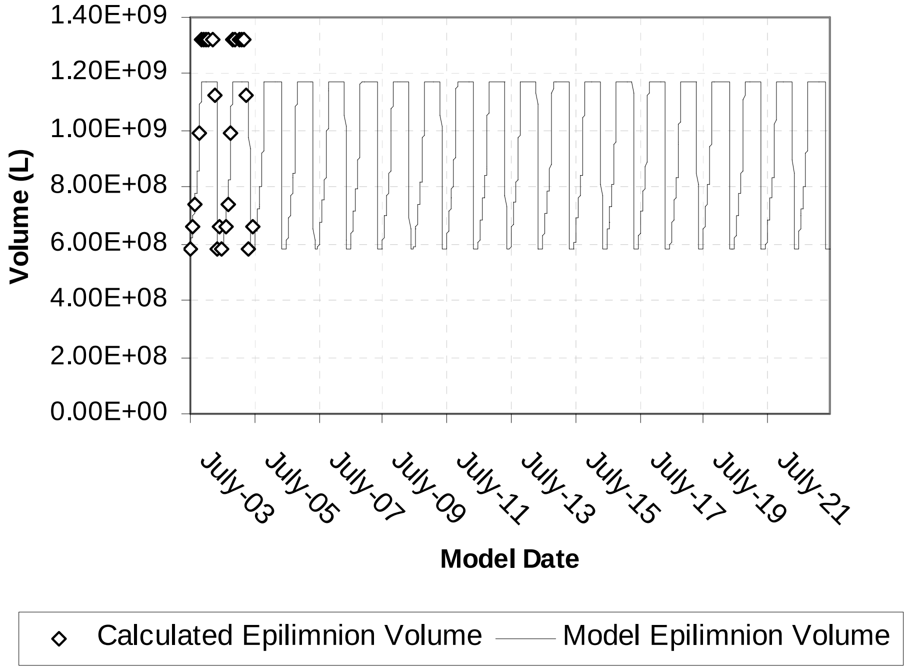
Table 6-4-7. Represents the distribution of the reservoir loading to the Lower Cascade Lake epilimnion ($f_{E,L}$) and hypolimnion ($f_{H,L}$).

	Lower Cascade Lake Reservoir Load Distribution	
	f_E,L	f_H,L
July	0.5	0.5
August	0.5	0.5
September	0.5	0.5
October	0.5	0.5
November	0.5	0.5
December	0.5	0.5
January	0.5	0.5
February	0.5	0.5
March	0.5	0.5
April	0.5	0.5
May	0.5	0.5
June	0.5	0.5

MODEL OUTPUT

The model simulated the concentrations and the volumes from July 2003 through June 2023. This allowed for a calibration period from July 2003 through March 2005 using the data accumulated by monthly water sampling of each lake. In addition to the calibration period, twenty-year forecasts of the lake system response were computed for current loading conditions, plus half and double the current loadings. In addition, system response to elimination of all Cl loading after 10 years of simulation was made assuming the current initial conditions.

The volume outputs for the UCL epilimnion and hypolimnion are represented in Fig. 6-4-8. LCL's epilimnion and hypolimnion volume calculations are represented in Fig. 6-4-9. The outputs for the Cl^- concentration model, with the application equal to the original loading, regarding UCL and LCL are shown in Figs. 6-4-10 and 6-4-11.



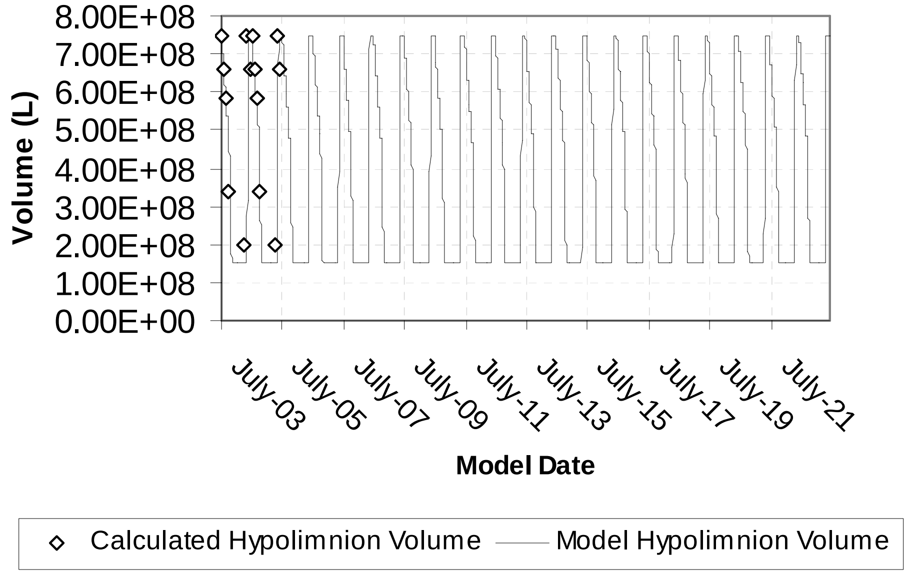
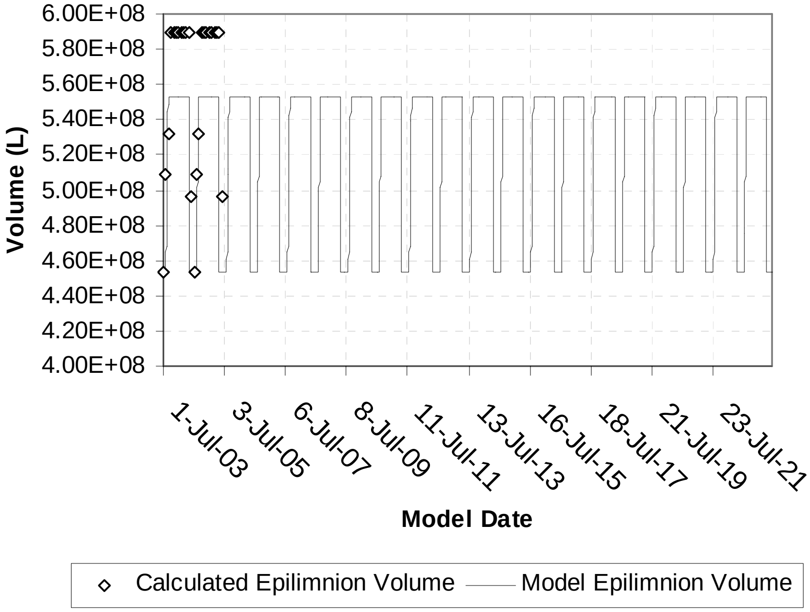


Fig. 6-4-8. Model calculated volumes for Upper Cascade Lake epilimnion (top) and hypolimnion (bottom) from July 2003 through June 2023.



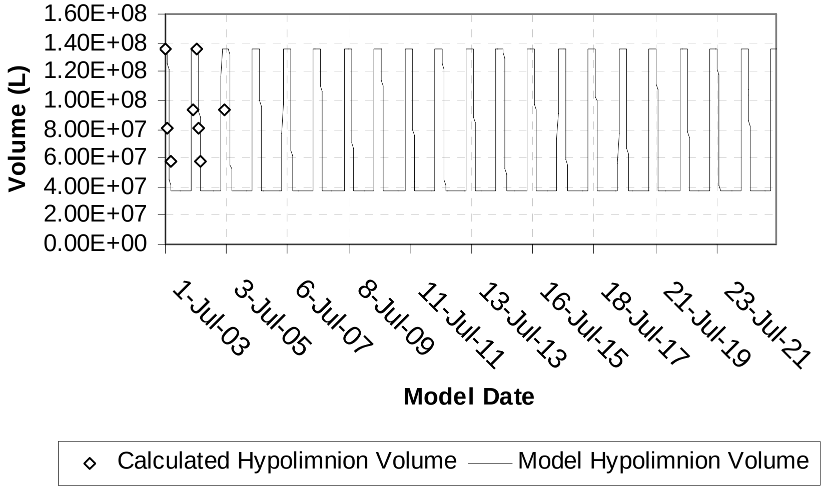
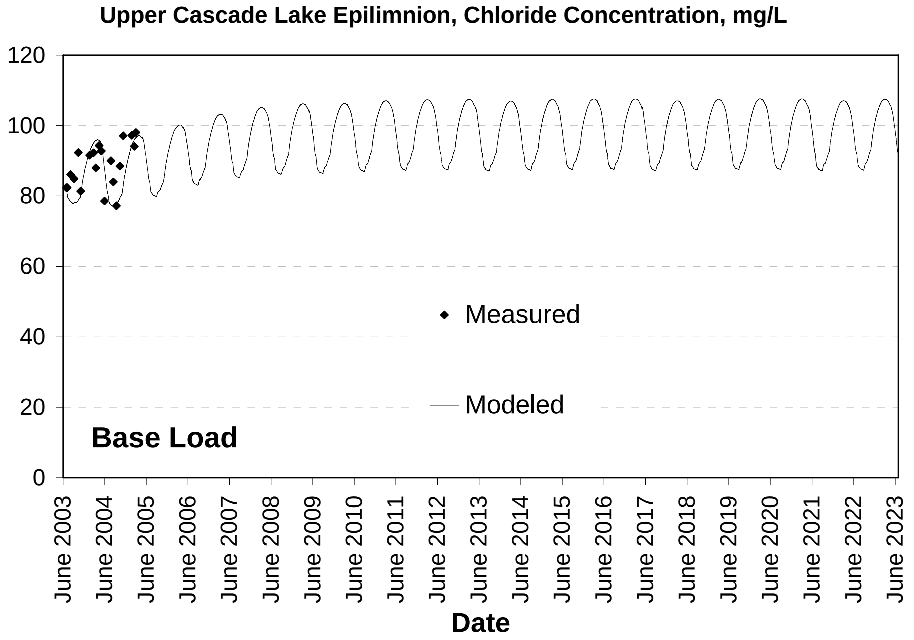


Fig. 6-4-9. Model calculated volumes for Lower Cascade Lake epilimnion (top) and hypolimnion (bottom) from July 2003 through June 2023.



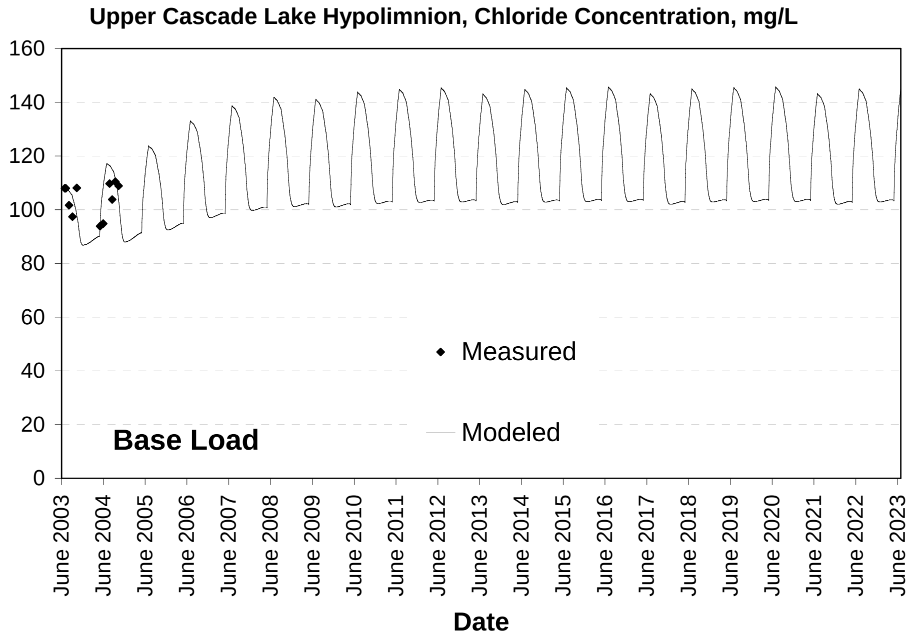
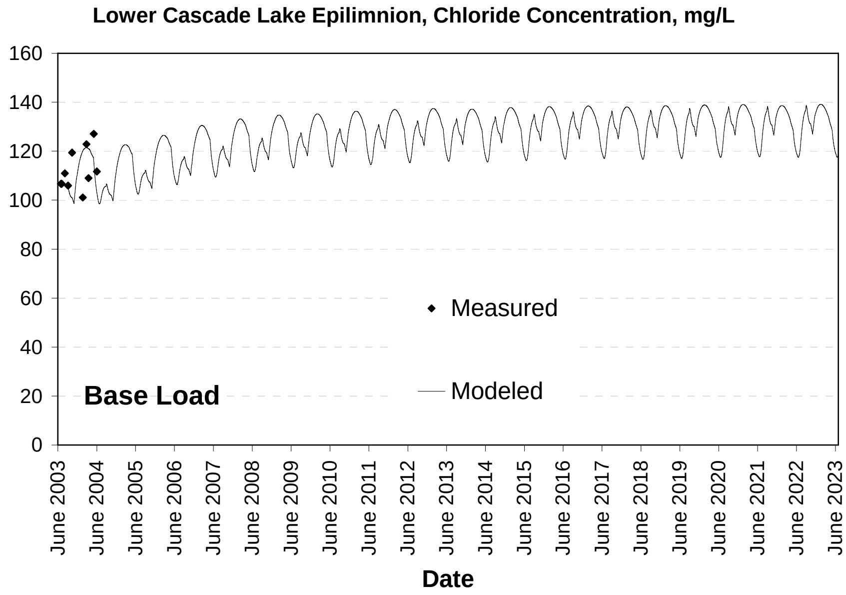


Fig. 6-4-10. Model output for the calculations of chloride concentration from July 2003 through June 2023 for Upper Cascade Lake epilimnion (top) and hypolimnion (bottom) using the base load application of anti-icing agents to State Route 73.



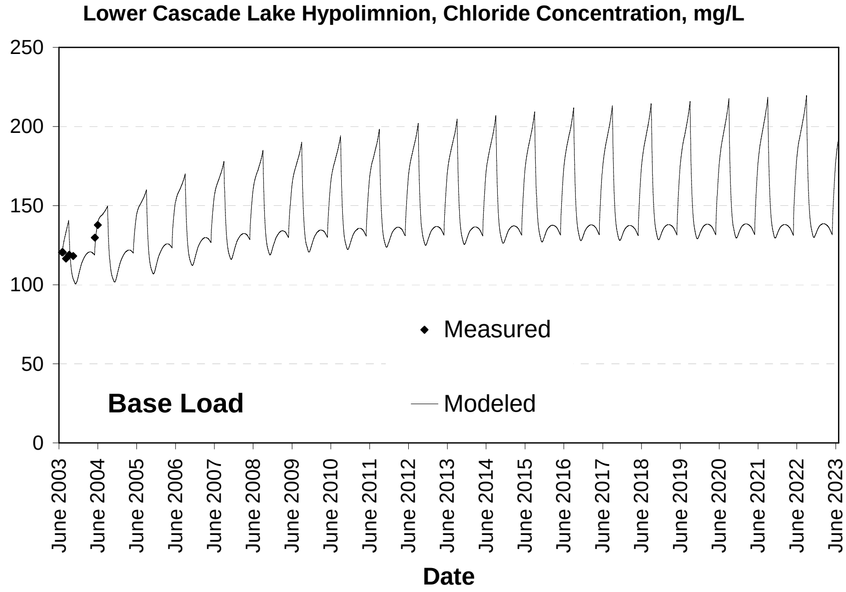
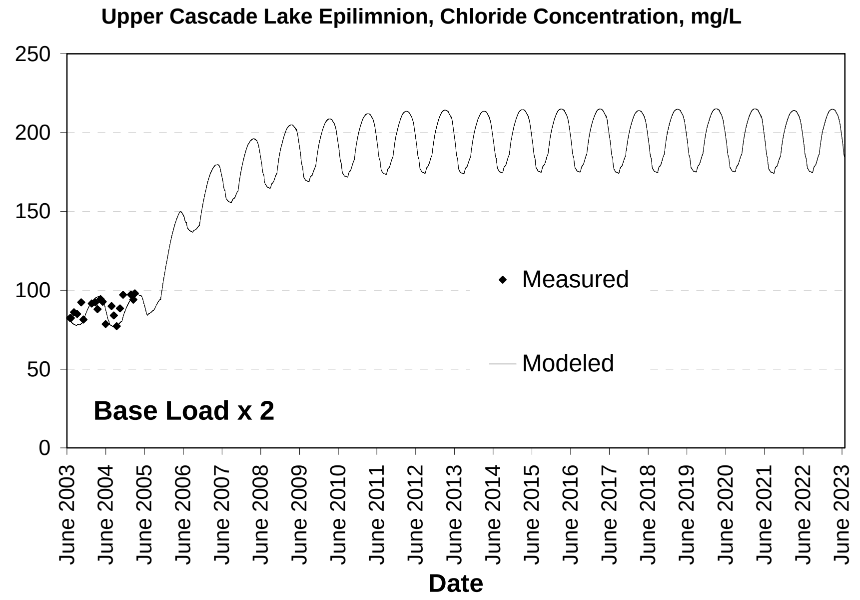


Fig. 6-4-11. Model output for the calculations of chloride concentration from July 2003 through June 2023 for Lower Cascade Lake epilimnion (top) and hypolimnion (bottom) using the base load application of anti-icing agents to State Route 73.

In order to use this model as an engineering tool, the loading was designed to have an adjustment factor. With this tool, the user can adjust the amount of anti-icing agent that is to be applied to the highway compared to the original loading by changing this loading factor. Fig. 6-4-12 and Fig. 6-4-13 represent the output of the model for the UCL with double the original application road salt and zero application, respectively. Fig. 6-4-14 and Fig. 6-4-15 represent the model output for LCL with double the original application of road salt and zero application, respectively. The output of the model if the base load is halved (50% of current loading) is shown in Fig. 6-4-16 and Fig. 6-4-17.



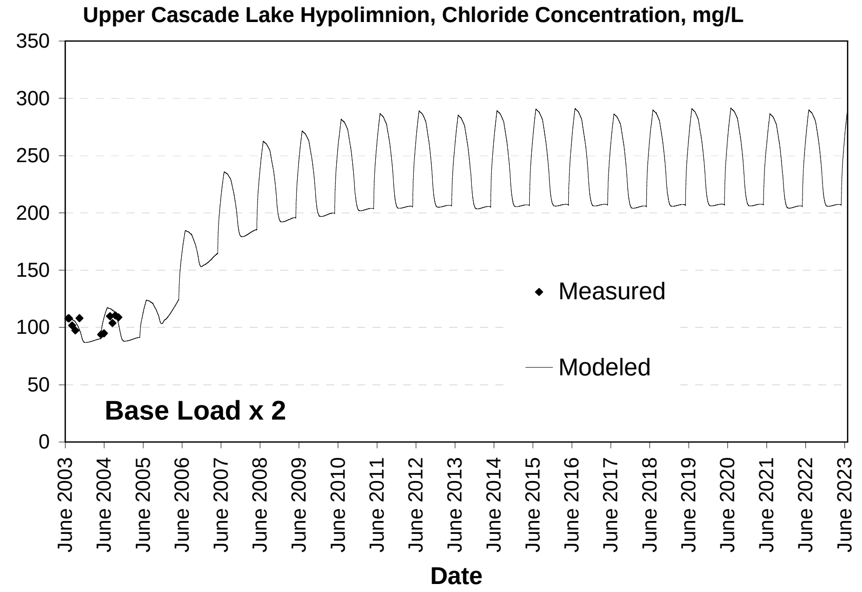
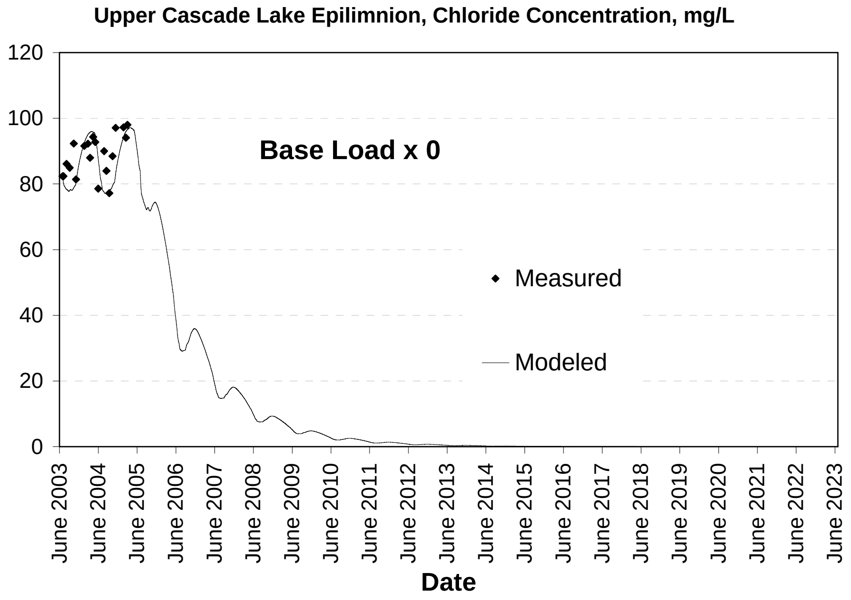


Fig. 6-4-12. Model output for Upper Cascade Lake epilimnion (upper graphic) and hypolimnion (lower graphic) chloride concentrations. The application rate of road salt to State Route 73 is double the original application beginning in July 2005 through June 2023.



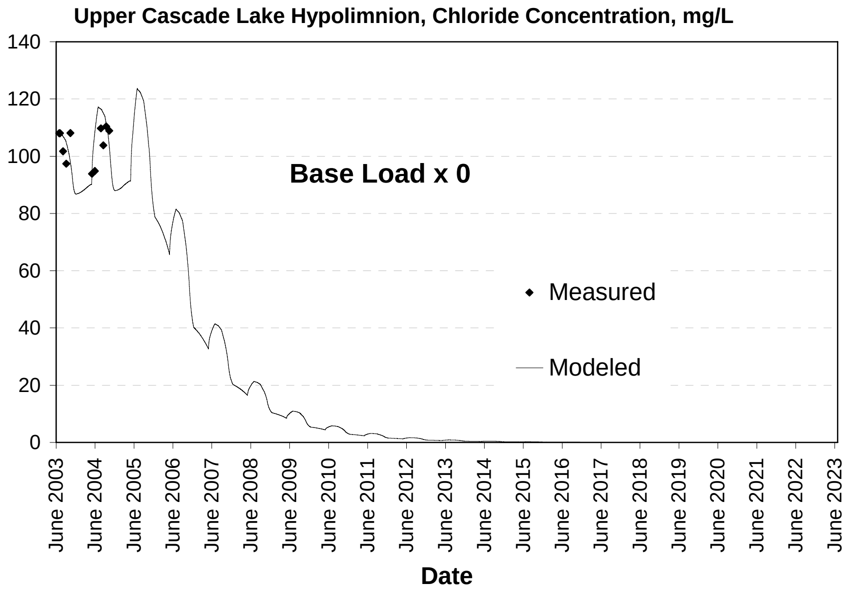
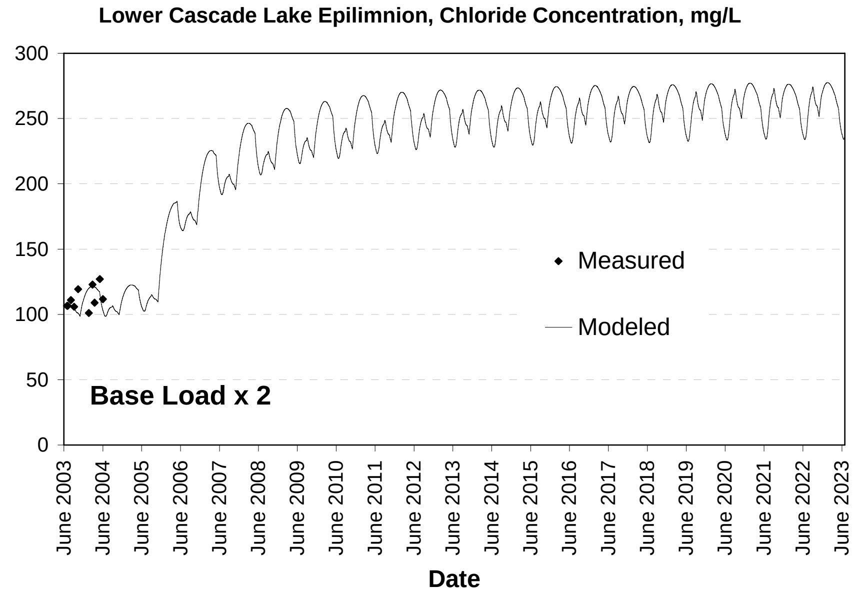


Fig. 6-4-13. The chloride concentrations in the Upper Cascade Lake epilimnion (top graphic) and hypolimnion (bottom graphic). There is no application of road salt to State Route 73 from July 2005 through June 2023.



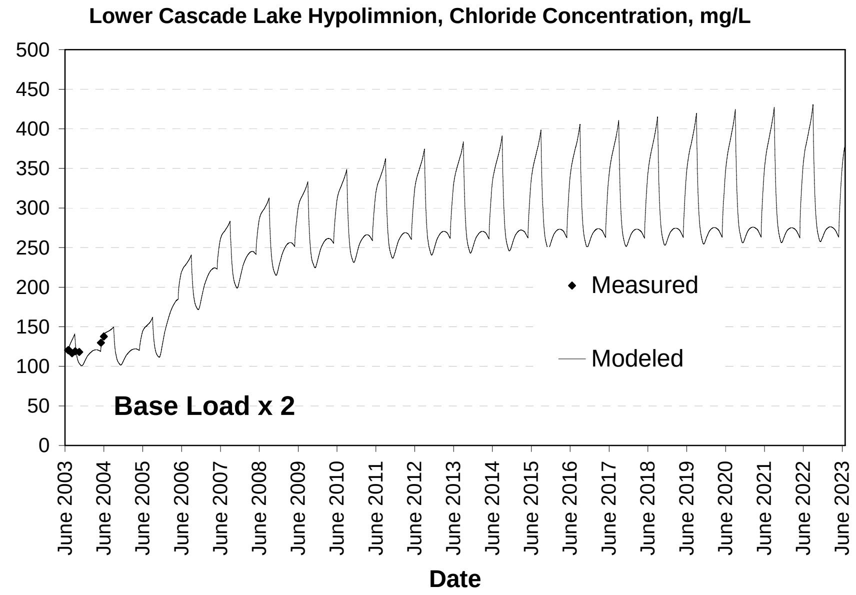
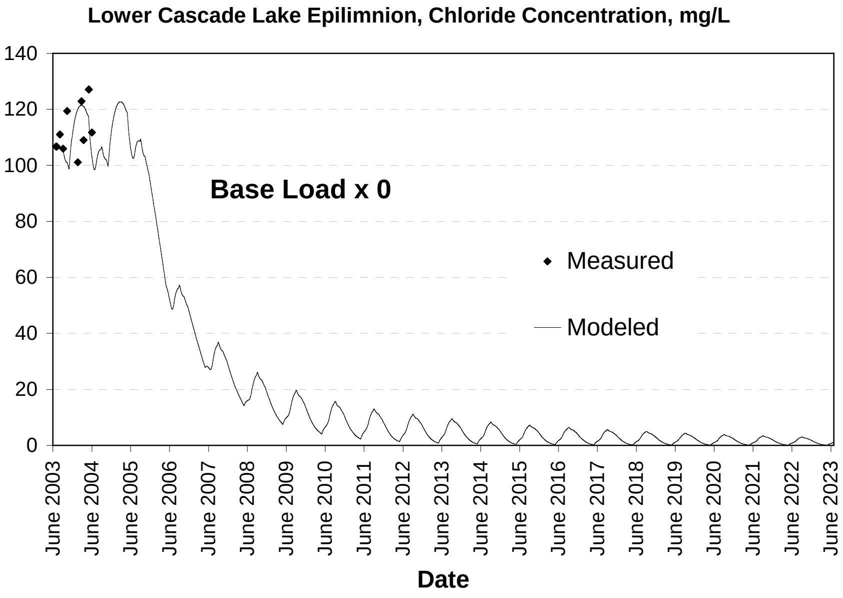


Fig. 6-4-14. Model output for Lower Cascade Lake epilimnion (upper graphic) and hypolimnion (lower graphic) chloride concentrations. The application rate of road salt to State Route 73 is double the original application beginning in July 2005 through June 2023.



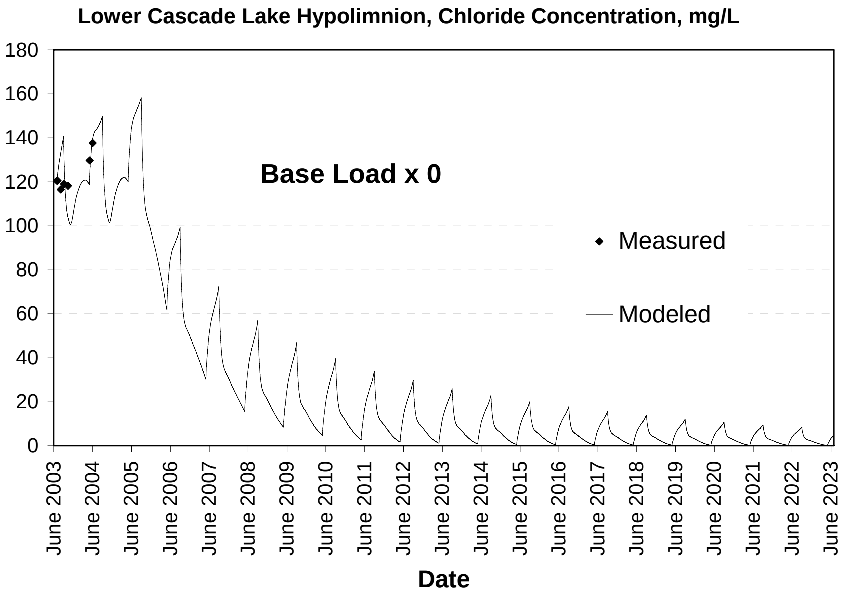
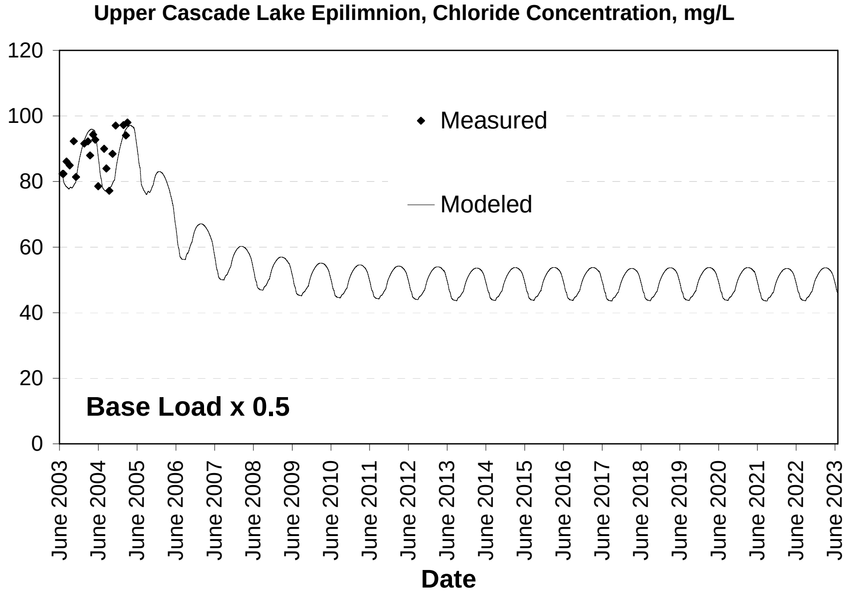


Fig. 6-4-15. The chloride concentrations in the Lower Cascade Lake epilimnion (top graphic) and hypolimnion (bottom graphic). There is no application of road salt to State Route 73 from July 2005 through June 2023.



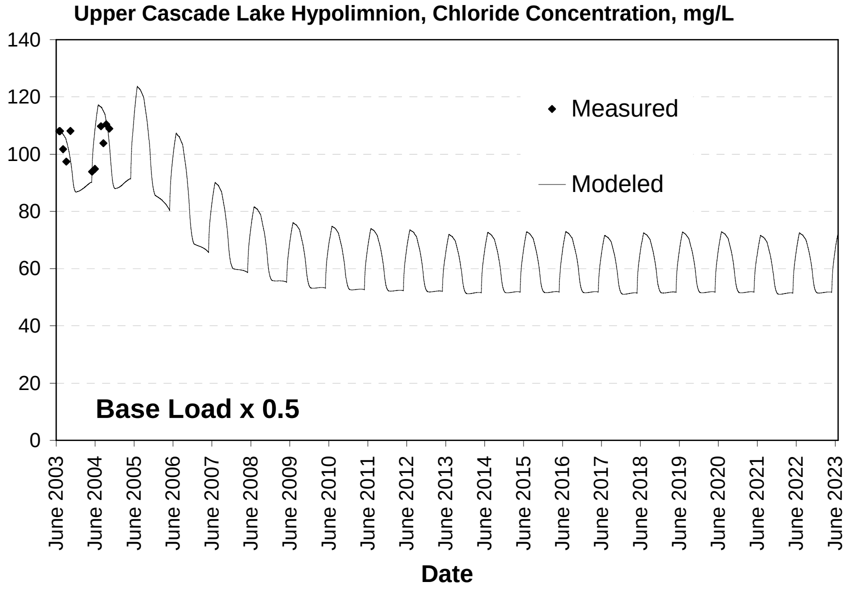
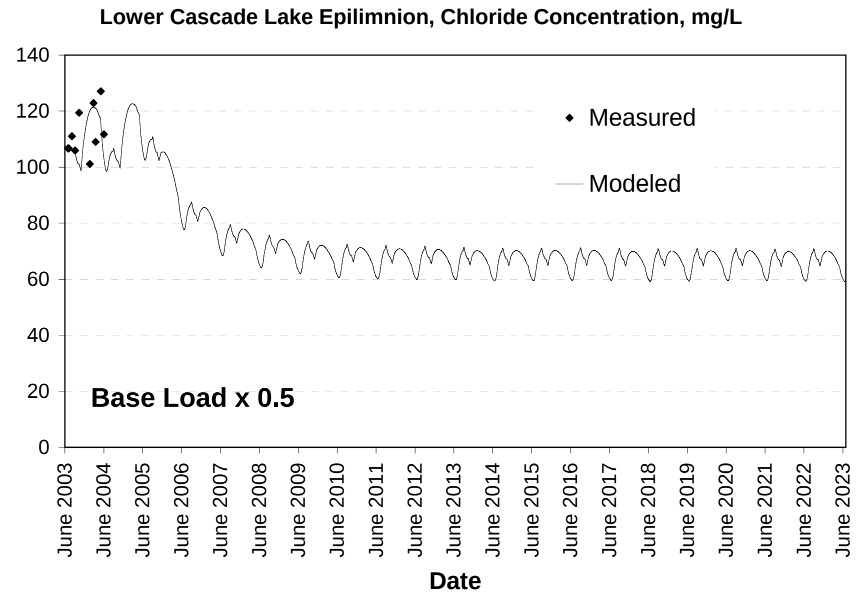


Fig. 6-4-16. The chloride concentrations in the Upper Cascade Lake epilimnion (top graphic) and hypolimnion (bottom graphic). The application rate of road salt to State Route 73 is one-half the base load application beginning in July 2005 through June 2023.



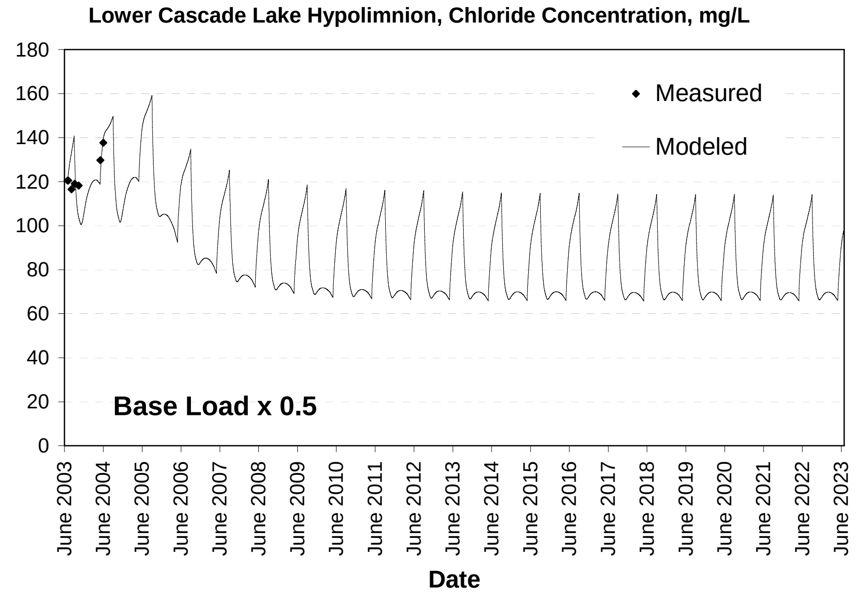


Fig. 6-4-17. The chloride concentrations in the Lower Cascade Lake epilimnion (top graphic) and hypolimnion (bottom graphic). The application rate of road salt to State Route 73 is one-half the base load application beginning in July 2005 through June 2023.

DISCUSSION

Volume Simulation

The time series of epilimnion and hypolimnion volumes for both UCL and LCL as calculated by the model are shown in Fig. 6-4-8 and Fig. 6-4-8 demonstrate capture of the seasonal changes in layer volume experienced by each lake. The volume for the epilimnion of both UCL and LCL is at the maximum value during the winter season when the thermocline has decomposed due to the cooling of each lake and the lakes, theoretically, pass from an isothermal condition to ice-covered winter stratification. The epilimnetic volume then begins to decrease during the spring and summer seasons when the hypolimnion appears with the re-establishment of the thermocline.

The volume for the hypolimnion for both UCL and LCL shows a similar seasonal trend in the model. During the winter period, the hypolimnion no longer actually exists since the lake turns over in the fall and winter. For computational purposes, however, the model retains a small hypolimnetic volume remains during the winter months in UCL and LCL. There are two reasons for this: 1) in spite of an apparent isothermal profile, followed by an inverse stratification, both of which suggest a complete mix of the water column, a persistent layer of elevated Cl was observed to remain over-winter in the deeper waters of both UCL and LCL and 2) the model required a non-zero hypolimnetic volume for mass balance computations.

Base Loading

The output of the model using the base loading of anti-icing agents indicated that UCL (Fig. 6-4-10) will experience a general increase in Cl^- concentration over the next twenty years with the current application of anti-icing agents to SR-73.

LCL (Fig. 6-4-11) also displays a general increase in concentration over the next twenty years. However, the hypolimnion of LCL display a marked increasing range in concentration with each passing year. This phenomenon is attributable to the increasing significance of the subsurface reservoir of Cl that – within the model framework – loads the system during periods when the de-icing chemicals are no in active use. Of course, the reservoir is an hypothetical construct and more research is needed to ascertain the extent to which it may actually function to affect the fate of and Cl levels in the Cascade Lakes.

Variations on Highway Application Output

When the anti-icing agent application was doubled, both UCL and LCL (Fig. 6-4-13 and Fig. 6-4-14) exhibited a more rapid rise in concentrations than observed with the original loading rate. The concentrations were expected to rise more rapidly with the increased amounts of Cl^- being introduced to the watershed. The rise in concentrations associated with a doubled application rate to SR-73 may approach and possibly exceed the national criterion of 227 mg/L Cl^- (United States Environmental Protection Agency 1988) for LCL in twenty years and but possibly as soon as five to 10 years for the LCL hypolimnion. The Cl levels in the UCL epilimnion are projected to remain below the criterion, but the hypolimnion may achieve exceedences.

If the anti-icing agent application were to be reduced by 50% or even eliminated starting in July 2005, a very fast drop in Cl^- concentration can be observed (Fig. 6-4-16 and Fig. 6-4-17). With a 50% reduction, the UCL epilimnion and hypolimnion would approach steady state concentrations that would average between 50 mg/L and 60 mg/L, LCL would show somewhat greater steady-state values with the epilimnion and hypolimnion concentrations averaging between 65 mg/L and 85 mg/L. For each lake, a rapid drop in Cl^- concentration was observed within the first two years of reducing or eliminating the loading to SR-73 (Fig. 6-4-15 and Fig. 6-4-16). For this model, this event is a "washout", dropping the Cl^- concentration in each lake to the concentration similar to that found in the inflow streams but augmented by the slow drain from the reservoir. This indicates that a concerted effort to protect the lakes such as eliminating the use of Cl^- anti-icing agents may show substantial results in a relatively short amount of time.

Data were not sufficient to support a direct calculation between the inflow stream Cl^- concentration and the amount of anti-icing agent application to SR-73. However, it was assumed that changes to the application rates in the adjacent highway reach eventually would be reflected in the UCL inflow stream concentration and, eventually, through the system. In future studies of the Cascade Lakes, further sampling and research on this stream is necessary to allow a direct calculation between the anti-icing agent application to SR-73 and the concentration in the inflow stream.

Data Comparison

Table 6-4-8 presents a simplified view of the dependence of the two lakes Cl concentrations on the conditions of Cl loading by way of de-icing chemicals. Presented in the table are the Cl concentrations predicted by the model after twenty years of simulation using the indicated rate of application of anti-icing agents. The base loading is that obtained from the 5 year average chemical usage rate noted previously for the Sr-73 reach of interest.

Table 6-4-8. Chloride concentration comparison between the current magnitude and those found twenty years in the future with variations on the application of anti-icing agents.

	Chloride Concentration, mg/L			
	Upper Cascade Lake		Lower Cascade Lake	
Current Conditions	82	108	107	121
Current Loading Rate, 20 Years	97	122	128	170
Double Loading Rate, 20 Years	196	248	251	335
Halve Loading Rate, 20 Years	50	61	64	85
Zero Loading, 20 Years	~ 0	~ 0	~ 1	~ 2.5

As shown, increases in the application of anti-icing agents will increase the Cl^- concentrations found in each lake. Similarly, decreasing the load will reduce the Cl concentration in each system. It is noteworthy, however, that after 20 years with no chemical application, the LCL continues to show the effects of past practices, primarily from slow unloading of the contents of the reservoir.

Section 6-5. **FUTURE RESEARCH NEEDS**

Increasing the accuracy of the model will require improved monitoring of stream flows and to continued monitoring of water quality in each lake. The streams should be sampled regularly and the construction of a weir would facilitate accurate flow measurement for the inflow and outflows of UCL and LCL. Monitoring the complete inflow of both lakes directly, however, will continue to be difficult. In addition, watershed studies must be conducted in order to correlate the anti-icing agent application and the reservoir system to the stream and possibly groundwater Cl concentrations. This can be accomplished by further sampling of the stream to accumulate Cl concentrations while maintaining a log of anti-icing agent application to SR-73 from NYSDOT. Comprehensive watershed analysis can be helpful, too, for improving knowledge and confidence about overland and subsurface flows through the system.

A more focused study on the groundwater and the reservoir system also would be valuable for improved specification of the hydraulic framework of the system. Sediment cores and even monitoring wells would be extremely useful to determine the actual concentrations of Cl- in the groundwater in the reservoir. Not only would the cores and wells aid in giving a true characterization of the reservoir, but would also aid in mapping the groundwater system as the water moves into each lake. This process would yield a better time estimation on how long the lakes will need to recover. The use of water tracers could be useful in determining the transport rates of the reservoir in addition to the monitoring wells and the continued lake studies to yield an accurate retention time of the groundwater system.

Section 6-6. **CONCLUSIONS**

The model results indicate that over the next twenty years the Cl^- concentrations will continue to slowly increase with a loading similar to the current load that is applied to SR-73. This increase may also increase the possible onset of meromictic conditions in both UCL and LCL.

In terms of highway management, the model was also used to predict the Cl^- concentrations when the applied anti-icing agent load to SR-73 was doubled, reduced, and eliminated (zero load). The doubled load indicated an increase in Cl^- concentrations in each lake compared to the management practices taking place presently. This increases the possibility of both UCL and LCL becoming meromictic and of aquatic organisms will become stressed. By terminating the use of road salt, the Cl^- concentration decreased rapidly and approached background for the region. This being the case, the model indicates that the Cl^- concentrations in each lake can be managed.

In addition, the hypothesized reservoir system allowed the model to mimic an important feature of the system – loading to the lakes that was time-shifted from the application, which allowed the model to mimic the increases in lake Cl^- concentration during the spring and summer months when anti-icing agents were not being applied. This also explains why the concentrations in the lakes would not respond immediately to the winter loading. Regardless of the amount of anti-icing agent being applied to the highway, there will still be a significant time-delayed source of Cl^- loading into each lake from the reservoir. This means that minimizing the application of anti-icing agents to the highway and allowing some time to pass is the only combination of remediation techniques that will begin to decrease the concentration of Cl^- in each lake.

CHAPTER 7

CONCLUSIONS & RECOMMENDATIONS

Preparing to collect water samples, Lower Cascade Lake. Photo by Michael Twiss.

Chapter 7. **CONCLUSIONS AND RECOMMENDATIONS**

Report prepared by Tom A. Langen and the Project Team

Human activities have impacted forest cover and water quality at the Cascade Lakes beginning in the 19th century, and anthropogenic environmental change accelerated with the improvement of NYS Route 73 in the 1930s. Beginning in the 1980s, the magnitude of human-caused environmental stress has further increased, as a consequence of heavy use of sand abrasive and sodium chloride chemical deicer. In the 1980s, as much as 1144 metric tons of sand and 225 tons of salt were applied in the watershed of Upper and Lower Cascade Lakes each winter, and in recent years (2001 – 2005) up to 7 tons of sand and 434 tons of sodium chloride have been applied. Chapel Pond has shown qualitatively similar but quantitatively less severe trends over time.

In this chapter, we summarize and synthesize our research findings (section 7-1), make recommendations for mitigating and reducing environmental damage caused by winter road management (section 7-2), and suggest some potential research topics that may provide data that is useful for monitoring and mitigating damage caused by winter road management in the Cascade Lakes / Chapel Pond region and throughout the Adirondack Park (section 7-3).

Section 7-1. **GENERAL CONCLUSIONS**

HISTORICAL TRENDS IN WINTER ROAD MANAGEMENT AND ENVIRONMENTAL CHANGE

Historical records indicate that the Cascade Lakes have been subject to heavy human activity in the 19th and early 20th centuries in the forms of logging, iron ore smelting, fish stocking, and maintenance of a hotel and horse livery. Recurrent wildfires, torrential rains, and landslides have also impacted the region. Nevertheless, analyses of lake sediment cores and historical data indicate that the largest changes to the Cascade Lakes and to Chapel Pond have occurred since road improvement in the 1930s, and some of the most dramatic changes have occurred beginning in the 1980s. Changes include an increase in sedimentation rate, altered sediment composition, and an increase in relative abundance of chloride-tolerant diatom species. Changes since the 1980s appear to be associated with the heavy use of sand abrasive and sodium chloride chemical deicer, and are likely also due to sediment from soil erosion along the roadside and road maintenance activities.

In the 1980s, the primary substance applied to the road was sand, with lesser (but still substantial) quantities of sodium chloride as a chemical deicer. At present, the primary substance applied to the road is sodium chloride, with minor applications of calcium chloride and sand. Current road salt application rates per run are low relative to many areas of the northeastern US, but climatic conditions result in a large number of runs over a winter, and thus result in a high annual loading to the lakes' watersheds. Current cumulative salt applications per lane km are among the highest documented in North America and Europe.

Within the last ten years, NYSDOT has experimented with using liquid magnesium chloride, and with a mixture of magnesium chloride and a sugar-based anticorrosive (MAGIC™, Sears Ecological Applications Inc.). Neither alternative chemical deicer product was judged to perform satisfactorily at NYS Route 73.

DEGRADATION OF ROADSIDE SOIL

Heavy application of sand and sodium chloride to Route 73 during winter road maintenance has resulted in dramatic changes to the physical and chemical properties of the roadside soil at the Cascade Lakes, and qualitatively similar but quantitatively less dramatic changes to roadside soil near Chapel Pond. Changes to the soil structure at the Cascade Lakes were already documented in the 1980s, but are more severe now.

The roadside soil is predominately composed of sand particles, and is denser, less permeable, and drier than soil sampled at a distance from the road.

Accumulation of sodium (from sodium chloride) has contributed to a breakdown in soil particle structure, and loss of clay and silt particles. Loss of vegetation cover (caused by changes in soil fertility and direct toxicity of road salt), and loss of clay particles have contributed to a dramatic decline in soil organic matter. As a consequence, the cation exchange capacity of the roadside soil is very low.

Accumulation of sodium ions (derived from sodium chloride) has contributed to a loss of the nutritive cations magnesium, calcium, and potassium. Chloride levels are not elevated in roadside soil, because the anion is readily transported away with surface and groundwater flow.

Other potential changes to the road soil that may have occurred as a consequence of winter road management, but which were not investigated in this study, include a probable reduction in soil nitrogen and phosphorus, a possible increase in the bioavailability or mobility of toxic metals, and a probable loss of soil microorganisms.

DAMAGE TO ROADSIDE VEGETATION

The dieback of adult paper birch (*Betula papyrifera*) trees along Route 73 at the Cascade Lakes is, in part, due to natural stand senescence. Birch in the Cascade Lakes gorge form a similar-aged cohort, as inferred from tree-ring analysis and historical records on stand recruitment. Throughout the gorge trees are dead or dieing. The die-off of this cohort is not surprising, given the natural history of this relatively short-lived species and the severe climatic conditions of the location; the trees are at the age when survival senescence typically becomes apparent.

However, there is also evidence that the loss of adult trees along the road has been accelerated relative to other sites in the gorge. One cause is direct damage and structural instability associated with rockslides and erosion as a consequence of road structural maintenance. A likely second cause is toxic accumulation of road salt on leaves and twigs from road spray and other forms of aerial deposition. The third cause is loss of soil fertility. The soil is nutrient poor, structurally unstable, and droughty. Although uninvestigated by us, the roadside soil may inhibit the colonization of mutualistic ectomycorhizal fungi.

The major difference between the roadside vegetation and elsewhere is the lack of recruitment of new plants, and overall loss of vegetative cover near the road. As a consequence of the infertility of the roadside soil, neither birch nor other trees are present as seedlings; either seeds fail to germinate or young trees are rapidly killed. Although not quantified by us, even herbaceous ground cover (including grasses) is depauperate, and there are extensive areas that completely lack ground cover.

The decline in vegetative groundcover results in lower organic matter in the soil and an increased rate of soil erosion. As a consequence, the soil becomes even less fertile, resulted in further losses in vegetative cover and lowered likelihood of recovery.

WATER QUALITY OF THE CASCADE LAKES AND CHAPEL POND

The water quality of Upper and Lower Cascade Lake has been noticeably impacted by road salt inputs. Concentrations of chloride in these lakes are upwards of 100-fold greater than expected in average Adirondack Mountain lakes. To a lesser degree, chloride contamination of Chapel Pond is noticeable but it is not as severe as in the Cascade Lakes. Despite the elevated chloride and sodium levels in the Cascade Lakes, there is no evidence to suggest that meromixis is occurring. Observations of the thermal structure in the lake over a two year period reveals these lakes to be dimictic, with isothermal conditions allowing full water column mixing in early spring and ate fall, and thermal stratification in summer and winter (inverse).

Owing to the interest in the population of round whitefish in each of these lakes, the dissolved oxygen characteristics in each lake was evaluated. All of the study lakes had a clinograde oxygen profile, characterized by a decrease in oxygen with depth during periods of thermal stratification (summer and winter). In the case of Lower Cascade Lake, hypoxia dominated the bottom 3 m of the water column throughout the summer months, a situation that could challenge the resident round whitefish population by forcing them from their ideal environment. However, the hypoxia cannot be attributed to any salt-induced cause, e.g. meromixis.

The Cascade Lakes are considered to be mesotrophic, and Chapel Pond is oligotrophic. This trophic classification is supported by both the standing crop of phytoplankton pigment (chlorophyll-a) concentrations, as well as the benthic macroinvertebrate and zooplankton density and community composition observed in these lakes.

A general observation in the Cascade Lakes was increased salt concentrations with depth. This is a function of two causes: the source of salt laden groundwater into the lake, and the washing out of the hypolimnion during all seasons since water does flow through the lakes even during ice cover.

Surveillance of the water chemistry should continue in order to build onto the data set begun in this study. The installation of a surface or subsurface buoy with conductivity and temperature sensors at various depths will serve as a continuous sentinel of water quality and assist in refining any future modeling effort. In addition, it would provide an established background data set with which to compare any changes in lake water chemistry that could arise from a change in management practice, e.g. a change in NaCl loading in the watershed or a switch from NaCl to Mg-based de-icers.

IMPACTS TO THE BIOTA OF THE CASCADE LAKES AND CHAPEL POND

Chloride concentrations in the Cascade Lakes and Chapel Pond are elevated enough to alter the composition of algal communities, as demonstrated by changes in the composition of chloride-sensitive epilimnetic diatom communities at the three lakes. Our experiments on the effects of elevated chloride concentrations on the Cascade Lakes and Chapel Pond algae (within a range of possible future concentrations given plausible loadings) indicate that periphyton and phytoplankton communities are likely to change in species composition as chloride concentrations increase, and some particular algal groups (e.g. cyanobacteria) may become more prevalent. It is unclear whether primary productivity will be altered.

The Cascade Lakes have higher primary productivity than Chapel Pond, but this is probably more a consequence of higher input of organic matter at the Cascade Lakes from the steep forested slopes surrounding them, and lake morphology (i.e. a relatively large littoral zone). Nutrient enrichment from past human activities (e.g. phosphorus-contaminated sand abrasive, waste products from past human use at the narrow land dividing the two lakes) may also have contributed to eutrophication, but is probably unimportant at present.

Benthic macroinvertebrates differ significantly among the three lakes. In terms of abundance, the Cascade Lakes are higher than Chapel Pond, which correlates with the higher primary productivity of these lakes. Upper Cascade Lake and Chapel Pond are similar in species composition and relative abundance of species, whereas Lower Cascade is quite different from these two lakes. Although Lower Cascade Lake had slightly more taxa detected than the other two lakes, the Simpson diversity index value is much lower. This indicates that relative abundances are less even in this lake, a generally-accepted indicator of environmental stress. Another taxon composition - based index (EPT/C) also indicates that Lower Cascade Lake has higher organic decomposition, which has resulted in a greater relative abundance of benthic macroinvertebrates that can tolerate low-oxygen conditions.

Research by fisheries biologists indicate that round whitefish (*Prosopium cylindraceum*) are abundant but stunted and thin for their length in comparison to Upper Cascade Lake and other Adirondack Lakes, including lakes where fish of Lower Cascade Lake origin have been stocked. Chloride concentrations in Lower Cascade Lake are well below that which is likely to be a threshold for physiological stress. A more likely cause is stressful conditions during late summer thermal stratification. Our data indicate that in late summer (e.g. August), oxygen levels are too low to support whitefish in the hypolimnion of Lower Cascade Lake, whereas the temperature in the epilimnion is too high. Either fish are frequenting areas of the lake that cause thermal or respiratory stress, or they are crowded in a narrow belt of the metalimnion (thermocline). Spatial mapping of the distribution of whitefish, behavioral observations, and diet analysis during summer thermal stratification should help at understanding how the fish cope with physiologically stressful conditions.

Low oxygen conditions in the hypolimnion of Lower Cascade Lake in late summer are a consequence of the lake's morphology (large littoral zone plus one small, deep basin), and the morphology of the surrounding watershed, which results in high input of organic matter into the lake. The strong chloride concentration gradient that forms in Lower Cascade Lake could potentially result in a delay of thermal mixing in fall, prolonging the period of physiological stress to whitefish. By comparing the timing of turnover at Upper Cascade versus Lower Cascade Lake, we detect no indication of resistance to thermal mixing. Because the chloride concentration gradient is expected to intensify in the next few years (see next section), prolonged stratification should remain a future concern, as should the possibility of Lower Cascade Lake becoming meromictic, which would be catastrophic for whitefish and most other elements of the lake biota.

FUTURE TRENDS

Without active remediation, the roadside soils at the Cascade Lakes will remain infertile and prone to erosion. Sheet erosion will continue, because vegetative cover is unlikely to reestablish without active management. Chapel Pond roadside soils approach the conditions of the Cascade Lakes in some spots, and we suspect that soil fertility will continue to decline at Chapel Pond due to heavy loading of sodium chloride, and may eventually approach the severely degraded condition of the Cascade Lakes roadside.

Our transport model of chloride at the Cascade Lakes predicts that chloride concentrations will continue to rise over the next five to ten years, and the gradient in concentration with depth will intensify. The Lower Cascade hypolimnion, which presently has the highest chloride levels, will show the greatest increase (around 40%). After 2015, the model predicts that peak chloride concentrations in the deepest section of Lower Cascade Lake will approach the USEPA recommended maximum limits for chronic exposure to aquatic life (230 ppm). Lower Cascade Lake will be at heightened risk of becoming meromictic.

The chloride transport model also indicates that there is a lag in the system between deicer application onto Route 73 and eventual transport into one of the Cascade Lakes. Based on the model, it appears that it takes about seven years for lake chloride levels to reequilibrate after a change in the amount of sodium chloride applied to the road. This may explain the observed lag between the increase in salt applications in the late 1990s onto NYS Route 73 and the apparent sharp increase in chloride concentrations in the Cascade Lakes a few years later.

There is a 1:1 correspondence between salt loading on NYS Route 73 and the concentration of chloride in each lake. Doubling the annual salt loading would double each lake's concentrations of chloride, resulting in chloride levels that exceed USEPA guidelines at all depths of each lake. Halving the salt loading would halve the concentration of chloride in each lake. The chloride transport model successfully predicted lake chloride concentrations measured in the 1990s, when salt loadings were half that of the present. If sodium chloride deicer use is terminated, for instance if there would be a switch to an acetate-based chemical deicer, chloride levels are predicted to drop to near zero within ten years.

Section 7-2. **RECOMMENDATIONS**

CONTINUE MONITORING OF LOWER CASCADE LAKE

Our findings show that Lower Cascade Lake is under increasing stress: conditions are presently poor during summer thermal stratification for organisms that require cold, well-oxygenated water (e.g. round whitefish), and chloride concentrations are increasing, as is the magnitude of the chloride gradient with depth. Although presently a dimictic lake with no evidence that the chloride concentration gradient impedes fall thermal mixing (fall turnover), the lake is at increasing risk of becoming meromictic, which would be catastrophic for fish and many other endemic aquatic organisms.

We recommend that this lake be monitored via water sampling, ideally quarterly, but at a minimum twice per year. Sampling should be done in April, when complete thermal mixing is expected and dissolved oxygen concentrations high, and in August, when the strongest thermal stratification is expected to occur and oxygen depletion at depth should be most severe. Minimally the following should be measured, at the deepest point in the lake (our sampling station) and at a uniform distribution of depths from the surface water to the lake bottom: water temperature, dissolved oxygen, chlorophyll-a, secchi depth, and the chloride concentration. If practical, it would be worthwhile to sample Upper Cascade Lake in the same manner.

The information provided by periodic monitoring will allow managers to continuously evaluate Lower Cascade Lake in terms of environmental conditions affecting organisms that require cold, oxygen-rich water, and in terms of productivity. Monitoring will also alert managers whether the lake is more at risk of becoming meromictic, e.g. via intensification of the dissolved sodium chloride gradient.

ROADSIDE REVEGETATION

We concur with the recommendation of Fleck et al. (1988) that roadside revegetation at the Cascade Lakes be initiated. Successful reestablishment of ground cover will stabilize soil and reduce erosion, and provide much-needed organic matter into the soil. By doing so, it will facilitate the natural reestablishment of vegetative cover. Revegetation with shrubs may also serve as a barrier to road splash. Reestablishment of vegetation should improve the site aesthetics for visitors and motorists.

Given current winter road management practices, we believe it is unrealistic to attempt to reestablish paper birch or other large native trees at this time. Unfortunately, few native plant species that tolerate the climate of the high-altitude Adirondacks are also tolerant of high salt contamination. In our review and in consulting with authorities on Adirondack plants, we have identified two shrubs and three graminoids that may be good candidates for revegetation. Information on each species, including links to information on natural history and propagation, are found at the New York State Flora Atlas (Weldy and Werier 2005).

Red-osier dogwood (*Cornus sericea*) grows in the vicinity of the Cascade Lakes, and has been shown to tolerate heavy exposure to sodium chloride in soil. Unfortunately, it is classified as very sensitive to airborne highway spray. We have observed that although branch tips dieback, from road-salt and other causes, vegetative parts under the snow pack persist. Perhaps because of the 'tip-pruning', and also because of clonal reproduction, dogwood shrubs form a low, thick hedge. Nurseries grow red-osier dogwood, so it should be possible to acquire numbers for planting. Because of the infertile condition of the roadside soil, shrubs should be grown and planted in perforated biodegradable nursery pots, so that they have some nutrient-rich, fertile soil within which to become established before spreading into the surrounding, infertile soil.

Dwarf sand cherry (*Prunus pumila*) has also been suggested. This shrub is an Adirondack native, and tolerant of cold, sandy conditions. According to D. Leopold. (2005, *personal comm.*) *P. pumila* forms dense thickets about 3 feet high, and is an excellent soil stabilizer, especially of dry, sandy soils. It is tolerant of rocky soils, salt spray and alkaline conditions.

One grass that could be used in revegetation is slender wheatgrass (*Elymus trachycaulus* = *Agropyron trachycaulum*). Since it is used elsewhere for mitigation of salt contaminated soil in cold boreal regions, it should be suitable at the Cascade Lakes. As with dogwood, it may be necessary to establish initial plantings in perforated, biodegradable pots, with the expectation that established plants will spread onto the infertile roadside soil. Two other Adirondack native graminoids that have been suggested as having promise for revegetation, and which appear likely to tolerate the cold, sandy, nutrient poor conditions of the Router 73 roadside, are little bluestem (*Schizachyrium scoparium* = *Andropogon scoparius*) and fresh water cordgrass (*Spartina pectinata*). Both species would appear to have desirable characteristics (e.g. clonal spread) for effective revegetation.

It will require some carefully monitored experimentation, and consultation with ecologists, landscape architects and other appropriately trained professionals familiar with restoration ecology to develop an optimal strategy for species selection and propagation. However, the outcome is likely to be highly positive both in terms of environmental remediation and in terms of public perception.

EDUCATIONAL SIGNAGE

Given the widespread public concern about the environmental impacts of winter road management on the Cascade Lakes, and on the wider Adirondack region, we believe that it would be worthwhile to place educational signs providing an overview of NYSDOT's environmental research projects and snow and ice control program. These signs could be placed at the Cascade Lakes recreation area, at the Adirondack Northway (Interstate 87) Adirondack Park gateway rest area, and at the Chapel Pond parking area. Other potential locations for educational displays include the two Visitor Interpretive Centers (Paul Smiths, Newcomb) and the new Museum of Adirondack Natural History (Tupper Lake). We believe that Adirondack visitors will appreciate knowing that NYSDOT (and NYSDEC, and the APA) are concerned about the environmental impacts of winter road management, and are taking measures to reduce these impacts without compromising motorist safety.

Section 7-3. **RESEARCH NEEDS**

MEASURES TO REDUCE SALT LOADINGS TO THE CASCADE LAKES

Based on our numerical model, and given current winter management practices along Route 73, the concentration of chloride in the epilimnion (surface waters) of the Cascade Lakes will increase by a further 20% over the next decade, and the hypolimnion (bottom water) of Lower Cascade Lake will increase by 40%. While this does not appear to be high enough to significantly increase toxicity to most aquatic organisms, it does increase the risk of prolonged thermal stratification or meromixis, especially at Lower Cascade Lake.

A 20% reduction in salt loading to the Cascade Lakes watershed is likely to maintain the current level of chloride concentration in the Cascade Lakes, according to our model. To attain this level of deicing salt reduction while maintaining the required level of service will be difficult. From our discussions with NYSDOT personnel involved in winter management of this stretch of highway, a consensus exists that it is desirable to reduce chemical deicer use, and it also appears that the conventional best practices for reducing salt usage have been implemented (calibration of truck application rates, pre-storm anti-icing). We offer the following points as potential measures for further reducing salt usage. Each requires further research in terms of effectiveness, resource demands, environmental impact, and cost before being considered for implementation.

Controlling Surface Runoff and Wind-aided Deposition

Based on the comments of NYSDOT personnel involved in winter road management at Cascade Pass and our own observations, it is clear that a major reason why chemical deicer and abrasive loadings are historically so high at the Cascade Pass is the need for frequent applications of material due to wind deposition of snow onto the surface of the road and meltwater runoff onto the road surface from the adjacent mountain slope. It appears that the most effective ways to reduce the use of chemical deicer are to reduce wind deposition of snow and meltwater runoff onto the surface of the highway.

The orientation and topography of the valley result in strong, gusty winds that can shift in direction in unexpected ways. Snow fences and windbreaks would be desirable, but space limitations and orientation of the road relative to the prevailing wind direction make such technologies extremely difficult to install at Cascade Pass. We suggest that engineers with experience at developing innovative designs of snow fencing be consulted about feasible designs for this site.

The extremely narrow space between the road and the mountain slope and the presence of bedrock at the ground surface effectively limit construction of a conventional drainage ditch to prevent runoff onto the road surface. Currently, culverts are used to direct runoff flow under the road but these culverts are frequently blocked by ice, and therefore require clearing by mechanical and chemical (calcium chloride) measures. We suggest that engineers with experience at developing innovative designs for managing runoff along roads be consulted about feasible alternative designs for culverts, drainage ditches, and other technologies that are appropriate for Route 73 at the Cascade Pass.

Weather Station, Pavement Monitoring

NYSDOT personnel in the Cascade Lakes region report that they generally have an accurate knowledge of weather conditions at Cascade Pass, based on reports by operators and other personnel who frequently drive the road. Nevertheless, we suggest installation of a Remote Weather Information Station (RWIS) at the Cascade Lakes, to provide managers with accurate weather conditions for tailoring winter road management to the local weather conditions. Such a system would minimally include (1) a station to provide weather information, including wind speed and direction, air temperature, relative humidity, barometric pressure, solar radiation, and precipitation (amount and form); and (2) two or more road sensors for monitoring road conditions, including pavement temperature, wind speed, road surface condition (e.g. dry, wet, ice, snow), water film level, and residual salt content of the water film (for a commercial example, see Campbell Scientific Inc.: http://www.campbellsci.com/road-weather). Information would be relayed to the local NYSDOT residency, where managers could then apply recommended best practices for winter road management based on accurate site-specific information.

In principal, chemical deicers and abrasives will be used more efficiently (and therefore in lower amounts) because of better information. Potential obstacles to installing RWIS at the Cascade Lakes include the lack of electricity at the site, questionable cell phone service, and the need for permitting approval required by APA and NYSDEC. Should a functioning RWIS system be installed, we recommend that there be direct monitoring of materials use, and interviews with operators on the usefulness of the information provided by the RWIS and pavement monitors. The results of this monitoring will allow NYSDOT to determine whether deicer and abrasive applications are optimal for the desired level of service, and whether the increase in efficiency provided by the RWIS and pavement monitors are sufficient to compensate for the added expense of installing, maintaining, and using such systems. If so, wider use on the Adirondack Park highways and elsewhere may be warranted.

Lower Driving Speed

We suggest that a study be conducted by NYSDOT traffic engineers on whether it would be advisable to lower the speed limit along Route 73 at the Cascade Lakes during the snow and ice control season. By lowering driving speed, the amount and distance of chemical deicer dispersed to the roadside via scatter, road splash and aerial transport can potentially be reduced. In theory, less chemical deicer need be applied to maintain the same level of service, because deicer will be less rapidly transported off the road surface. Lowering the speed limit will also reduce the zone of intense aerial deposition of deicer, and thus narrowing the zone of impacted vegetation. Lowering driving speed may also increase safety on this stretch of road, which includes several small pull-offs, a difficult intersection at Route 73 and the Cascade Lakes recreational area, and significant recreational use of the highway corridor even during the snow and ice control season. Even if traffic data analysis determines that seasonal speed reduction is feasible, motorist behavior and weak enforcement will likely limit the achieved effectiveness, however.

MONITORED EXPERIMENTAL APPLICATION OF ALTERNATIVE CHEMICAL DEICERS

Under the present winter road management practices, the environmental impact of chemical deicer use at the Cascade Lakes is higher than similar stretches of Adirondack Highway studied, e.g. Chapel Pond, and is likely to increase somewhat in the next two decades. Three alternatives to the currently favored material, sodium chloride, have been applied on Route 73 at this stretch of highway: sand, liquid magnesium chloride, and agricultural-based deicing chemicals. Sand caused severe degradation of roadside soil and sedimentation problems in the lakes. The two first-generation magnesium chloride agricultural – based products did not perform adequately at this site, according to NYSDOT personnel at the Essex County residency.

The most promising remaining alternative chemical deicer, in terms of potentially reducing the environmental impact of chemical deicer, is calcium magnesium acetate (CMA), which has very low toxicity to terrestrial plants and aquatic organisms, and can improve soil fertility by maintaining soil structure, and preventing loss of nutrients. It can be applied as a solid or liquid, and can be applied as a mixture with sodium chloride. Unfortunately, in addition to CMAs extremely high cost, several site-specific factors make the use of CMA problematic, especially at this remote location. Included in these operational issues are: the need to construct additional storage facilities for the CMA and very limited space to do so, the need to add an additional dedicated "CMA only" truck to the Essex County Residency fleet, and the need to add an additional crew to operate the CMA truck.

Given these immediate and long-term issues related to CMA use, including permitting approval from APA and NYSDEC and additional funding and staffing, we recommend that the following alternatives be considered for use in the present term: (1) Pre-wetted salt with liquid Calcium Chloride or Magnesium Chloride; or (2) New generation "treated salts", e.g. I.M.U.S. Inc.'s Magic (carbohydrate plus MgCl) or Cargill Inc.'s Clearlane Enhanced (MgCl with no carbohydrate). Each of these three chemical deicer modifications will increase costs over existing practice. However, they will require less equipment retro-fitting, less storage facility modification, will not require additional staff - as compared to CMA - and could lower application rates of sodium chloride by approximately 20%. The experimental application will need to be carefully monitored for effectiveness, particularly since each of these alternatives is also a source of chloride loading, and therefore can potentially increase the chloride concentration of the lakes, should anticipated reductions in material use not be achieved. If monitored sodium chloride usage does not decrease by 20% or if water quality monitoring does not demonstrate improved water quality, NYSDOT should consider initiating a CMA pilot program at the Cascade Lakes as discussed in the following paragraphs.

The biggest environmental risk of CMA is its potential effects on biochemical oxygen demand (BOD), particularly in small, deep ice-covered roadside lakes. Microbial breakdown of acetate, which normally occurs within two weeks of application (it may be slower during very cold weather, but will occur rapidly once conditions have warmed above freezing), can result in lowered dissolved oxygen, and thus a source of stress to organisms with high oxygen demands (e.g. cold-water fish such as round whitefish and lake trout).

Lower Cascade Lake appears to have morphological characteristics that make it most vulnerable to elevated BOD, and indeed the lake has severely low dissolved oxygen in the hypolimnion in the late summer. Since acetate breakdown is rapid, elevated BOD is transitory and occurs within two weeks of application. Thus, elevated BOD will occur in early winter and early spring, at a time when oxygen levels are normally high in Lower Cascade Lake, and acetate deicer will have no effect on oxygen demand in late summer, the most critical time based on our two year dataset.

The potential benefits of using CMA include: (1) improved fertility of roadside soils; (2) elimination of dieback of vegetation due to aerial (or root) exposure to road salt; and (3) an eventual decline in chloride levels within the lake, reducing the risk of meromixis. Use of CMA has inherent risk, however, because of its effects on BOD.

We recommend using CMA as a principal or supplemental chemical deicer for a period of time, as the focus of a carefully-planned and monitored management experiment. CMA would be applied in conjunction with sodium chloride. We recommend using the model we have created for predicting chloride concentrations in the Cascade Lakes to estimate how much CMA must be used in place of rock salt to maintain current water quality at the Cascade Lakes. CMA works best at temperatures above -7 °C (20 °F), slightly higher than sodium chloride rock salt; other materials would be necessary to when pavement temperatures are colder than this.

The composition of different CMA products should be evaluated beforehand, since some products have excessive concentrations of phosphorus and nitrogen as a byproduct of manufacture. Detailed, accurate records should be maintained of application rates and timing. Water samples should be regularly collected (e.g. monthly) at Lower Cascade Lake (and ideally Upper Cascade Lake) at a uniform distribution of depths, and analyzed for dissolved oxygen, chlorophyll a, secchi depth, and chloride. These data would indicate whether at Cascade Lakes the use of CMA causes excessive oxygen depletion, provides a carbon source that promotes excessive phytoplankton growth, and results in a reduction in lake chloride levels similar to the predictions from our chloride transport model. If significant negative impacts to the lake water quality are detected (i.e., excessive oxygen depletion, excessive algal growth), use of CMA should be rapidly terminated and sodium chloride (or other chemical deicer) be substituted. If no significant negative environmental impacts are detected, and sodium levels begin to decline (predictably after a lag period corresponding to the displacement of monovalent Na in soil by divalent Ca and Mg ions), then CMA may be a worthwhile product for Route 73 in the Cascade Lakes region, and potentially other environmentally sensitive sites, assuming that it also performs adequately from the perspective of maintaining road safety and isn't too expensive.

Practical objections to experimental use of CMA include the cost of the material itself and the capital costs of creating a storage structure and purchasing any specialized equipment needed for effective spreading. It was beyond the scope of the present study to estimate the costs of implementing a CMA feasibility study, but we note that in recent years, costs of using CMA deicer has been declining and technology is improving (e.g., Ormsby 1999). Much information on practical issues related to use of CMA is found at the website of a commercial vendor, Cryotech Deicing Technology (http://www.cryotech.com/products/commercial.php).

LONGTERM MEASURES FOR REDUCING DEICER: STRUCTURAL DESIGN

No major capital improvements are planned in the near future for Route 73 at the Cascade Lakes. However, because of the geological instability of the area and the low surface rating of that section of roadway, it is inevitable that the road will need reconstruction in the future. Road reconstruction would provide the opportunity to address many of the issues discussed in this report. In particular, road reconstruction would provide an opportunity to investigate using surface materials that reduce ice bonding, coupled with barriers that prevent winter precipitation from being deposited or blowing onto the road surface. Ideally, a passive system for control of ice and snow of roads is desirable.

SafeLane™ Surface Overlay (by Cargill Inc.) provides a potential solution as an anti-icing pavement overlay. SafeLane™ is made up of a combination of epoxy and aggregate rock which absorbs and stores liquid deicer (which is applied before a winter storm event). The overlay then automatically releases the chemical deicer as conditions develop for the formation of ice or snow. Although Safelane™ does not eliminate the use of chemical deicer, it can substantially reduce the amounts used by absorbing/storing the liquid deicer and thus preventing the chemical from being removed via surface runoff, tire splash, or plowing. And the passive nature of the overlay can also reduce the need to send winter maintenance crews out in adverse icy and snowy weather. To date the majority of the applications of SafeLane™ overlays have been to bridge decks – further research is recommended on extending the performance characteristics of the overlay to a linear application such as the roadway along the Route 73 Cascade Pass.

Another alternative winter management practice that, in this case, fully eliminates the need for deicing chemicals while maintaining high levels of safety is the installation of an electrically conductive concrete overlay. This option would involve potentially higher construction costs and would be most feasible during complete reconstruction of the highway. Conductive concrete overlays capable of using resistance heat to clear snow and ice from forming on roadways have few known negative environmental impacts. The conductive concrete consists of a modified concrete mix design that includes steel fibers and other conductive aggregates. When electrically energized, the resistance of the concrete releases heat capable of breaking ice and snow bonding to roadway, and thus the conductive concrete overlay is effective at deicing and anti-icing applications. Commonly, such overlays take their power from offline but other more sustainable energy production such as wind or solar power could be utilized to make a self-sustainable system. Given that Cascade Pass is a remote location at a bottom of a deep gorge, powering a conductive concrete overlay along Route 73 at the Cascade Lakes will be challenging and require further research. Further, while the technology has been successfully applied to bridge decks in recent years, the extension of the use of conductive concrete to linear transportation projects such as roadways needs further study. Preliminary studies on the optimization of conductive concrete mix designs for such applications have been carried out at Clarkson University and show promise.

In order to be ready with innovative solutions when road reconstruction becomes a reality in the future, further research on structural design improvements including advanced surface materials is recommended as soon as possible. The goal of such research would help identify and eventually develop feasible structural alternatives to lower chemical deicer use on sections of roads where environmental concerns are especially high.

RESURVEY ADIRONDACK LAKES AT RISK OF ROAD SALT CONTAMINATION

The Adirondack Lake Survey Corporation (ALSC) survey of 1469 Adirondack lakes in the mid 1980s provides an invaluable database with which to compare present conditions at ponds and lakes that are subject to chemical deicer contamination. The ALSC data provides surface-water measurements of chloride concentrations, which for water bodies near roads may be affected by deicing salt contamination, during a period when sodium chloride applications were half of the present loadings.

Using geographic information systems (GIS) technology, and information provided by the ALSC on lake morphology, it is simple to identify lakes within the data set that are most likely to be subject to heavy chemical deicer contamination, and lakes where such contamination is extremely unlikely. Present loadings of road salt to lakes can be estimated based on agency data (e.g., NYSDOT, County Highway Dept.) and GIS analysis of the roadway and watershed. Sampling a set of each category of lakes (at risk of road contamination versus remote from roads) would provide information on changes in environmental health of Adirondack lakes since the 1980s caused by winter road maintenance, and provide a data set that would aid in accurately identifying lakes that are most at risk of environmental degradation because of road management activities. These data should be invaluable to environmental managers both within the Adirondack Park and elsewhere in North America subject to similar environmental conditions (e.g. New England and eastern Canada).

In addition to water quality monitoring, a sediment core could be collected from each lake, or a subset of lakes, for reconstruction of environmental change caused by road maintenance activities, including road construction and repair and winter road maintenance. Based on the Cascade Lakes and Chapel Pond sediment core data, it is clear that high quality information can be gained about environmental change over the last two centuries based on analysis of sedimentation rates and composition, and water quality reconstruction using diatoms. Diatom data provide accurate information on concentrations of contaminants such as chloride, on other environmental stressors, and on primary productivity. Used in conjunction with historical (ALSC) water quality data and contemporaneous sampling, insight into the chronic stresses caused by winter road management and other factors can be assessed for each sampled lake.

LITERATURE CITED

Adirondack Council. 2003. Summer Newsletter.

Albright, M. 2005. Changes in water quality in an urban stream following the use of organically derived deicing products. *Lake and Reservoir Management* 21:119-124.

Allen, J.A., J.L. Chambers, M. Stine. 1994. Prospects for increasing the salt tolerance of forest trees: a review. *Tree Physiology* 14: 843-853.

Amrhein, C., J.E. Strong. 1990. The effect of deicing salts on trace metal mobility in roadside soils. *Journal of Environmental Quality* 19:765-772.

Amrhein, C., J.E. Strong, P.A. Mosher. 1992. Effect of deicing salts on metal and organic matter mobilization in roadside soils. *Environmental Science and Technology* 26:703-709.

Appleton, B., V. Greene, A. Smith, S. French. 2003. *Trees and Shrubs that Tolerate Saline Soils and Salt Spray Drift*. Virginia Cooperative Extension Publication 430-03.

Armstrong, J.W., C.R. Liston, P.I. Tack, R.C. Anderson. 1977. Age, growth, maturity, and seasonal food habits of round whitefish, *Prosopium cylindraceum*, in Lake Michigan near Ludington, Michigan. *Transactions of the American Fisheries Society* 106:151-155.

Bäckström, M., S. Karlsson, L. Bächman, L. Folkeson, B. Lind. 2004. Mobilisation of heavy metals by deicing salts in a roadside environment. *Water Research* 38: 720-732.

Barbour, M.T.J., J. Gerritsen, B.D. Snyder, J.B. Stribling. 1999. *Rapid Bioassessment Protocols for use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish*, 2nd edition. *EPA 841-B-99-002*. US Environmental Protection Agency, Office of Water, Washington D.C.

Bailey, J. 1980. *History of the Town of Keene, NY.*

Bang, S.S., D. Johnston. 1998. Environmental effects of sodium acetate/formate deicer, Ice Shear™. *Archives of Environmental Contamination and Toxicology* 35: 580-587.

Baroga, E.V. 2005. *2002-2004 Salt Pilot Project*. Washington State Department of Transportation.

Benbow, M.E., R.W. Merritt. 2004. Road-salt toxicity of selected Michigan macroinvertebrates under different testing conditions. *Wetlands* 24:68-76.

Bicknell, S.H., W.H. Smith. 1975. Influence of soil salt, at levels characteristic of some roadside environments, on the germination of certain seed species. *Plant and Soil* 43:719-722.

Binford M. 1990. Calculation and uncertainty analysis of 210-Pb dates for PIRLA project lake sediment cores. *Journal of Paleolimnology* 3: 253-267.

Blackburn, R.R., K.M. Bauer, D.E. Amsler Jr., E.E. Boselly, A.D. McElroy. 2004. *Snow and Ice Control: Guidelines for Materials and Methods*. National Cooperative Highway Research Program Report 526. National Transportation Board, Washington D.C.

Blasius, B.J., R.W. Meritt. 2002. Field and laboratory investigations on the effects of road salt (NaCl) on stream macroinvertebrate communities. *Environmental Pollution* 120:219-231.

Blomqvist, G. 1998. *The Impact of De-Icing Salt on Roadside Vegetation. A Review*. VTI rapport 427A.

Blomqvist, G. 2002. De-icing salt and roadside environment – Strategies for impact analysis. *Proceedings of the XI International PIARC Winter Road Congress*. 28–31 January 2002, Sapporo, Japan.

Blomqvist, G. 2003. Patterns of chloride deposition next to a road as influenced by salting occasions and winds. *Proceedings of the Transportation Research Board 82nd Annual Meeting*, 12–15 January 2003, Washington D.C.

Blomqvist, G., E. Johansson. 1999. Airborne spreading and deposition of deicing salt – a case study. *Science of the Total Environment* 235:161-168.

Bode, R.W., M.A. Novak, L.E. Abele. 1991. Methods for rapid biological assessment of streams. New York State Department of Environmental Conservation, Albany New York.

Bode, R.W., M.A. Novak, L.E. Abele. 2002. *Quality Assurance Work Plan for Biological Stream Monitoring in New York State*. New York State Department of Environmental Conservation, Albany, New York.

Bogemans, J., L. Neirinckx, L., J.M. Stassart. 1989. Effect of deicing chloride salts on ion accumulation in spruce (*Picea abies*). *Plant and Soil* 113: 3-11.

Bowser, C.J. 1992. Groundwater pathways for chloride pollution of lakes. Pages 283-302 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Buckler, D.R., G.E. Granato. 1999. *Assessing Biological Effects from Highway-Runoff Constituents.* USGS Open-File Report 99-240.

Bryson, G.M., A.V. Barker. 2002. Sodium accumulations in soils and plants along Massachusetts roadsides. *Communications in Soil Science and Plant Analysis* 33: 67-78.

Bubeck, R.C., W.H. Diment, B.L. Deck, A.L. Baldwin, S.D. Lipton. 1971. Runoff of deicing salt: Effect on Irondequoit Bay, Rochester, New York. *Science* 172:1128-1132.

Burns, R.M, B.H. Honkala. 1990. *Silvics of North America*. Agricultural Handbook 654, Volume 2: Hardwoods. United States Department of Agriculture, Forest Service, Washington, D.C.

Butler, B.J., J. Addison. 2000. *Biological Effects of Road Salts on Soils*. Report submitted to the Environment Canada CEPA Priority Substances List Environmental Resource Group on Road Salt. Commercial Chemicals Evaluation Branch, Environment Canada, Hull Quebec.

Buttle, J.M., C.F. Labadia. 1999. Deicing salt accumulation and loss in highway snowbanks. *Journal of Environmental Quality* 28: 155-164.

Cain, N.P., B. Hale, E. Berkelaar, D. Morin. 2001. *Review of the Effects of NaCl And other Road Salts on Terrestrial Vegetation in Canada*. Report submitted to the Environment Canada CEPA Priority Substances List Environmental Resource Group on Road Salt. Existing Substances Branch, Environment Canada, Hull Quebec.

Campbell, K.A., C.D.B. Hawkins. 2003. Paper birch and lodgepole pine root reinforcement in coarse-, medium-, and fine-textured soils. *Canadian Journal of Forest Research* 33:1580-1586.

Carlson, D.M. 2002. *New York State Draft Recover Plan for Round Whitefish (Prosopium cylindraceum)*. New York State Department of Environmental Conservation.

Catling, PM, S.M. McKay. 1980. Halophytic plants in southern Ontario. *Canadian Field-Naturalist* 94:248-258.

Chapra, S.C. 1997. *Surface Water-Quality Modeling*. New York, New York: WCB McGraw-Hill.

Commission on the Adirondacks in the 21st Century. 1990. *The Adirondack Park in the 21st Century*. Published by New York State.

Connolly, J.P., P.R. Paquin, T.J. Mulligan, K-B. Wu, L. Davanzo. 1990. Calcium magnesium acetate biodegradation and it simpact on surface waters. Pp. 140-156 in *The Environmental Impact of Highway Deicing* (C.R. Goldman, G.M. Malyj, eds.). Proceedings of a symposium held 13 October 1989 at University of California Davis. Insitute of Ecology Publication 133.

Croser, C., S. Renault, J. Franklin, J. Zwiazek. 2001. The effect of salinity on the emergence and seedling growth of *Picea mariana*, *Picea glauca*, and *Pinus banksiana*. *Environmental Pollution* 115: 9-16.

Crowther, R.A., H.B.N. Hynes. 1977. The effect of road deicing salt on the drift of stream benthos. *Environmental Pollution* 14:113-125.

Curran, R.P., W.S. O'Neil, D.S. Smith. 1990. The environmental implications of development. Pages 346-405 in *The Adirondack Park in the 21st Century: Technical Reports Volume 2*, Commission on the Adirondacks in the 21st Century (eds.). Published by New York State.

Czerniawska-Kusza, I, G. Kusza, M. Duzynski. 2004. Effects of deicing salts on urban soils and health status of roadside trees in the Opole region. *Environmental Toxicology* 19:296-301.

Demers, C.L. 1992. Effects of deicing road salt on aquatic invertebrates in four Adirondack streams. Pages 245-252 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Demers, C.L., R.W. Sage Jr. 1990. Effects of road deicing salt on chloride levels in four Adirondack streams. *Water, Air, and Soil Pollution* 49:369-73.

Dickman, M.D., M.B. Gochnauer. 1978. Impact of sodium chloride on the microbiota of a small stream. *Environmental Pollution* 17:109-126.

Dillon, P.J., B.J. Clark, L.A. Molot, H.E. Evans. 2003. Predicting the location of optimal habitat boundaries for lake trout (*Salvelinus namaycush)* in Canadian Shield Lakes. *Canadian Journal of Fisheries and Aquatic Science* 60:959-970.

Dirr, M.A. 1976. Selection of trees for tolerance to salt injury. *Journal of Arboriculture* 2:209-216.

Dirr, M.A. 1978. Tolerance of seven woody ornamentals to soil-applied sodium chloride. *Journal of Arboriculture* 4:162-165.

D'Itri, F.M. 1992. *Chemical Deicers and the Environment*. Lewis Publishers, Boca Raton, Florida.

Dixit, S., J.P. Smol. 1994. Diatoms as indicators in the Environmental Monitoring and Assessment Program - Surface Waters (EMAP-SW). *Environmental Monitoring and Assessment* 31: 275-306.

Dixit, S.S., J.P. Smol, D.F. Charles, R.M. Hughes, S.G. Paulsen, G.B. Collins. 1999. Assessing water quality changes in lakes of the northeastern United States using sediment diatoms. *Canadian Journal of Fisheries and Aquatic Science* 56:131-152.

Dixit, S.S., A. Dixit, J.P. Smol, R.M. Hughes, S.G. Paulsen. 2000. Water quality changes from human activities in three northeastern USA lakes. *Lake and Reservoir Management* 16:305-3291.

Dobson, M.C. 1991. De-icing salt damage to trees and shrubs. Bulletin 101. Forestry Commission, Surrey England.

Driscoll, C.T., G.B. Lawrence, A.J. Bulger, T.J. Butler, C.S. Cronan, C. Eager, K.F. Lambert, G.E. Likens, J.L. Stoddard, K.C. Weathers. Acidic deposition in the northeastern United States: Sources and inputs, ecosystem effects, and management strategies. 2001. *Bioscience* 51:180-198.

Driscoll, C.T., K.M. Driscoll, K.M. Roy, M.J. Mitchell. 2003. Chemical response of lakes in the Adirondack region of New York to declines in acidic deposition. *Environmental Science and Technology* 37:2036-2042.

Eaton, L.J., J. Hoyle, A. King. 1998. Effects of deicing salt on lowbush blueberry flowering and yield. *Canadian Journal of Plant Science* 126:125-128.

Elliott, H.A., J.H. Linn. 1987. Effects of calcium magnesium acetate on heavy metal mobility in soils. *Journal of Environmental Quality* 16:222-226.

Environment Canada and Health Canada. 1999. *Environmental Protection Act 1999. Priority Substances List Assessment Report – Road Salt*.

ESRI. 2002. *Using ArcGIS 3D Analyst.* ESRI Publishing, Riverside CA.

Evans, M., C. Frick. 2001. *The Effects of Road Salts on Aquatic Ecosystems*. National Water Research Institute, Contribution 01-000.

Feick, G. 1972. Release of mercury from contaminated freshwater sediments by the runoff of road deicing salt. *Science* 175:1142-1143.

Fetter, C.W. 2001. *Applied Hydrogeology*. 4th ed. Prentice Hall: New Jersey.

Fischel, M. 2001. *Evaluation of Selected Deicers Based on a Review of the Literature*. Colorado Department of Transportation Research Branch, CDOT-DTD-R-2001-15.

Fleck, A.M., M.J.Lacki, J. Sutherland. 1988. Response by white birch (*Betula papyrifera*) to road salt applications at Cascade Lakes, New York. *Journal of Environmental Management* 27:369-377.

Forman, R.T.T., L.E. Alexander. 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics* 29: 207-231.

Forman, R.T.T., D. Sperling, J.A. Bissonette, A.P. Clevenger, C.D. Cutshall, V.H. Dale, L. Fahrig, R. France, C.R. Goldman, K. Heanue, J.A. Jones, F.J. Swanson, T. Turrentine, T.C. Winter. 2003. *Road Ecology: Science and Solutions*. Island Press, Washington D.C.

Fostad, O., P.A. Pedersen. 2000. Container-grown tree seedling responses to sodium chloride applications in different substrates. *Environmental Pollution* 109:203-210.

Foster, A.C., M.A. Maun. 1978. Concentration of highway deicing agents along roadsides near London. *Canadian Journal of Botany* 56:1081-1085.

Gales, J.E., J. VanderMeulen. 1992. Deicing chemical use on the Michigan State Highway System.. Pages 135-184 in *Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Gibson, C.E., D.A. Stewart. 1972. Changes in the water chemistry of Lough Neagh over a 10-year period. *Limnology and Oceanography* 17: 633-35.

Godwin, K.S., S.D. Hafner, M.F. Buff. 2003. Long-term trends in sodium and chloride in the Mohawk River, New York: the effect of fifty years of road-salt application. *Environmental Pollution* 124:273-81.

Goldman, C.R., F.S. Lubnow, J.J. Elser. 1992. Environmental effects of calcium magnesium acetate on natural phytoplankton and bacterial communities in northern California lakes. Pages 229-244 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Granato, G.E., P.E. Church, V.J. Stone. 1993. Mobilization of major and trace constituents of highway runoff in groundwater potentially caused by deicing chemical migration. *Transportation Research Record* 1483:92-103.

Greenway. H., R. Munns. 1980. Mechanisms of salt tolerance in nonhalophytes. *Annual Review of Plant Physiology* 31:149-190.

Gunn, J.M., R. Pitblado. 2004. Lake trout, the Boreal Shield, and the factors that shape lake trout ecosystems. Pages 3-19 in *Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment* (J.M. Gunn, R.J. Steedman, R.A. Ryder, eds.). CRC Press, Boca Raton, Florida.

Hakala, A. Meromixis as a part of lake evolution – observations and a revised classification of true meromictic lakes in Finland. *Boreal Environment Research* 9:37-53.

Hart, B.T., P. Bailey, R. Edwards, K. Hortle, K. James, A. McMahon, C. Meredith, K. Swadling. 1991. A review of the salt sensitivity of the Australian freshwater biota. *Hydrobiologia* 210:105-144.

Hausenbuiller, R.L. 1985. *Soil Science Principles and Practices*, 3rd ed. Wm. C. Brown Company Publishers.

Hauser, E., A.R. Wunderlich 2004. *Rising Salt Concentrations in Tributaries of the Hudson River Estuary*. Kingston, New York, 6 December 2004. http://www2.marist.edu/~en04/PUBS.HTML

Hanes, R.E., L.W. Zelazny, R.E. Blaser. 1970. Effects of deicing salts on water quality and biota. Literature review and recommended research. *National Cooperative Research Program Report* 91. Transportation Research Board.

Hanes, R.E., L.W. Zelazny, R.P. Verghese, R.P. Bossert, E.W. Carlson Jr., R.E. Blaser, D.D. Wolf. 1976. Effects of deicing salt on plant biota and soil. Experimental Phase. *National Cooperative Research Program Report* 170. Transportation Research Board.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomology* 20:31-39.

Hoffman, R.W., C.R. Goldman, S. Paulson, G.R. Winters. 1981. Aquatic impacts of deicing salts in the central Sierra Nevada Mountains, California. *Water Resources Bulletin* 17:280-285.

Hofstra, G., D.W. Smith. 1984. The effects of deicing salt on the levels of ions in roadside soils in southern Ontario. *Journal of Environmental Management* 19:261-271.

Hofstra, G., R. Hall. 1971. Injury to roadside trees: leaf injury on pine and white cedar in relation to foliar levels of sodium and chloride. *Canadian Journal of Botany* 49:613-622.

Hofstra, G., R. Hall, G.P. Lumis. 1979. Studies of salt-induced damage to roadside plants in Ontario. *Journal of Arboriculture* 5:25-31.

Horner, R.A. 1990. Environmental effects of calcium magnesium acetate: Emphasizing aquatic ecosystem effects. Pp. 97-119 in *The Environmental Impact of Highway Deicing* (C.R. Goldman, G.M. Malyj, eds.). Proceedings of a symposium held 13 October 1989 at University of California Davis. Insitute of Ecology Publication 133.

Howard, K.W.F., P.J. Beck. 1993. Hydrogeochemical implications of groundwater contamination by road deicing chemicals. *Journal of Contaminant Hydrology* 12: 245-268.

Hutchinson, F.E. 1970. Environmental pollution from highway deicing compounds. *Journal of Soil and Water Conservation* 144-146.

Iowa Department of Natural Resources. 2003. *Ambient Aquatic Life Criteria for Chloride*. http://www.iowadnr.com/water/standards/files/cissue.pdf.

Iowa Department of Transportation. 2001. *Production and Evaluation of Calcium Magnesium Acetate*. Publication HR-243.

Isabelle, P.S., L.J. Fooks, P.A. Keddy. 1987. Effects of roadside snowmelt on wetland vegetation: an experimental study. *Journal of Environmental Management* 25:57-60.

Isachsen, Y.W., W. Fisher. 1970. *Geologic Map of New York: Adirondack Sheet*. The University of the State of New York State Education Department. Reprinted 1995.

Jackson, R.B., E.G. Jobbagy. 2005. From icy roads to salty streams. *Proceedings of the National Academy of Science, USA* 41:14487-14488.

James, G.I., G.M. Courtin. 1985. Stand structure and growth form of the birch transition community in an industrially damaged ecosystem, Sudbury Ontario. *Canadian Journal of Forest Research* 15:809-817.

Jonca, Z., W. Lewandowski. 2004. Verification of measurement capabilities of flame atomic spectrometry for the determination of sodium, potassium, magnesium, and calcium in natural fresh water. Part II: Measurement of recoveries. *Polish Journal of Environmental Studies* 13:281-93.

Jones, M.D., D.M. Durall, S.M.Harniman, D.C. Classen, S.W. Simard. 1998. *Ectomycorrhizal Diversity of Paper Birch and Douglas Fir Seedlings Grown in Single-Species And Mixed Plots in the ICH Zone of Southern British Columbia*. Ministry of Forests Research Program. Publication EN19.

Jones, P.H., B.A. Jeffrey, P.K. Watler, H. Hutchon. 1992. Environmental impact of road salting. Pages 1-116 in *Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Judd, J.H. 1970. Lake stratification caused by runoff from street deicing. *Water Research* 4:521-532.

Judd, K.E., H.E. Adams, N.S. Bosch, J.M. Kostrzewski, C.E. Scott, B.M. Schultz, D.H. Wang, G.W. Kling. 2005. A case history: Effects of mixing regime on nutrient dynamics and community structure in the Third Sister Lake, Michigan during late winter and early spring 2003. *Lake and Reservoir Management* 21:316-329.

Kalff, J. 2002. *Limnology*. Upper Saddle River, New Jersey: Prentice Hall.

Karraker, N. 2006. Road de-icers used near wetlands: Are they worth their salt? *National Wetlands Newsletter* 28:15-19.

Kaushal, S.S., P.M. Groffman, G.E. Likens, K.T. Belt, W.P. Stack, V.R. Kelly, L.E. Band, G.T. Fisher. 2005. Increased salinization of fresh water on the northeastern United States. *Proceedings of the National Academy of Science, USA* 102:13517-13520.

Kayama, M., A.M. Quoreshi, S. Kitaoka, Y. Kitahashi, Y. Sakamoto, Y. Maruyama, M. Kitao, T. Koike. 2003. Effects of deicing salt on the vitality and health of two spruce species, *Picea abies* Karst., and *Picea glehnii* Masters planted along roadsides in northern Japan. *Environmental Pollution* 124: 127-137.

Kelsey, P.D., R.G. Hootman. 1992. Deicing salt dispersion and effects on vegetation along highways. Case study: Deicing salt deposition on the Morton Arboretum. Pages 253-281 in *Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Khasa, P.D., B. Hambling, G. Kernaghan, M. Fung, E. Ngimbi. 2002. Genetic variability in salt tolerance of selected boreal woody seedlings. *Forest Ecology and Management* 165:257-269.

Kjensmo, J. 1997. The influence of road salts on the salinity and the meromictic stability of Lake Svinsjoen, southeastern Norway. *Hydrobiologia* 347:151-158.

Kudish, M. 1992. *Adirondack Upland Flora: An Ecological Perspective.* The Chauncey Press, Saranac Lake NY.

Labadia, C.F., J.M. Buttle. 1996. Road salt accumulation in highway snow banks and transport through the unsaturated zone of the Oak Ridge Moraine, Southern Ontario. *Hydrological Processes* 10:1575-1589.

Lawrence, G.B., B. Momen, K.M. Roy. 2004. Use of stream chemistry for monitoring acid deposition effects in the Adirondack region of New York. *Journal of Environmental Quality* 33:1002-1009.

Legret, M., C. Pagotto. 1999. Evaluation of pollutant loadings in the runoff waters from a major rural highway. *Science of the Total Environment* 235:143-50.

Lewis, P.A., D.J. Klemm, W.T. Thoeny. 2001. Perspectives on the use of a multimetric lake bioassessment integrity index using benthic macroinvertebrates. *Northeastern Naturalist* 8:233-246.

Lewis, W. M. 1999. *Studies of the Environmental Effects of Magnesium Chloride Deicer in Colorado*. Colorado Department of Transportation Research Branch, CDOT-DTD-R-99-10.

Lewis, W. M. 2001. *Evaluation and Comparison of Three Chemical Deicers for Use in Colorado*. Colorado Department of Transportation Research Branch, CDOT-DTD-R-2001-17.

Leiser, A.T., S.A. John. 1990. Evaluation of the effects of calcium magnesium acetate on selected plant species. Pp. 49-96 in *The Environmental Impact of Highway Deicing* (C.R. Goldman, G.M. Malyj, eds.). Proceedings of a symposium held 13 October 1989 at University of California Davis. Insitute of Ecology Publication 133.

Liiri, M., H. Setälä, J. Haimi, T. Pennanen, H. Fritze. 2001. Relationship between soil microarthropod species diversity and plant growth does not change when the system is disturbed. *Oikos* 96:138-150.

Löfgren, S. 2001. The chemical effects of deicing salt on soil and stream water of five catchments in Southeast Sweden. *Water, Air, and Soil Pollution* 130: 863-868.

Longley, P., M. Goodchild, D. Maguire, D. Rhind. 2001. *Geographic Information Systems and Science*. Wiley Publishing.

Lui, C., J.B. Evett. 2003. *Soil Properties: Testing, Measurement, and Evaluation*. 5th ed. Prentice Hall.

Lumis, G.P., G. Hofstra, T. Hall. 1973. Sensitivity of roadside trees and shrubs to aerial drift of deicing salt. *Canadian Journal of Plants Science* 8:475-477.

McBean, E., S. Al-Nassari. 1987. Migration pattern of deicing salts from roads. *Journal of Environmental Management* 25: 231-238.

McCuen, R.H. 1989. *Hydrologic Analysis and Design*. Prentice Hall, Englewood Cliffs, NJ.

McFarland, B.L., K.T. O'Reilly. 1992. Environmental impacts and toxicological characteristics of CMA. Pages 193-228 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

McMartin, B. 1994. *The Great Forest of the Adirondacks*. North Country Books, Utica NY.

McMartin, B. 2004. *The Privately Owned Adirondacks*. Lake View Press, Canada Lake NY.

Mandeville, S.M. 2002. Benthic macroinvertebrates in freshwaters: Taxa tolerance values, metrics, and protocols. *http://chebucto.ca/Science/SWCS/SWCS.html* . Soil and Water Conservation Society of Metro Halifax, Nova Scotia Canada.

Magee, D.W., H.A. Ahles. 1999. *Flora of the Northeast: A Manual of the Vascular Flora of New England and Adjacent New York.* University of Massachusetts Press, Amherst MA.

Marquis, D.A., D.S. Solomon, J.C. Bjorkbom. 1969. *A Silvicultural Guide for Paper Birch in the Northeast*. USDA Forest Research Paper NE-130.

Mason, C.F., S.A. Norton, I.J. Fernandez, L.E. Katz. 1999. Deconstruction of the chemical effects of road salt on stream water chemistry. *Journal of Environmental Quality* 28:82-91.

Mattson, M.D., P.J. Godfrey. 1994. Identification of road salt contamination using multiple regression and GIS. *Environmental Management* 18:767-773.

Mayer, T., W.J. Snodgrass, D. Morin. 1999. Spatial characterization of the occurance of road salts and their environmental concentrations as chlorides in Canadian surface waters and benthic sediments. *Water Quality Research Journal of Canada* 34:545-574.

Meyers, J. 2002. Road salt builds up in lakes: Scientist fears rare fish at risk. *Adirondack Explorer*, November 2002 Issue.

Michigan Department of Transportation. 1993. *The Use of Selected Deicing Materials on Michigan Roads: Environmental and Economic Impacts.* http://www.michigan.gov/documents/toc-deice_51451_7.pdf

Moll, R.A., R. Rossmann, J.A. Barres, F.J. Horvath. 1992. Historical trends of chlorides in the Great Lakes. Pages 302-322 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.).

Lewis Publishers, Boca Raton, Florida.

Molles, M.C. 1980. *Effects of Road Salting on Aquatic Invertebrate Communities*. USDA Forest Service, Eisenhower Consortium Bulletin 10.

Moran, V.M., L.A, Abron, L.W. Weinberger. 1992. A comparison of conventional and alternative deicers: An environmental impact perspective. Pages 341-261 *in Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Morin, D., W. Snodgrass, J. Brown, P.A. Arp. 2000. *Impacts Evaluation of Road Salt Loads on Soils and Surface Waters*. Report submitted to the Environment Canada CEPA Priority Substances List Environmental Resource Group on Road Salt. Commercial Chemicals Evaluation Branch, Environment Canada, Hull Quebec.

National Acid Precipitation Assessment Program. 1998. *Biennial Report to Congress: An Integrated Assessment*. National Oceanic and Atmospheric Administration, Washington D.C.

New York State Department of Environmental Conservation. 1999. *High Peaks Wilderness Complex Unit Management Plan: Wilderness Management for The High Peaks of the Adirondack Park.* Office of Natural Resources – Region 5. 1-342.

New York State Department of Transportation. 2001. *Environmental Handbook for Transportation Operations.* NYSDOT Environmental Analysis Bureau.

New York State Department of Transportation. 2002. *Deicer Advisory Guidelines: The Importance of Considering the Phosphorus Content of Deicers in the Protection of Drinking Water Supplies in the New York City Watershed.* NYSDOT Watershed Inspector General.

New York State Department of Transportation. 2003. *Highway Maintenance Guidelines: Snow and Ice Control*.

Nielsen, D.L., M.C. Brock, K. Crossle, K. Harris, M. Healey, I. Jarosinski. 2003. The effects of salinity on aquatic plant germination and zooplankton hatching in two wetland sediments. *Freshwater Biology* 48:2214-2223.

Norrström, A.C., G. Jacks. 1998. Concentration and fractionation of heavy metals in roadside soils receiving de-icing salts. *Science of the Total Environment* 218: 161-174.

Norrström, A.C., E. Bergstedt. 2001. The impact of road deicing salts (NaCl) on colloid dispersion and base cation pools in roadside soils. *Water, Air, and Soil Pollution* 127: 281-299.

Novotny, V. 1998. Cyanide and metal pollution by urban snowmelt: Impact of deicing compounds. *Water Science and Technology* 38:223-230.

Oberts, G.L. 1986. Pollutants associated with sand and salt applied to roads in Minnesota. *Water Resources Bulletin* 22:479-484.

Oberts, G.L. 1991. Influence if snowmelt dynamics on storwater runoff quality. *Watershed Protection Techniques* 1:55-61.

Ormsby, W.C. 1999. New technologies improve cost-effectiveness of CMA. *Public Roads* 63:3, http://www.tfhrc.gov/pubrds/novdec99/cmaupdate.htm.

Paludan-Müller, G., H. Saxe, L.B. Pedersen, L.B., T.B. Randrup. 2002. Differences in salt sensitivity of four deciduous tree species to soil or airborne salt. *Physiologia Plantarum* 114: 223-230.

Paschka, M.G., R.S. Ghosh, D.A. Dzombak. 1999. Potential water-quality effects from iron cyanide anticaking agents in road salt. *Water Environment Research* 71:1235-1239.

Patterson III, W.A., J.J. Olson. 1983. Effects of heavy metals on radicle growth of selected woody plant species germinated on filter paper, mineral soil, and organic soil substrates. *Canadian Journal of Forest Research* 13:233-238.

Pederson, L.B., T.B. Randrup, M. Ingerslev. 2000. Effects of road distance and protective measures on deicing NaCl deposition and soil solution chemistry in planted median strips. *Journal of Aboriculture* 26: 238-244.

Pollock, S.J. 1992. Remediating highway deicing salt contamination of public and private water supplies in Massachussetts. Pages 519-538 in *Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Ramakrishna, D.M., T. Viraraghavan. 2005. Environmental impact of chemical deicers – A review. *Water, Air, and Soil Pollution* 166:49-63.

Ramstack, J.M., S.C. Fritz, D.R. Engtrom. 2004. Twentieth century water quality trends in Minnesota lakes compared with presettlement variability. *Canadian Journal of Fisheries and Aquatic Sciences* 61:561-576.

Raymond, L.A. 2002. *Petrology: The Study of Igneous, Sedimentary, and Metamorphic Rocks*. 2nd ed. McGraw-Hill: New York.

Renault, S., C. Lait, J.J. Zwiazek, M. MacKinnon. 1998. Effect of high salinity tailings waters produced from gypsum treatment of oil sands tailings on plants of the boreal forest. *Environmental Pollution* 102:177-184.

Renault, S., C. Croser, J.A. Franklin, J.J. Zwiazek, M. MacKinnon. 2001. Effects of consolidated tailings water on red-osier dogwood (*Cornus stolonifera* Michx) seedlings. *Environmental Pollution* 113:27-33.

Renault, S., C. Qualizza, M. MacKinnon. 2004. Suitability of altai wildrye (*Elymus angustus)* and slender wheatgrass (*Agropyron trachycaulum*) for initial reclamation of saline composite tailings of oil sands. *Environmental Pollution* 128:339-349.

Rhodes, A.L., R.M. Newton, A. Pufall. 2001. Influences of land use on water quality of a diverse New England watershed. *Environmental Science and Technology* 35: 3640-3645.

Richburg, J.A., W.A. Patterson, F. Lowenstein. 2001. Effects of road salt and *Phragmites australis* invasion on the vegetation of a western Massachusetts Calcareous Lake-Basin Fen. *Wetlands* 21: 247-255.

Rimmer, A., Y. Aota, M. Kumagai, W. Eckert. 2005. Chemical stratification in thermally stratified lakes: a chloride mass balance model. *Limnology and Oceanography* 50:147-157.

Rosenberg, D.M., V.H. Resh. 1993. *Freshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hall, New York.

Rosenberry, D.O., P.A. Bukaveckas, D.C. Buso, G.E. Likens, A.M. Shapiro, T.C. Winter. 1999. Movement of road salt to a small New Hampshire lake. *Water, Air, and Soil Pollution* 109:179-206.

Route 73 Steering Committee. 1999. *New York State Route 73 Scenic Corridor Management Plan*. Adirondack Park Agency, Raybrook NY.

Ryan, P.A., T.R. Marshall. 1994. A niche definition for lake trout (*Salvelinus namaycush)* and its use to identify populations at risk. *Canadian Journal of Fisheries and Aquatic Science* 51:2513-2519.

Sanzo, D., S.J. Hecnar. 2005. Effects of deicing salt (NaCl) on larval wood frogs (*Rana sylvatica*). *Environmental Pollution* in press.

Salt Institute. 1994. *Deicing Salt Facts: A Quick Reference*. Salt Institute. Alexandria, Virginia.

Salt Institute. 2004. *Highway Salt and Our Environment*. Salt Institute. Alexandria, Virginia.

Scott, W.B., E.J. Crossman. 1973. Freshwater fishes of Canada. *Fisheries Research Board of Canada Bulletin* 184:1-966.

Scott, W.S. 1980. Deicing salt levels in Toronto stream banks and roadside soils. *Bulletin of*

Environmental Contamination and Toxicology 25:208-214.

Scott, W.S. 1981. An analysis of factors influencing deicing salt levels in streams. *Journal of Environmental Management* 13:269-287.

Semlitsch, R.D. 2003. *Amphibian Conservation*. Smithsonian Institution Press. Washington D.C.

Setälä, H., V. Huhta. 1991. Soil fauna increase *Betula pendula* growth: Laboratory experiments with coniferous forest floor. *Ecology* 72:665-671.

Shanley, J.B. 1994. Effects of ion exchange on stream solute fluxes in a basin receiving highway deicing salts. *Journal of Environmental Quality* 23:974-986.

Shi, X., J.M. Staples, O. Stein. 2004. Highway runoff BMPs and winter traction materials management: A cold regions perspective. In review, see *www.coe.montana.edu/wti/wwwshare/Corrosion/Winter_Traction_Materials.pdf*

Siver, P.A., R.W. Canavan, C.K. Field, L.J. Marsicano, A.M. Lott. 1996. Historical changes in Connecticut Lakes over a 55-year period. *Journal of Environmental Quality* 25:334-345.

Smith, C.L. 1985. *The Inland Fishes of New York State*. New York State Department of Environmental Conservation. Albany, New York.

Smol, J.P., S.R. Brown, R.N. McNeely. 1983. Cultural disturbances and trophic history of a small meromictic lake from central Canada. *Hydrobiologia* 103:125-30.

Staples, J.M., L. Gamradt, O. Stein, X. Shi. 2004. *Recommendations for Winter Traction Materials Management on Roadways Adjacent to Bodies of Water*. Montana Department of Transportation Research Section, FHWA/MT-04-008/8117-19.

Steedman, R.J., C.J. Allan, R.L. France, R.S. Kushneriuk. 2004. Land, water, and human activity on boreal watersheds. Pages 59-85 in *Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment* (J.M. Gunn, R.J. Steedman, R.A. Ryder, eds.). CRC Press, Boca Raton, Florida.

Sutherland, J.W., S.A. Kisbaugh, J.A. Bloomfield, W.T. Lavery, F.E. VanAlstyne. 1990. Water resources and water quality in the Adirondack Park. Pages 594-701 in *The Adirondack Park in the 21st Century: Technical Reports Volume 1*, Commission on the Adirondacks in the 21st Century (eds.). Published by New York State.

Szabolcs, I. 1994. Soils and salinization. Pages 3-11 in: *Handbook of Plant and Crop Stress* (ed. M. Pessarakali). Marcel Decker, NY.

Thompson E.H., E.R. Sorenson. 2000. *Wetland, Woodland, Wildland: A Guide to the Natural*

Communities of Vermont. University of New England Press.

Thorp, J.H., A.P. Covich. 1991. *Ecology And Classification of North American Freshwater Invertebrates*. Academic Press, San Diego CA.

Thunqvist, E.L. 2003. Increased chloride concentration in a lake due to deicing salt application. *Water Science and Technology* 48:51-59.

Thunqvist, E.L. 2004. Regional increase of mean chloride concentration in water due to the application of deicing salt. *Science of the Total Environment* 325:29-37.

Trombulak, S.C., C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14:18-30.

Transportation Research Board. 1991. *Special Report 235: Highway deicing – comparing salt and calcium magnesium acetate*. National Research Council Washington, D.C.

Turtle, S.L. 2001. Embryonic survivorship of the spotted salamander (*Ambystoma maculatum*) in roadside and woodland vernal pools in southeastern New Hampshire. *Journal of Herpetology* 34:60-67.

United States Environmental Protection Agency. 1988. *Ambient Aquatic Life. Water Quality Criteria for Chloride*. Office of Research and Development, Environmental Research Laboratory, Duluth, Minnesota. EPA 440/5-88-001.

United States Environmental Protection Agency. 1986. *Ambient Water Quality Criteria for Dissolved Oxygen*. Office of Research and Development, Environmental Research Laboratories, Duluth, Minnesota. EPA 440/5-86-003.

United States Environmental Protection Agency. 1999. *Storm Water Management Fact Sheet – Minimizing Effects From Highway Deicing*. Office of Water, Publication 832-F-99-016. Washington, D.C.

United States Environmental Protection Agency. 2002. *Source Water Protection Practices Bulletin – Managing Highway Deicing To Prevent Contamination Of Drinking Water*. Office of Water (4606). Publication 816-F-02-019. Washington, D.C.

VanDiver, B.B. 1976. *Rocks and Routes of the North Country New York*. W.F. Humphrey Press, Geneva, New York.

VanDiver, B.B. 1985. *Roadside Geology of New York*. Mountain Press Publishing, Missoula, Montana.

Viskari, E., L. Karenlampi. 2000. Roadside Scots pine as an indicator of deicing salt use – a comparative study from two consecutive winters. *Water, Air, and Soil Pollution* 122: 405-419.

Vitaliano, D.F. 1992. Environmental costs of road salting. Pages 441-459 in *Chemical Deicers and the Environment* (F.M. D'Itri, ed.). Lewis Publishers, Boca Raton, Florida.

Ylioja, T., M. Rousi. 2001. Soil fertility alters susceptibility of young clonal plantlets of birch (*Betula pendula*) to a dipteran stem miner. *Ecoscience* 8: 191-198.

Wagner, G.H., K.F. Steele. 1989. Na^+/Cl^- ratios in rain across the USA, 1982-1986. *Tellus* 41B:444-451.

Wang, J.S., P.M. Huang, W.K. Liaw, U.T. Hammer. 1991. Kinetics of the desorption of mercury from selected freshwater sediments as influenced by chloride. *Water, Air, and Soil Pollution* 56:533-542.

Weldy, T., D. Werier. 2005. *New York Flora Atlas*. http://atlas.nyflora.org. New York Flora Association, Albany NY.

Wetzel, R.G., G.E. Likens. 2000. *Limnological Analyses* (3rd ed.). New York, New York: Springer.

Wilcox, D.A. 1984. The effects of NaCl deicing salts *Spagnum recurvum* P. Beauv. *Environmental and Experimental Botany* 24:295-304.

Wilcox, D.A. 1986. The effects of deicing salts on vegetation in Pinhook Bog, Indiana. *Canadian Journal of Botany* 64:865-874.

Wilcox, D.A., R.E. Andrus. 1987. The role of *Spagnum fimbriatum* in secondary succession in a road impacted bog. *Canadian Journal of Botany* 65:2270-2275.

Williams, D.D., N.E. Williams, Y. Cao. 1997. Spatial differences in macroinvertebrate community structure in springs in southeastern Ontario in relation to their chemical and physical environments. *Canadian Journal of Zoology* 75:1404-1414.

Williams, D.D., N.E. Williams, Y. Cao. 1999. Road salt contamination of groundwater in a major metropolitan area and development of a biological index to monitor its impact. *Water Resources* 34:127-38.