Subject: Written Testimony for the 2024 NYS Joint Legislative Hearing
January 24, 2024

Honorable Members of the Transportation Committee,

On behalf of Paul Smith's College, I am writing to submit my testimony for the 2024 New York State Joint Legislative Hearing on Transportation. I appreciate the opportunity to contribute insights and recommendations that can shape the policies and initiatives important to the well-being and progress of our state. With a focus on supporting safe and efficient transportation systems and a clean and healthy environment, our testimony focuses on protecting clean water and the health and well-being of our communities. Thank you for considering our perspective, and I look forward to contributing to the collaborative work to support New York's transportation practices.

The Paul Smith's College Adirondack Watershed Institute (AWI) has been a leader in protecting clean water for several decades. One of our most prominent research initiatives focuses on safeguarding waterways and protecting private drinking wells from the adverse effects of road salt contamination. Through years of scientific research, AWI has provided important contributions into the impact of road salt application on water quality in the Adirondack Park. AWI's studies have revealed that road salting, particularly in adherence to New York State Department of Transportation deicing protocols, significantly elevates sodium and chloride concentrations in lakes, streams and private drinking wells. Our research was foundational to the recommendations included in the *Adirondack Road Salt Reduction Task Force Report.* The research underpinning the recommendations in the Report have far-reaching implications for the protection of drinking wells and aquatic ecosystems, emphasizing the need for sustainable winter road maintenance practices to mitigate the harmful human health and environmental consequences of salt runoff.

AWI's research cited below holds significant implications for the NYS Transportation Committee, emphasizing the urgent need to look closely at road salt application on state roads. We are requesting that the committee take into account the environmental consequences outlined in the studies and consider the following recommendations:

- Amend the Transportation Law to add a new Article to Chapter 61A of the Consolidated Laws of New York establishing the NY Road Salt Reduction Council and the NY Road Salt Reduction Advisory Committee.
- Support NYS Department of Environmental Conservation's adoption of a water quality standard for chloride for the protection of aquatic life following US EPA's recommended chronic (230 mg/L) and acute (860 mg/L) criteria.
- Support funding for long-term monitoring of Adirondack waters to track progress of road salt reduction efforts implemented by the state.

A 2012 study published in Water Research by AWI scientists Kelting, Laxson, and Yerger provides information on the impact of paved roads on sodium and chloride concentrations in lakes within the Adirondack Park. The authors analyzed 138 lakes to establish baseline concentrations for sodium and chloride in watersheds without paved roads. Their findings revealed that lakes in the least impacted watersheds, devoid of paved roads, exhibited very low concentrations of sodium and chloride, with the majority registering concentrations below 1 mg/L for both chemicals. However, in watersheds with paved roads, particularly those maintained in winter according to New York State Department of Transportation deicing protocols, there was a significant increase in sodium and chloride concentrations, attributing road salting on state highways as the primary cause of elevated levels in lakes within the

Adirondack Park. The study also introduced a model correlating state roads' proximity to the shoreline with lake sodium and chloride concentrations, offering a valuable tool for identifying areas that require different treatment strategies to mitigate environmental impacts. (add link)

A 2015 study published in Environmental Monitoring and Assessment by AWI scientists Regalado and Kelting estimated the portion of lands and waters impact by road runoff in the Adirondack Park. The authors used hydrologic modeling to delineate surface flow downslope of paved roads, demonstrating the potential movement of pollutants, such as road salt, through the environment. Their findings revealed that 11% of the land area, 77% of the lake surface area, and 52% of the stream length in the Adirondack Park may be impacted by road runoff, with much of this coming from state and federal highways. Notably, 70% of the surface water in areas designated as Wilderness receive road runoff. This work documents the potential magnitude of the impact from road runoff in the region and that Forest Preserve lands are also at risk. Finally, the model results provide a useful tool for identifying areas to implement best management practices, design monitoring studies, and guide state investments.

A 2019 study published in Lake and Reservoir Management by AWI scientists Wiltse, Yerger, and Laxson documented a reduction in spring mixing in Mirror Lake (Lake Placid, NY). The study showed that chloride was accumulating at the lake bottom and reducing the natural mixing processes in the lake. The reduction in mixing was linked to decreased dissolved oxygen concentrations and a reduction in habitat availability for lake trout. Mirror Lake is one of small number of lakes in the country to have impacts on the physical mixing properties as a result of road salt documented in the scientific literature.

A 2019 study that is currently in preparation for publication and a key part of the Road Salt Task Force Report by AWI scientists, Kelting, Yerger, and Laxson documented widespread groundwater contamination by road salt by analyzing data from 529 private drinking water wells across the Adirondack Park. The study used the runoff model discussed above to classify wells based on the type of road runoff they receive (State, Local, or None). The results show that median chloride concentrations for wells downslope of state roads is 12-times higher than wells downslope of local roads and 83-times higher than wells that receive no road runoff. The study also documented that 63% of wells downslope of state road exceed EPA guidance criteria for sodium, while 20% of wells downlope of local roads exceed this criteria, and wells that do not receive road runoff do not exceed this criteria. Finally, over 60% of wells downslope of state roads have highly corrosive water, indicating the potential for leaching toxic heavy metals such as lead. This is in comparison to 20% of wells downslope of local roads having highly corrosive water and 3% of wells that receive no road runoff. This study was critical to the task force report and demonstrates the need to address road salt for the protection of human health.

A 2023 report submitted to the Lake Champlain Basin Program by AWI scientist Brendan Wiltse quantified the de-icing salt pollution load to Mirror Lake and the Chubb River in Lake Placid, NY. This study estimated that 15-16% of the de-icing salt originates from local roads, 33-35% from state roads, 3-4% from sidewalks, and 42-49% is unaccounted. The unaccounted salt likely originates from commercial and private application to sidewalks, driveways, and parking lots. This underscores the importance of addressing all sources of salt pollution, especially in urban environments. Additionally, the report documents a reduction in salt retention in the lake in response to the implementation of best management practices, demonstrating that these practices do result in improvements in the environment.

Thank you for your consideration,

ZSmith

Zoë Smith, Executive Director, Adirondack Watershed Institute

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Regional analysis of the effect of paved roads on sodium and chloride in lakes

*Daniel L. Kelting**, *Corey L. Laxson*, *Elizabeth C. Yerger*

Adirondack Watershed Institute, Paul Smith's College, Routes 86 & 30, Post Office Box 265, Paul Smiths, NY 12970, United States

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ABSTRACT

Salinization of surface water from sodium chloride (road salt) applied to paved roads is a widely recognized environmental concern in the northern hemisphere, yet practical information to improve winter road management to reduce the environmental impacts of this deicer is lacking. The purpose of our study was to provide such information by developing baseline concentrations for sodium and chloride for lakes in watersheds without paved roads, and then determining the relationship between these ions and density, type, and proximity of paved roads to shoreline. We used average summer (June–September) sodium and chloride data for 138 lakes combined in a watershed based analysis of paved road networks in the Adirondack Park of New York, U.S.A. The watersheds used in our study represented a broad range in paved road density and type, 56 of which had no paved roads. Median lake sodium and chloride concentrations in these 56 watersheds averaged 0.55 and 0.24 mg/L, respectively. In contrast, the median sodium and chloride concentrations for the 82 lakes in watersheds with paved roads were 3.60 and 7.22 mg/L, respectively. Paved road density (lane-km/km^2) was positively correlated with sodium and chloride concentrations, but only state roads were significantly correlated with sodium and chloride while local roads were not. State road density alone explained 84 percent of the variation in both ions. We also successfully modeled the relationship between road proximity to shoreline and sodium and chloride concentrations in lakes, which allowed us to identify sections of road that contributed more to explaining the variation in sodium and chloride in lakes. This model and our approach could be used as part of larger efforts to identify environmentally sensitive areas where alternative winter road management treatments should be applied.

1. Introduction

Sodium chloride (NaCl, commonly referred to as "road salt") is widely used in the northern hemisphere to maintain clear roads in the winter months (Albright, 2005; Löfgren, 2001; Rodrigues et al., 2010). Over 19 million metric tons of sodium chloride is applied to provincial and state highways annually in North America alone (NCHRP, 2007). Though alternative deicing treatments are available, due to the comparatively low cost and high availability of sodium chloride, it continues to be the most commonly applied chemical used to maintain clear roads and its use on US highways has increased steadily over the last 50 years (Jackson and Jobbagy, 2005).

A number of studies have reported increased sodium, chloride, and conductivity in surface and groundwater near salted roads, with these increases being attributed to road

* *Corresponding author*. Tel.: +1 011 1 518 3276213; fax: +1 011 1 518 3276350.
E-mail addresses: dkelting@paulsmiths.edu (D.L. Kelting), claxson@paulsmiths.edu (C.L. Laxson), eyerger@paulsmiths.edu (E.C. Yerger).

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density, impervious surfaces, and road salt application rates (Kaushal et al., 2005; Löfgren, 2001; Mullaney et al., 2009; Rosfjord et al., 2007; Siver et al., 1996). Sodium chloride has variable impacts on aquatic environments which makes it difficult to generalize effects and establish water quality criteria for aquatic biota (Corsi et al., 2010). This said, recent authors have stated that the steady increase in sodium and chloride concentrations observed in surface waters should be a cause for concern and efforts should be made now to reduce the amount of salt applied to roads (Jackson and Jobbagy, 2005; Kaushal et al., 2005). This is particularly important when considering recently published evidence for sodium and chloride retention in watersheds (Kelly et al., 2008; Kincaid and Findlay, 2009) which suggests that concentrations of these ions in surface water may remain high even if application rates are reduced.

The New York State Department of Transportation (NYSDOT) is the largest user of sodium chloride for winter road management in North America (NCHRP, 2007). The NYSDOT applies 680,000–860,000 metric tons of sodium chloride to state roads every winter with application rates ranging from 10 to 14 metric tons per lane-km per year (Albright, 2005; Kelting and Laxson, 2010). The northern region of New York State is dominated by the 2.4 million hectare Adirondack Park (AP), a mosaic of public and privately owned lands created in 1892 to protect water and timber resources (NYSAPA, 2011). The AP contains several thousand lakes and ponds and over 48,000 km of rivers and streams. The AP is intersected by 4530 lane-km of NYSDOT roads that are consistently treated with sodium chloride and 12,380 lane-km of roads maintained by local municipalities that receive a variety of winter treatments (snow plow only, sand, sand plus salt, and salt).

The AP has comparatively low paved road densities and sodium and chloride concentrations in Adirondack lakes are on average much lower than in other northeastern lakes (Rosfjord et al., 2007). However, given that the AP was created in part to protect water resources and the fact that salinization will steadily increase if current practices continue, we conducted a study with the overall goal of providing science-based information to aid in improving winter road management practices to reduce the loading of sodium and chloride to lakes in the Adirondack Park. Though our study focused on the AP, we believe that our research approach is applicable to other lake regions in the northern hemisphere. The specific objectives of our study were to (1) provide baseline estimates of sodium and chloride in lakes in watersheds with and without paved roads, (2) determine the effects of paved road type and density on sodium and chloride concentrations in lakes, and (3) determine the effect of paved road proximity to shoreline on sodium and chloride concentrations in lakes.

2. Materials and methods

The bedrock geology in the central part of the AP is dominated by metaplutonic rocks such as granite gneiss, metanorthosite, and metagabbro with metasedimentary rocks forming the bedrock on the periphery of the AP (Isachsen et al., 2000). This bedrock is overlain by glacial deposits of till and outwash that form the parent materials for the coarse textured soils that dominate the region (Isachsen et al., 2000). These soils support a largely forested region that receives about 100 cm of precipitation per year (Sullivan et al., 2006). The forests cover over 95% of the land area and consist of hardwood, conifer, and mixed forests. The land area of the AP is 43% public lands that are protected and cannot be developed or harvested and 57% private lands that are managed largely for timber (NYSAPA, 2011). The AP has a very low population density with only about 130,000 year-round residents living on its 2.4 million hectares of land.

We utilized Geographic Information Systems (GIS) and water quality data to investigate the effects of paved roads on sodium and chloride concentrations in 138 lakes throughout the Adirondack Park (Fig. 1). Lake area ranged from 0.01 to 83.94 km^2 with a median area of 0.66 km^2. Watershed characteristics for our study region are summarized in Table 1. The watershed areas for each individual lake were digitized on top of a USGS 1:100 k topographical map using ArcGIS software (ESRI, Redlands, CA). Land cover, road network, and geologic features were clipped to each of the watershed boundaries. The percentage of each land cover class and surface geology type was calculated by dividing the total area of the cover class by the total watershed area. Road densities (lane-km/km^2) were calculated on a lane-km basis by multiplying the total length of each road type by the number of lanes (state = 2, county = 1.5, town = 1.5, local = 1.5, interstate = 4, US highway = 2) and dividing this value by the watershed areas. All GIS layers were obtained from the NYS Adirondack Park Agency GIS database (NYSAPA, 2011).

2.1. *Lake chemistry data sources*

The sodium and chloride dataset for the 138 lakes was compiled from two sources. Data for 84 of the lakes were from the Adirondack Watershed Institute (AWI) lake monitoring program that represents a large range in paved road density (some also without roads) and type. One surface water sample was collected from each of these lakes once a month by boat (May–October, 2010). The samples were taken at the deepest part of each lake by deploying a Kemmerer water bottle to 1.5 m depth. Samples were analyzed for sodium by inductively coupled plasma optical emission spectroscopy (Varian Instruments, 720-ES, Walnut Creek, CA) and chloride by suppressed ion chromatography (Lachat Instruments, QC8500, Loveland, CO). Data for the 54 remaining lakes were obtained from the Adirondack Lake Survey Corporation (ALSC), whose long term dataset includes analysis of sodium and chloride. ALSC data were collected from the surface water of lakes in least impacted watersheds (though some with roads) on a monthly basis and analyzed for sodium by atomic absorption spectroscopy and chloride by ion chromatography. For this study, the dataset was compiled and analyzed as average summer concentration of sodium and chloride (June–September) for each lake using 2010 data from the AWI and 2009 data from the ALSC (the 2010 ALSC dataset was not yet available at the time of this analysis). In addition, the complete time series of ALSC average summer concentration of sodium and chloride from 1992 through 2009 was used to examine the variation in these ions through time: this analysis was done in part to check the validity of comparing

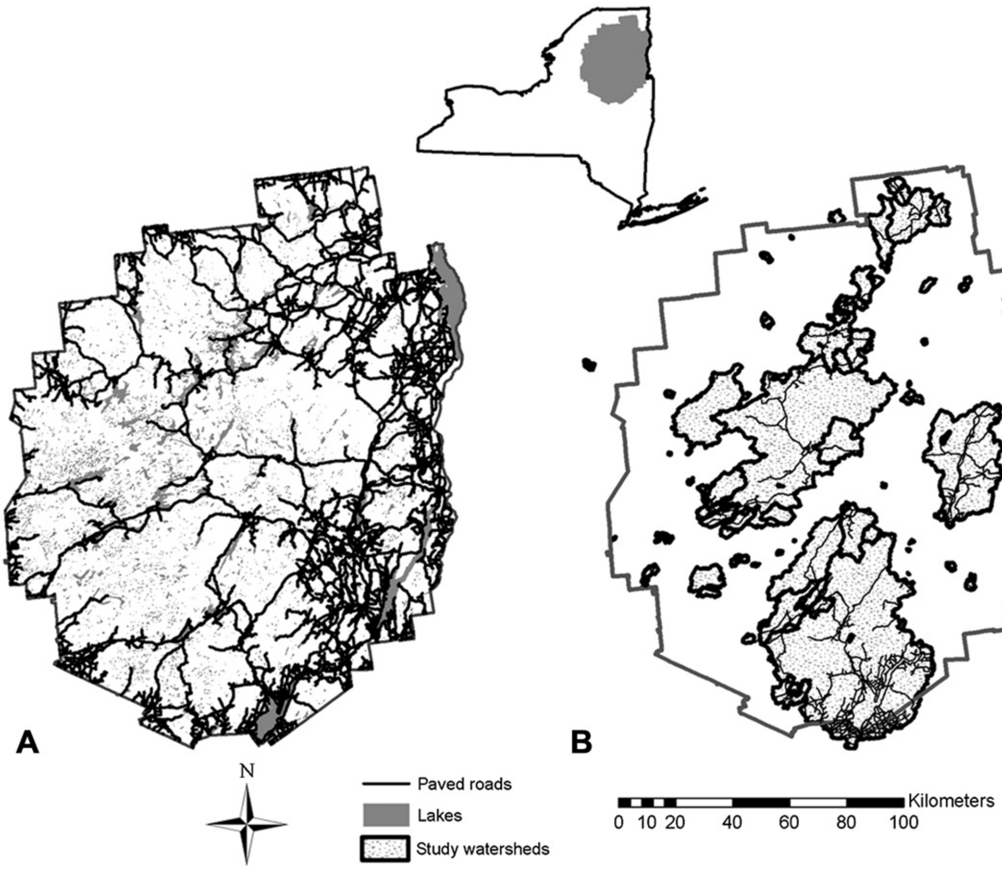

Fig. 1 – Paved road network and lakes (A) and locations of 138 study watersheds with associated paved road networks (B) in the Adirondack Park of New York, U.S.A.

datasets collected from different years but also to understand the historical trends in sodium and chloride concentrations under least impacted watershed conditions.

2.2. *Effect of road presence*

To examine the effect of presence of paved roads on sodium and chloride concentrations in our study lakes the data were split into two groups, watersheds with paved roads ($n = 82$) and those without paved roads ($n = 56$). General watershed characteristic in these two groups were similar, only differing in terms of presence of paved roads, watershed size, and area of surface water, though the ratios of watershed area to surface water area were very similar (Table 1). The sodium and chloride datasets did not follow the normal distribution, so differences in sodium and chloride concentrations between these two groups were tested for using the two-sample Wilcoxon rank sum test. In addition, cumulative frequency distributions were constructed to examine the distribution patterns between the two groups.

2.3. *Effect of road density and type*

The effects of road density and type on sodium and chloride concentrations were examined using only the 82 lakes in watersheds with paved roads. Simple linear regression was used to examine the effect of total road density on sodium and chloride concentrations and multilinear regression was used to examine the influence of road type for the six types of paved roads that exist within the Adirondack Park: county roads, interstate highways, local roads, state roads, town roads, and US routes. As multicollinearity is possible in multiple regression, co-correlations among road types were first evaluated using simple linear correlations, for which no statistically significant correlations among the road types were found (maximum $r = 0.3$ and all p-values were greater than 0.2).

2.4. *Influence of road proximity to shoreline*

We used GIS to capture the length of paved roads that existed in a range of proximities to our 82 study lakes. Our approach was to add a series of buffers of increasing width to each of the shorelines in the watersheds. Buffer widths of 10, 20, 40, 80, 160, 320, 640, and 1280 m were added to each water body, and then clipped to the watersheds to prevent inter-watershed overlap. The road network shapefile was clipped to each of the eight buffer widths, and then road density was calculated for each buffer width. The relationship between road density in each of the buffer widths and sodium and chloride concentration in the lakes was analyzed using simple linear regression. A nonlinear equation was fit to the regression results to model the relationship between the percent

Table 1 – General characteristics for watersheds without paved roads ($n = 56$) and with paved roads ($n = 82$) used in an analysis of the effects of paved road type and density on sodium and chloride concentrations in lakes located in the Adirondack Park of New York, U.S.A.

Characteristic	1st quartile		Median		3rd quartile	
	No roads	Roads	No roads	Roads	No roads	Roads
Watershed area (km^2)	1.2	5.3	2.6	17.1	6.7	96.1
Surface water (km^2)	0.1	0.9	0.2	2.0	0.6	8.9
WA/SW ratio[a]	6.0	5.6	10.5	8.8	16.1	14.1
Land cover						
Developed (%)	0	0	0	0	0	1
Deciduous (%)	51	63	69	73	81	79
Conifer (%)	4	6	8	11	19	18
Mixed (%)	5	7	10	9	16	13
Forested (%)	89	91	97	94	99	98
Agriculture (%)	0	0	0	0	0	1
Wetlands (%)	0	0	1	4	8	7
Surface geology						
Sand and gravel (%)	0	0	0	13	11	31
Glacial till (%)	57	48	80	68	94	78
Bedrock (%)	0	1	6	8	19	13

a WA/SA ratio = watershed area divided by surface water area.

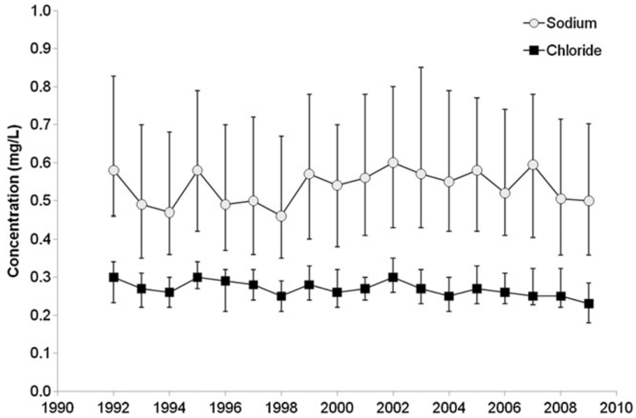

Fig. 2 – Yearly median summer (June–September) sodium (open circles) and chloride (solid squares) concentrations (mg/L) for lakes in the ALSC program from 1992 through 2009. Vertical bars represent first and third quartiles ($n = 54$ each year).

variation in sodium and chloride explained by road density and the corresponding buffer width from the lake shoreline.

3. Results

3.1. *Historical sodium and chloride concentrations*

Analysis of eighteen years of lake monitoring data from the ALSC program dataset showed that median summer sodium and chloride concentrations have remained relatively stable over time although a slight downward trend was observed for chloride (Fig. 2). The maximum year to year difference in median sodium and chloride concentrations were 0.1 and 0.05 mg/L, respectively. Over the entire sampling period median sodium was within a range of 0.35–0.85 mg/L and median chloride was within a range of 0.20–0.35 mg/L. The median sodium concentration across all years was 0.54 mg/L and the median chloride concentration across all years was 0.27 mg/L.

3.2. *Effect of road presence*

With median concentrations of 0.55 mg/L for sodium and 0.24 mg/L for chloride in lakes in watersheds without paved roads, these values were nearly identical to their respective historical median concentrations in the ALSC dataset. The median sodium concentration in lakes in watersheds without paved roads was 5.5 times lower than the median sodium concentration of 3.60 mg/L measured in lakes in watersheds with paved roads ($p < 0.001$). The median chloride concentration in lakes in watersheds without paved roads was 29 times lower than the median chloride concentration of 7.22 mg/L measured in lakes in watersheds with paved roads ($p < 0.001$).

Sodium concentrations in lakes in watersheds without paved roads ranged from 0.1 to 3.7 mg/L with 98% of lakes having concentrations less than 1.5 mg/L (Fig. 3A). Sodium concentrations in lakes in watersheds with paved roads ranged from 0.1 to 32.8 mg/L with 70% of lakes having concentrations greater than 1.5 mg/L. Chloride concentrations in lakes in watersheds without paved roads ranged from 0.1 to 5.3 mg/L with 90% of lakes having concentrations less than 2.5 mg/L (Fig. 3B). Chloride concentrations in lakes in watersheds with paved roads ranged from 0.1 to 58.4 mg/L with 80% of lakes having concentrations greater than 2.5 mg/L.

The molar equivalent concentration of sodium to chloride in lakes in watersheds without paved roads showed a weak positive correlation (slope = 2.5, slope $p = 0.001$, $r^2 = 0.26$), while the molar equivalent concentration of sodium to chloride in lakes in watersheds with paved roads showed a strong positive correlation and a significantly lower slope coefficient compared to without paved roads (slope = 0.85, slope $p < 0.001$, $r^2 = 0.96$). The 95% confidence interval on the slope coefficient for paved roads was 0.82–0.89.

3.3. *Effect of road density and type*

Sodium and chloride concentrations were both positively correlated with total paved road density in the watersheds, with road density explaining 22% of the variation in sodium and 26% of the variation in chloride (Fig. 4). Though both linear regressions were statistically significant ($p < 0.001$ for all model coefficients), the relationships were weak and there was a lot of residual error around the fitted lines, particularly at higher road density.

When paved road density was analyzed by road type in a multilinear regression model, the variation in both sodium and chloride explained by paved road density increased substantially, with road density by type explaining 85% of the

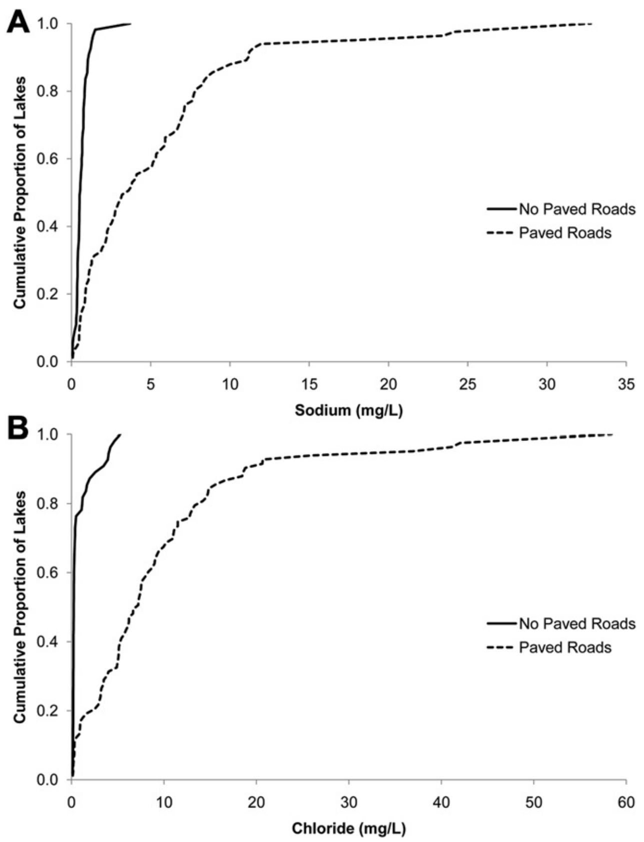

Fig. 3 – Cumulative proportions of sodium (A) and chloride (B) concentrations in lakes in watersheds with no paved roads (solid lines) and lakes in watersheds with paved roads (dashed lines) for 138 watersheds located in the Adirondack Park of New York, U.S.A.

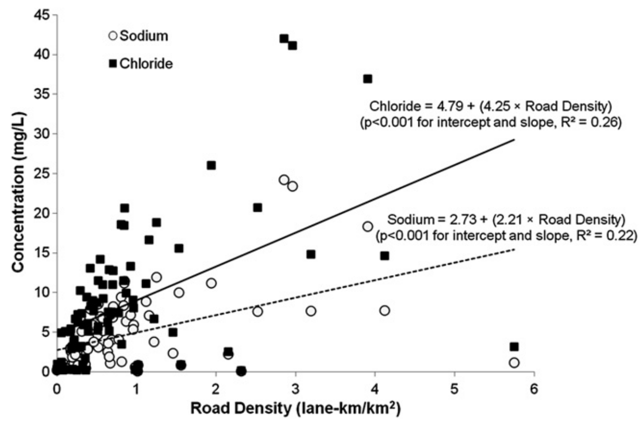

Fig. 4 – Relationship between sodium (open circles) and chloride (solid squares) concentrations (mg/L) in lakes and total paved road density (lane-km/km^2) for 82 watersheds located in the Adirondack Park of New York, U.S.A. The dashed and solid lines represent simple linear regression fits through the sodium and chloride data, respectively.

variation in sodium and 87% of the variation in chloride (Table 2). In the sodium model interstate, state, and US routes were significant variables ($p < 0.05$) in the regression model, with these three road types explaining a combined total of 82% of the variation in sodium. In the chloride model local roads and interstate, state, and US routes were significant variables in the regression model, though the standard error for the slope coefficient of local roads was very high and local roads explained less than 1% of the variation in chloride. Interstate, state, and US routes explained a combined total of 82% of the variation in chloride. Thus, for both sodium and chloride local roads (county, local, and town) were not significant contributors to explaining the concentration of these ions in lakes, while state roads (interstate state, and US routes) explained almost all of the variation for both ions.

When the relationship between sodium and chloride concentrations in lakes and road density was reanalyzed using only paved state road density (=interstate + state + US routes), state road density explained 84% of the variation in both sodium and chloride (Fig. 5). The strength of this relationship was improved over that shown for total road density, with clear positive correlations shown between concentration and state road density for both ions. The slope coefficients show that the chloride concentration increases at a higher rate than sodium for each lane-km per km^2 of state roads in the watershed. The ratio of the sodium to chloride slopes is 0.56, which is equivalent to a molar ratio of 0.86, the same molar ratio determined for the correlation between sodium and chloride for lakes in watersheds with paved roads.

3.4. *Influence of road proximity to shoreline*

Road density in all lakeshore buffer widths contributed to explaining some portion of the variation in sodium and chloride, with the percent variation explained by road density increasing with buffer width in a nonlinear manner (Table 3). Road density in the 10 m buffer explained 36 and 30% of the variation in sodium and chloride, respectively. While at 320 m, 78 and 76% of the variation in sodium and chloride was explained by road density in this buffer. Roads closer to the shoreline contribute disproportionally more to explaining the variation in sodium and chloride than roads further away. For example, 22% of the total road length lies within the 160 m buffer, yet these roads explain 64% of the variation in both ions, while doubling the amount of road length in the 320 m buffer only explains an additional 14 and 12% of the variation in sodium and chloride, respectively. All of the variation in sodium explainable by road density (84%) is captured within the 640 m buffer, which constitutes 66% of the total road length in the study watersheds. We were able to model the variation in sodium and chloride as a function of road proximity to shoreline using a nonlinear asymptotic function, with buffer width explaining 87% of the variation (Fig. 6).

4. Discussion

The lake sodium and chloride concentrations in the undeveloped watersheds in our study represent reference conditions for the Adirondack Park and these values agree closely with data reported for lakes in other undeveloped watersheds

Table 2 – Summary of regression and analysis of variance results of a multilinear regression analysis of sodium and chloride concentrations (mg/L) in lakes versus the density (lane-km/km^2) of county, local, town, interstate, state, and US roads in 82 watersheds with paved roads located in the Adirondack Park of New York, U.S.A.

Variable	Regression			Analysis of variance		
	Coefficient	Standard error	*p*-Value	d.f.	Sums of squares	% contribution
Sodium						
Constant	1.805	0.288	<0.001			
County Route	0.084	0.540	0.877	1	19.16	1.1
Local Road	28.950	24.100	0.234	1	0.82	0.0
Town Road	−0.110	0.203	0.588	1	37.02	2.1
Interstate	5.073	0.735	<0.001	1	399.65	22.3
State Route	5.536	0.325	<0.001	1	985.71	54.9
US Route	8.764	1.766	<0.001	1	88.01	4.9
Regression				6	1530.37	85.3
Residual Error				75	264.57	
Total				81	1794.94	
Chloride						
Constant	3.108	0.482	<0.001			
County Route	0.145	0.902	0.873	1	46.41	0.8
Local Road	86.340	40.280	0.035	1	16.26	0.3
Town Road	0.097	0.339	0.777	1	209.42	3.6
Interstate	7.584	1.229	<0.001	1	1201.07	20.9
State Route	9.827	0.542	<0.001	1	3037.28	52.9
US Route	20.721	2.952	<0.001	1	491.99	8.6
Regression				6	5002.43	87.1
Residual Error				75	738.82	
Total				81	5741.25	

in the region. D'Arcy and Carignan (1997) reported a median sodium concentration of 0.51 mg/L and a range of 0.27–0.92 mg/L for 30 lakes in undeveloped watersheds with similar granitic geology and glacial till and outwash soils located on the Canadian Shield in Quebec, Canada. Our sodium and chloride concentrations also overlap the range of 0.63–1.49 mg/L for sodium and 0.22–0.54 mg/L for chloride reported in the USEPA Eastern Lakes Survey for 10 lakes in

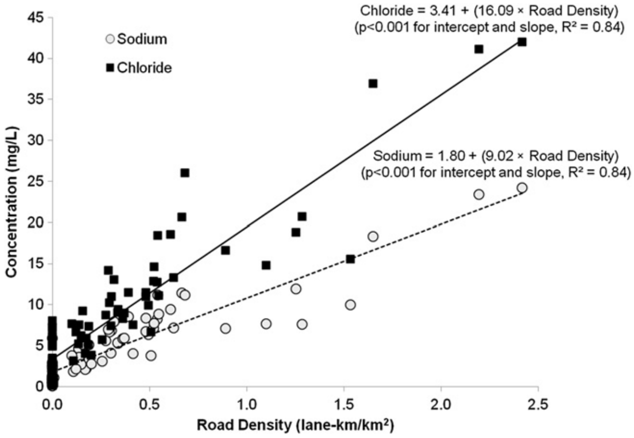

Fig. 5 – Relationship between sodium (open circles) and chloride (solid squares) concentrations (mg/L) in lakes and paved state road density (lane-km/km^2) for 82 watersheds located in the Adirondack Park of New York, U.S.A. State roads are the sum of road lane kilometers of state, interstate, and US highways in each watershed. The dashed and solid lines represent simple linear regression fits through the sodium and chloride data, respectively.

undeveloped watersheds located in the interior of Maine with similar geology and soils as in our study (Landers et al., 1988). The low concentrations of sodium and chloride reflect the low annual inputs of these ions from the combination of atmospheric deposition and rock weathering. Atmospheric deposition contributes about 0.8 kg/ha/yr of sodium and 1.6 kg/ha/yr of chloride to watersheds in the Adirondacks (Johnson and Lindberg, 1992), which are comparable to atmospheric deposition rates on the Canadian Shield (Ouimet and Duchesne, 2005) and the interior of Maine (NADP, 2009). With respect to rock weathering sources, studies on similar granitic parent materials on the Canadian Shield and in the northeastern US report rock weathering releases from 2.6 to 6.0 kg/ha/yr of sodium (Lovett et al., 2005; Ouimet and Duchesne, 2005), while the amount of chloride released via rock weathering from these same materials has been estimated at less than 0.12 kg/ha/yr (Lovett et al., 2005) with this input generally considered insignificant to the overall chloride budget in watersheds with similar granitic parent materials (Lovett et al., 2005; Nimiroski and Waldron, 2002; Rosfjord et al., 2007). The higher concentration of sodium relative to chloride in our lakes in watersheds without paved roads most likely reflects the higher input of sodium from atmospheric deposition plus rock weathering.

The high correlation between road density and sodium and chloride concentrations points clearly to road salt as the primary source of salt loadings to the lakes in our study, but there are other potential anthropogenic sources of sodium and chloride that may have contributed to the higher concentrations in the lakes in watersheds with paved roads, as homes are also present in these watersheds. Two other potentially significant sources of sodium and chloride are

Table 3 – Percent of total variation in sodium and chloride in lakes explained by paved state road density within buffers created from the shoreline of increasing width in 82 watersheds with paved roads located in the Adirondack Park of New York, U.S.A. State road length and percent of total state road length associated with each buffer width are also shown. Variation explained was obtained via simple linear regression, and all regression *p*-values were less than 0.001 ($n = 82$ per regression).

Buffer width (m)	Sodium (%)	Chloride (%)	Road length (lane-km)	% of total road length
10	36	30	13	1
20	34	29	25	2
40	39	36	69	4
80	48	42	170	10
160	64	64	361	22
320	78	76	673	41
640	84	82	1065	66
1280	84	82	1489	92
All roads	84	84	1624	100

sewage disposal systems and water softeners (Rosfjord et al., 2007). Kelly et al. (2008) estimated that sewage disposal and water softeners accounted for 4 and 3%, respectively, of the total sodium chloride load while road salt accounted for 91% of the total sodium chloride load in a rural watershed in southeastern NY. Another study in a more developed watershed in Connecticut but also on similar geology and soils as in our study estimated the load from individual sewage disposal at 15% for sodium and about 1% for chloride, with road salting estimated to contribute 66 and 90% of the sodium and chloride load, respectively (Nimiroski and Waldron, 2002). Given that our watersheds are greater than 90% forested and that our

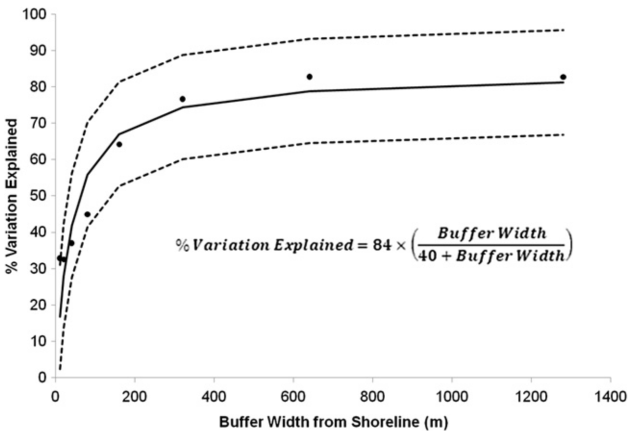

Fig. 6 – Percent of total variation in sodium and chloride explained by the density of paved state roads within increasing buffer widths from the shoreline for 82 watersheds located in the Adirondack Park of New York, U.S.A. The points represent the mean variation of sodium plus chloride explained by paved state road density within each buffer width, the solid line represents a fit of a nonlinear asymptotic function through the data, and the dashed lines represent 95% confidence intervals on the nonlinear fit ($n = 8$, $r^2 = 0.87$).

population density is much lower than in these other studies (the AP has a population density of 0.05 persons/ha, while the population density in the southeastern NY study was 0.4 persons/ha), we believe that the contribution of sewage disposal and water softeners to the salt load in our study lakes to be negligible relative to road salt.

Though sodium and chloride concentrations in our lakes in watersheds with paved roads were significantly higher than the least impacted condition, interpreting this difference from a lake ecosystem health perspective is challenging. For example, acute toxic response from select invertebrates and fish typically occurs when chloride concentrations are well above 1000 mg/L (Corsi et al., 2010; reviewed by Environment Canada, 2001), while chronic effects can be detected at an order of magnitude lower (Corsi et al., 2010; Elphick et al., 2011; USEPA, 1988). However, road salt may impact lake ecosystems in other ways at concentrations below those documented to be toxic to aquatic organisms. Increased salt concentrations have been linked to colonization of invasive species (Richburg et al., 2001), mobilization of heavy metals from roadside soils (Amrhein and Strong, 1990), and prolonged stratification of the water column (Bubeck et al., 1971). Thus, as suggested by Kelly et al. (2008), there may be significant ecological effects at concentrations lower than those published for aquatic biota.

Although the concentrations detected in our study are still low, it is important to recognize that these increases have occurred in a short period of time in watersheds that are primarily rural. The use of large amounts of road salt in the Adirondack region was initiated in the late 1970s as part of a clear roads policy enacted prior to the 1980 Olympic Winter Games in Lake Placid, New York, thus the bulk of the changes to sodium and chloride concentrations have probably occurred within the last three decades. Other researchers in the U.S and Europe have observed increased salinization of lakes in similarly short time frames (Müller and Gächter, 2012; Rosfjord et al., 2007). For example, Siver et al. (1996) demonstrated a significant increase in both sodium and chloride in a 26 year time period in a study of 40 lakes in Connecticut. Some of the lakes influenced by an increase in residential and urban development saw increases as great as 2.5 times for sodium and 2.9 times for chloride. These increases are substantially lower than the 5.5 fold increase for sodium and 29 fold increase for chloride shown in our study, which would have occurred in about the same timeframe.

We believe there are two major explanations for the more rapid increase in sodium and chloride in lakes in watersheds with paved roads in the AP compared to other studies. First, compared to other states in the northeastern US, the NYSDOT applies the largest amounts of sodium chloride to state roads annually (Kelting and Laxson, 2010), thus the total load of sodium chloride applied to state roads in the AP may be higher than in other regions. Second, the AP is dominated by coarse textured glacial till and outwash soils of granitic origin (Sullivan et al., 2006; and Table 1), these soils have high infiltration and percolation rates, thus sodium and chloride dissolved in the soil pore water can leach rapidly with little opportunity for retention in the soil matrix. Though a portion of infiltrated sodium would be retained via cation exchange, evidence for which is provided by the 0.85 M ratio of sodium to chloride that indicates preferential sodium retention (Jackson and Jobbagy, 2005), much of the sodium would leach to surface or groundwater because of the coarse textured soils. Chloride is retained in soils via chlorination of organic matter, anion exchange, plant uptake, and entrainment in soil micropores (Bastviken et al., 2006; Kelly et al., 2008; Lovett et al., 2005; Svensson et al., 2007). The capacity of these mechanisms to retain chloride in AP soils is most likely lower than in other regions that have finer textured soils, which would generally have more organic matter and micropores (Plante et al., 2004; Brady and Weil, 2010) and thus greater soil retention capacity for chloride. Thus, the combination of higher loads and lower capacity for soil retention probably explains the greater salinization rate in AP lakes.

The impact of winter road maintenance on the chemistry of receiving waters has been well covered in the literature, yet in order to mitigate these impacts it is essential to have an understanding of the relationship between road density and surface water salinization. Although numerous studies have demonstrated the relationship between road salt use and increase salinization of surface waters, few studies have carefully examined the impact of varying road densities, and no studies have looked specifically at their impact on lake chemistry, only stream chemistry. Most studies only relate salinization to roads in a general sense by examining percent impervious surfaces (Kaushal et al., 2005), road density in one watershed (Kelly et al., 2008), or a small number of watersheds (Gardner and Royer, 2010). One exception is Daley et al. (2009) who determined that total road pavement density in 44 watersheds explained 78% and 75% of the variability in sodium and chloride concentrations in stream water in New Hampshire. In contrast, our analysis of lake chemistry of 82 watersheds revealed that a weak relationship exists with total road density but a much stronger relationship exists when road type is considered. The combination of Interstate highways, US routes, and State routes explain 84% of the variation in both sodium and chloride concentrations in lakes. The weak relationship we found between total road density and sodium and chloride concentration is likely due to highly variable winter road maintenance procedures across state, local, and county roads. For example, Interstate highways, US routes, and State routes are all managed by the New York State Department of Transportation. The NYSDOT relies heavily on road salt to prevent snow and ice from bonding to the pavement (anti-icing) and to loosen snow and ice that has already bonded (deicing) (NYSDOT, 2006). Because local, town and county roads in the AP receive less road salt and are typically managed with a combination of plowing, sanding, and salting, their contribution to explaining the total variation in sodium and chloride in lakes is significantly less than state roads. Specific data on how local municipalities treat non-state roads would improve our understanding of the effect of paved roads on sodium and chloride concentrations in lakes.

As recommended by Jackson and Jobbagy (2005) managers need tools to help identify sensitive areas to receive alternative treatments in order to reduce salt loads to surface waters. In New York State, the NYSDOT has begun an environmental initiative that includes protection and restoration of water quality as a principal goal (NYSDOT, 2011). Attaining this goal first requires identifying sensitive areas along NYSDOT roads within the Adirondack Park. Our model could be used as part of the NYSDOT planning process in a GIS framework to help identify sections of road that contribute more to sodium and chloride concentrations in lakes based on road proximity to shoreline.

For illustration purposes only, we applied our model to the entire 4530 lane-km state road network in the AP (see Fig. 1A), identifying all road segments that were within 80 m of lakeshore. There are 320 lane-km of state roads within 80 m of lakeshore within the AP, which represents only 7% of total state roads. Assuming that our model represents the AP as a whole, and we believe that it does given the spatial extent and range of watershed characteristics captured in our dataset, this 7% or 320 lane-km of state roads located within 80 m of shoreline would account for 45% of the variation in sodium and chloride (see Fig. 6).

An assumption we make is that targeting road segments within buffers for alternative treatment will reduce sodium and chloride loads and ultimately lake concentrations. We don't know the rate at which lakes will respond in terms of observing reduced concentrations of sodium and chloride; this would depend on the treatments and how long it takes for accumulated salt to be released from the watersheds. No reductions in stream sodium concentrations were observed in a study on similar geology and soils that monitored concentrations for 10 years after reducing the road salt application rate by 60% (Nimiroski and Waldron, 2002). These authors suggested that the stream sodium concentration was being maintained by mobilization of accumulated exchangeable soil sodium by the calcium in their alternative deicing treatment of calcium chloride. A significant portion of the chloride in road salt accumulates in groundwater and this contaminated groundwater contributes a significant load of chloride to surface waters (Kelly et al., 2008), which may maintain a high chloride concentration even after the road salt application rate is reduced. Thus, though the concentrations increased rapidly with the onset of road salting in our study, given that retention and accumulation have occurred, the reverse trend would likely be slower. But, stabilization at current concentrations may be an acceptable and reasonably achievable goal, and we believe that our approach of linking regional monitoring data to road networks would be applicable to other regions and countries interested in identifying more sensitive areas to achieve similar goals.

5. Conclusions

- Lakes in least impacted watersheds without paved roads in the AP have very low sodium and chloride concentrations, with the majority having concentrations less than 1 mg/L for both ions.
- Compared to concentrations in least impacted watersheds, road salting has increased sodium and chloride concentrations dramatically in watersheds with paved roads.
- Roads maintained in winter following NYSDOT deicing protocols are the greatest contributors to salinization of lakes in the Adirondack Park.
- The contribution of state roads to lake sodium and chloride concentrations can be modeled as a function of road

proximity to shoreline, which provides a useful tool for identifying areas to treat differently to reduce environmental impacts.

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Landscape level estimate of lands and waters impacted by road runoff in the Adirondack Park of New York State

Sean A. Regalado · Daniel L. Kelting

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Abstract Road runoff is understood to be a significant stressor in terrestrial and aquatic ecosystems, yet the effects of this stressor are poorly understood at large spatial scales. We developed an efficient method for estimating the spatial impact of road runoff on lands and waters over large geographic areas and then applied our methodology to the 2.4 million ha Adirondack Park in New York State. We used TauDEM hydrologic modeling and a series of ESRI GIS processes to delineate surface flow downslope of paved roads, illustrating the potential movement of pollutants originating from paved roads through the USGS 10 m DEM topography. We then estimated the land and surface water areas, number of water bodies, and total stream length potentially impacted by road runoff from paved roads. We found that as much as 11 % of land area, 77 % of surface water area, 1/3 of the water bodies, and 52 % of stream length in the Adirondack Park may be impacted by road runoff. The high degree of hydrologic association between paved roads and the lands and waters of this region strongly suggests that the environmental impacts of road runoff should be evaluated along with other regional stressors currently being studied. Being able to estimate the spatial impact of road runoff is important for designing monitoring programs that can explicitly monitor this stressor while also providing opportunities to understand the interaction of multiple environmental stressors.

S. A. Regalado · D. L. Kelting (✉)
Adirondack Watershed Institute, Paul Smith's College,
Routes 86 & 30, PO Box 265, Paul Smiths, NY 12970, USA
e-mail: dkelting@paulsmiths.edu

S. A. Regalado
e-mail: sregalado@s.paulsmiths.edu

Keywords Road runoff · Road salt · GIS · TauDEM · Multiple environmental stressors · Adirondack Park

Introduction

The 2.4 million ha Adirondack State Park in northern New York State, USA, encompasses the world's largest intact temperate forest which contains a globally unique landscape of wetlands, northern hardwood and boreal forests, alpine tundra, and vast fresh water resources (Jenkins and Keal 2004). This large rural landscape is exposed to multiple environmental stressors operating at varying spatial scales and often in interacting ways that are poorly understood (Allan 2004; Palmer and Yan 2013). Historically, the principal environmental stressors studied in the region have been acid deposition and climate change (Ito et al. 2002; Stager et al. 2009). In recent years, road runoff has been identified as a significant threat to aquatic and terrestrial ecosystems in the region, comparable to acid deposition (Kaushal et al. 2005; Kelting and Laxson 2010). Over 8000 km of paved roads traverse the Adirondack Park; therefore, the potential land and surface water area receiving road runoff pollutants may be significant and likely also occurs on protected state lands which represent 45 % of the area.

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The main categories of pollutants found in road runoff are road de-icing salts, heavy metals, herbicides, and polycyclic aromatic hydrocarbons (Trombulak and Frissell 2000; Fay and Shi 2012; Tang et al. 2013). Road runoff pollutants have been shown to induce tree mortality (Fan et al. 2014), shift the composition of floral and faunal communities (Kaspari et al. 2010; Ke et al. 2013; Neher et al. 2013; Snell-Rood et al. 2014), promote establishment of terrestrial and aquatic invasive species (Johnston and Johnston 2004; Crooks et al. 2011), and elevate Na concentrations in roadside soils, displacing base cations (Ca, Mg, K) essential to ecosystem function (Norrström and Bergstedt 2001). Flux of displaced cations and Cl into streams has been reported downslope of salted roads (Daley et al. 2009; Price and Szymanski 2013; Corsi et al. 2015), and Na and Cl concentrations in lakes correlate positively with paved road density (Kelting et al. 2012). Road runoff pollutants in lakes and rivers can increase fish and amphibian mortality (Yousef et al. 1983; Karraker et al. 2008) and may induce top-down (Sloman et al. 2003) and bottom-up trophic cascades (Hodkinson and Jackson 2005). Road runoff pollutants can affect ecosystem function similarly to other regional stressors (Norrström and Bergstedt 2001) and can often interact synergistically, enhancing their effects on terrestrial and aquatic communities (Holmstrup et al. 2010). Pollutants commonly found in road runoff have been shown to exacerbate the effects of warming temperatures (Qiang et al. 2012), drought (de Silva et al. 2012), acid deposition (Rosfjord et al. 2007), eutrophication (Ferreira et al. 2008), and disease (Kiesecker 2002).

Although the impacts of road runoff on terrestrial and aquatic ecosystems are well known, no studies have been published that model the spatial extent of road runoff over large geographic areas. Accurate knowledge of the spatial extent of road runoff is essential to understanding and managing the interacting impacts of road runoff and other regional stressors on aquatic and terrestrial ecosystems (Soranno et al. 2014). Our objectives were to develop an efficient method of estimating the spatial extent of road runoff impacts on lands and waters over large geographic extents and to quantify the land area and surface water areas, number of water bodies, and total length of streams potentially impacted by road runoff from state and federal, county, and local roads within the Adirondack Park.

Methods

Study area

The Adirondack Park (Park) is located in New York State, USA (Fig. 1). The Park is 2.4 million ha in size and is roughly divided equally between private and public lands that are dominated by natural forest cover. The Park contains over 8000 km of paved roads, broken out into 1965 km of state and federal roads, 1803 km of county roads, and 4421 km of local roads (Table 1). The Park also contains about 103,000 ha of surface water (lakes and ponds) and over 13,000 km of streams.

Estimating spatial extent of road runoff

Spatial data for depth to bedrock and groundwater flow was lacking for the Adirondack region, so we used surface topography to delineate the potential flow of runoff downslope of paved roads. We used ArcGIS - ArcMap 10.2 (Esri, Redlands, CA) with a hydrologic terrain analysis toolset TauDEM 5.1.1 (TauDEM Toolbox 5.1.1, www.hydrology.usu.edu/taudem) to develop a series of parameters: a hydrologically relevant surface based on the D-infinity flow direction algorithm (Tarboton 1997), indicator grids of rasterized road networks, effective precipitation weights, and decay multipliers (Fig. 2). TauDEM 5.1.1's "D-Infinity concentration limited accumulation" tool used these parameters to delineate flow downslope of the rasterized road networks. We then used a series of ArcGIS processes to quantify the total land area represented by the downslope accumulations, as well as to identify the hydrologically connected water bodies, and to quantify the length of rivers and streams downstream of road runoff. We used Adirondack Park Agency (APA) land use classification shapefiles to determine the estimated lands and waters impacted by road runoff in the Park's forest preserves. Our methods are briefly described here, a more detailed description is provided in the Appendix.

Producing a hydrologically relevant surface

We obtained 202 USGS 10 m New York State 7.5-min Digital Elevation Models (DEMs) from the Cornell University Geospatial Information Repository (cugir.mannlib.cornell.edu). These DEMs were mosaicked; then, sinks were removed by raising their elevation to the lowest pour point using the TauDEM

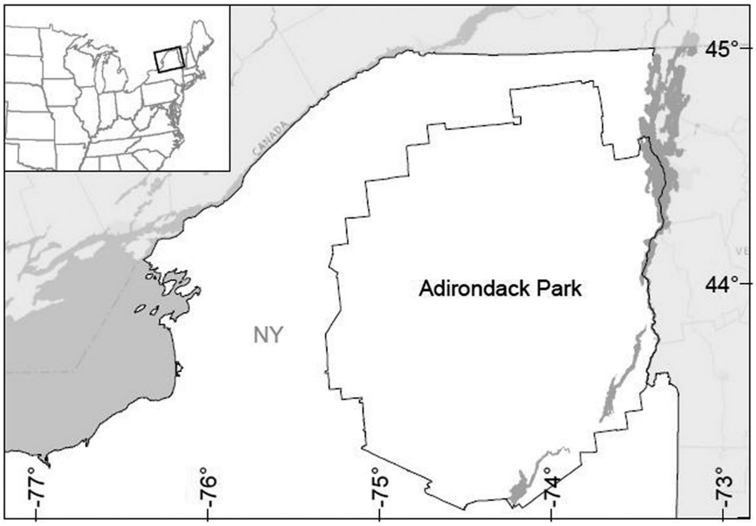

Fig. 1 Location of Adirondack Park in New York State, USA

5.1.1 Pit Remove tool (TauDEM Toolbox 5.1.1, www.hydrology.usu.edu/taudem). Pits often occur due to deficiencies in DEM production (Jenson and Domingue 1988). A D-infinity flow direction grid was derived from the pit removed DEM using the TauDEM D-infinity flow direction algorithm (Tarboton 1997).

Producing road disturbance grids

Road disturbance grids indicate the spatial location of road runoff inputs in the flow delineation process. Road polyline shapefiles were created by extracting road polylines from NYSDOT Local Highway Inventory datasets by three separate road categories: state and federal (SF), county (C), and local (L) roads (apa.ny.gov/gis). The ESRI "buffer" tool was used to create 100 m road spray influence shapefiles for each category, including a combined dataset of all paved roads (SFCL). The buffer added 100 m to each side of the road. These eight shapefiles (SF, C, L, and SFCL and 100 m SF, 100 m C, 100 m L, and 100 m SFCL) were converted into 10 m cell size raster datasets using the ESRI "polyline to new raster" tool and the "polygon to new raster" tool. Field experiments have shown vehicular spray, and wind can transport pollutants hundreds of meters from the road (e.g., Zechmeister et al. 2005; Bernhardt-Roemermann et al. 2006), so the 100 m buffer was selected as a conservative estimate of road spray.

Table 1 Total road length by category, total area of surface water, and total length of streams in the Adirondack Park, New York

Landscape parameter	Total park
State and federal roads (km)	1965
County roads (km)	1803
Local roads (km)	4421
Surface water (ha)	102,792
Stream length (km)	13,153

Producing input weights

An effective runoff grid and a first-order decay multiplier grid were required model parameters. Effective runoff grids represent the precipitation over an area. For simple contributing area delineations such as this, the weighting field was a constant raster set to 1 (Tarboton et al. 2009). The decay multiplier grid value was an exponential decay factor. The decay factor for road runoff was set to a constant raster of 1; this value was arbitrary for we were only interested in the spatial extent of runoff, not any quantitative accumulation values.

Delineating overland flow grids

TauDEM's "D-Infinity concentration limited accumulation" tool was used to delineate substance accumulation downhill of the eight disturbance indicator grids, SF, C, L, and SFCL and the 100 m buffered SF, C, L, and SFCL.

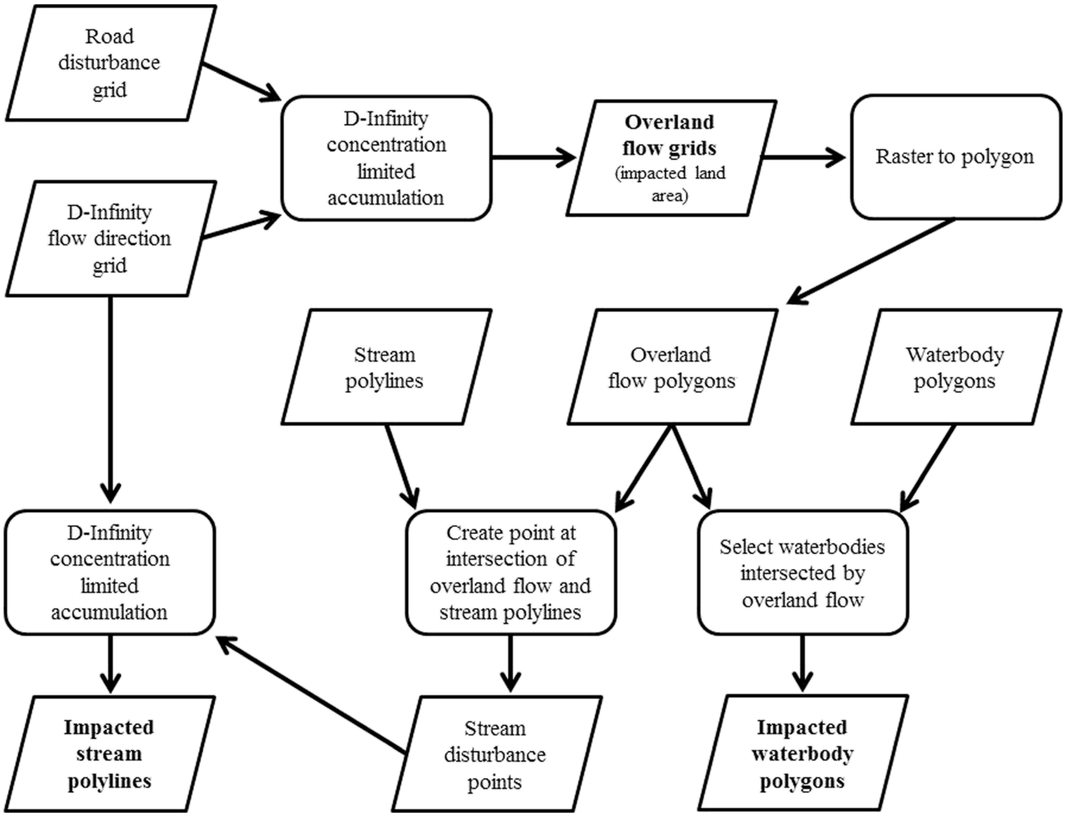

Fig. 2 Flow chart of TauDEM hydrologic modeling and ArcGIS processes used to delineate surface flow downslope of paved roads and to estimate the land area, surface water area, and total stream length impacted by road runoff from paved roads

Quantifying impacted land area

Cells with a value greater than zero represented overland flow. The >0 cells were extracted using the ArcGIS "extract by attribute" tool. To exclude overland flow cells that traversed open water areas, extracted cells were multiplied by a park wide open water raster dataset (cell values: open water=NoData, land=1) using the ArcGIS "times" tool.

Identifying simultaneous impacts of multiple road categories

Lands impacted by multiple road sources were found by multiplying the overland flow rasters by category using the ArcGIS "times" tool (SF×C, SF×L, SF×C×L, et cetera). Lands not impacted by the multiple roads in question resulted in a NoData output value.

Identifying impacted lakes and ponds

The original eight overland flow grid outputs (those without open water cells removed) were converted to polygons using the ArcGIS "raster to polygon" tool. ArcGIS's "select by location" tool was used to select lakes and ponds from the NYS area hydrography shapefile that intersected with the overland flow polygons. Intersected lakes and ponds were extracted according to the road category they intersected with. These intersected lakes and ponds were hydrologically connected to road runoff based upon the DEM-derived flow direction.

Calculating downstream length of impacted streams

The converted polygons of the eight overland flow categories were intersected with linear hydrography polylines from a NYS hydrography dataset of rivers and streams (gis.ny.gov) using the ArcGIS "intersect" tool. Point shapefiles were created and then converted to raster datasets using the ArcGIS "point to raster" tool and were treated as disturbance indicator grid parameters for the TauDEM "concentration limited accumulation" tool. The eight subsequent flow grids were then converted to polylines. The eight polylines shape files (SF, C, L, and SFCL and 100 m SF, C, L, and SFCL impacted streams) represented impacted stream channels.

Quantifying impacted lands and waters in APA forest preserve lands

The ArcGIS "zonal statistics as table" tool was used to categorize total impacted terrestrial area for each of the 15 APA land use types in the Adirondack Park Land Use and Development Plan dataset (apa.ny.gov/gis).

Results

A total of 156,561 ha of land area was estimated to be impacted by road runoff, which constituted 6.5 % of the land area of the Park (Table 2). Of this total, runoff from state and federal roads was estimated to impact 36,991 ha of land area, while runoff from county and local roads was estimated to impact 44,182 and 96,550 ha of land area, respectively. The total land area estimated to be impacted by road runoff increased to 254,224 ha when including the 100 m buffer on all roads, which constituted 11 % of the land area of the Park. The relative contribution to impacted land area from state and federal, county, and local roads was the same with and without the 100 m buffer and was proportionate to the length of each road type in the Park (e.g., local roads represented 54 % of total road length and 61 % of the impacted land area). An example of the relationship between the three road types and land area impacted by road runoff is provided for a 5000-ha area in the vicinity of the Village of Saranac Lake within the Park (Fig. 3). In this example, runoff from state and federal roads impacted 6 % of the land area (Fig. 3a), runoff from county roads impacted 5 % of the land area (Fig. 3b), runoff from local roads impacted 11 % of the land area (Fig. 3c), and all three road types together impacted 18 % of the land area (Fig. 3d). The close association between the road network and impacted land and surface waters is also clearly seen.

We found a total of 1787 ha of land to be impacted by state, federal, county, and local roads simultaneously (Table 3). This value increased to 4042 ha when including the 100 m buffer. State, federal, and county roads simultaneously impacted 3255 ha without the buffer and 6980 ha with the buffer. State, federal, and local roads simultaneously impacted 8378 ha without the buffer and 19,009 ha with the buffer. County and local roads simultaneously impacted 11,228 ha without the buffer and 20,837 ha with the buffer.

A total of 77,860 ha of surface waters was estimated to be impacted by road runoff, which constituted 76 % of the surface water area of the Park (Table 2, Fig. 4d). Note that two large lakes, Great Sacandaga Reservoir and Lake George, constitute 26 % of this total. Including all lakes, runoff from state and federal roads was

Table 2 Land and surface water area, number of lakes, and stream length impacted by surface runoff from paved roads with and without a 100 m buffer in the Adirondack Park, New York

Landscape parameter	Road category			Total[a]
	State and federal	County	Local	
Land area (ha)	36,991	44,182	96,550	156,561
Land area (ha) (100 m)	63,018	71,291	164,823	254,224
Surface water area (ha)	62,656	62,218	72,982	77,860
Surface water area (ha) (100 m)	62,852	62,656	73,777	78,792
Number of lakes	326	299	644	820
Number of lakes (100 m)	351	328	711	884
Stream length (km)	2795	2859	4915	5934
Stream length (km) (100 m)	3238	2976	5230	6830

[a] Total is not the sum of the land and water impact of each road category as it includes land and water values with overlap

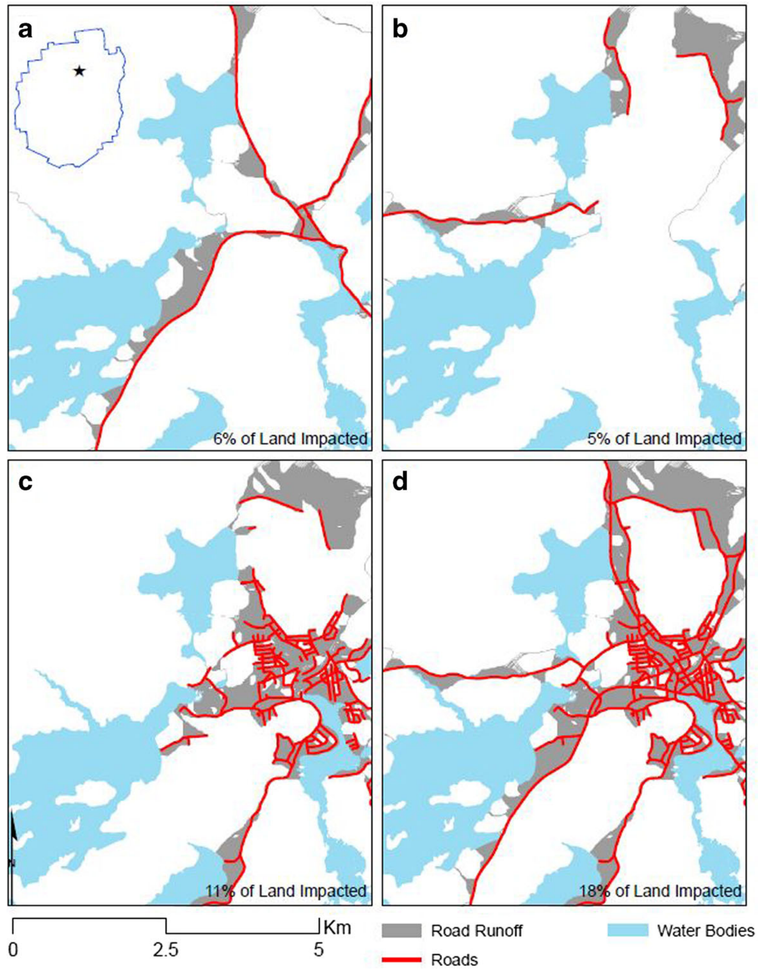

Fig. 3 Estimated land area impacted by road runoff from state and federal (**a**), county (**b**), local (**c**), and the total paved road networks (**d**) in a 5000-ha area centrally located within the Adirondack Park, New York. Location is indicated by the star symbol on map inserted into Fig. 3a

estimated to impact 62,656 ha (61 %, Fig. 4a) of surface water area, while runoff from county and local roads was estimated to impact 62,218 ha (61 %, Fig. 4b) and 72,982 ha (71 %, Fig. 4c) of surface water area, respectively. The total surface water area estimated to be impacted by road runoff increased slightly to

Table 3 Land and surface water area, number of lakes, and stream length impacted by surface runoff from multiple road categories with and without a 100 m buffer in the Adirondack Park, New York

Landscape parameter	Multiple road categories			
	State, federal, and county	State, federal, and local	County and local	State, federal, county, and local
Land area (ha)	3255	8378	11,228	1787
Land area (ha) (100 m)	6980	19,009	20,837	4042
Surface water area (ha)	61,503	63,217	63,268	59,893
Surface water area (ha) (100 m)	62,279	63,601	64,915	60,633
Number of lakes	207	287	269	184
Number of lakes (100 m)	216	312	291	194
Stream length (km)	1005	1295	2056	922
Stream length (km) (100 m)	1305	1731	2442	1222

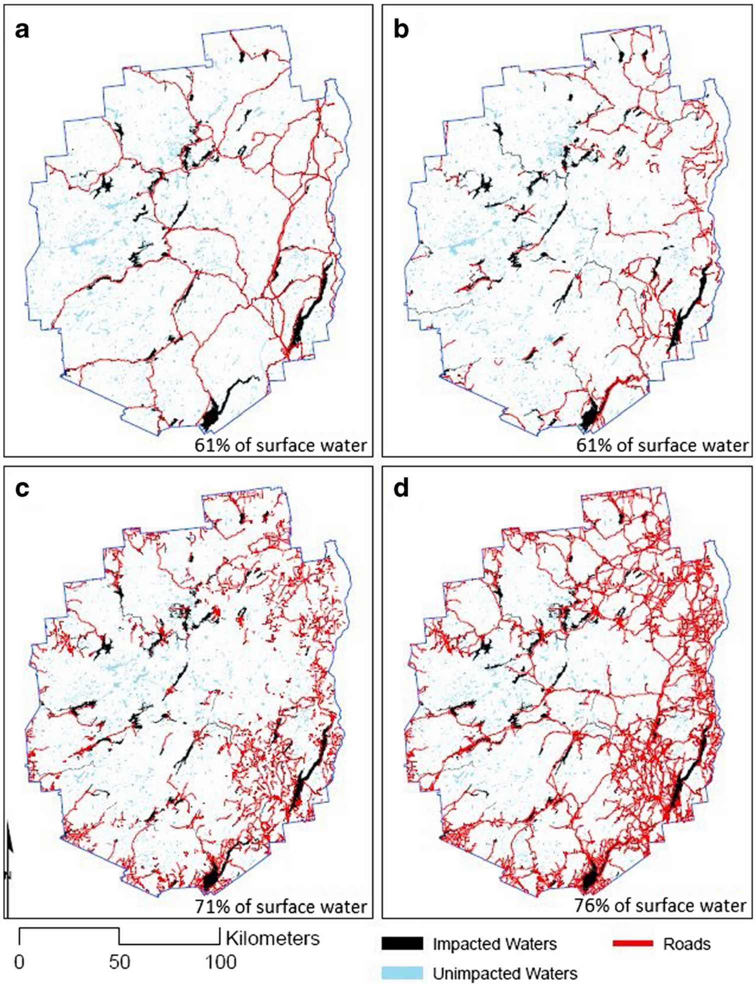

Fig. 4 Estimated surface waters impacted by road runoff from state and federal (**a**), county (**b**), local (**c**), and the total paved road networks (**d**) within the Adirondack Park, New York

78,792 ha when including the 100 m buffer on all roads, which constituted 77 % of the surface water area of the Park. There were a total of 820 waters impacted by road runoff, with 326 waters impacted by state and federal roads, 299 waters impacted by county roads, and 644 waters impacted by local roads (Table 2). The relative contribution to impacted surface water area from state and federal, county, and local roads was the same with and without the 100 m buffer.

We found a total of 59,893 ha of surface water area to be impacted by the four road types (state, federal, county, and local roads) simultaneously, with this value increasing slightly when including the 100 m buffer (Table 3). The other multiple road categories had similar areas of surface waters impacted. The four road types impacted 184 lakes and 922 km of stream length simultaneously, with these values increasing to 194 lakes and 1222 km of stream length when including the 100 m buffer. The number of lakes and stream length impacted by the other multiple road categories was more variable compared to surface water area; for example, county and local roads simultaneously impacted over twice the stream length (2056 km) compared to the four road types together (922 km).

A total of 5934 km of stream length was estimated to be impacted by road runoff, which constituted 45 % of stream length in the Park (Table 2, Fig. 5d). Of this total, runoff from state and federal roads was estimated to impact 2795 km (21 %, Fig. 5a) of stream length, while runoff from county and local roads was estimated to impact 2859 km (22 %, Fig. 5b) and 4915 km (37 %, Fig. 5c) of stream length, respectively. The total stream length estimated to be impacted by road runoff increased to 6830 km (a 15 % increase) when including the

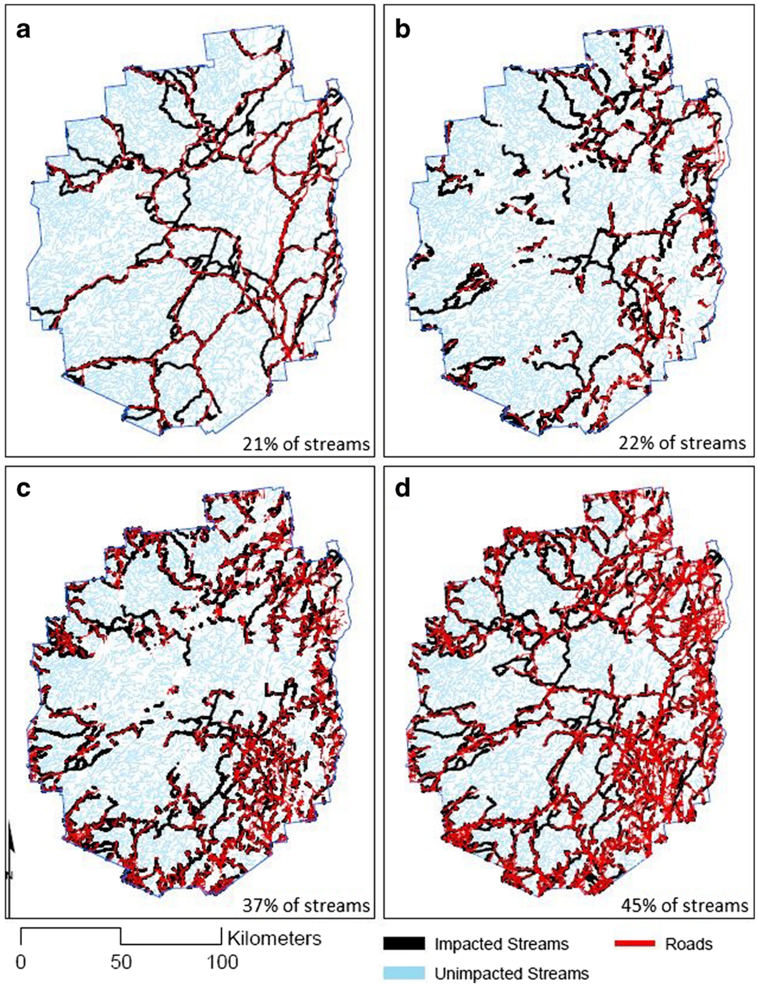

Fig. 5 Estimated streams impacted by road runoff from state and federal (**a**), county (**b**), local (**c**), and the total paved road networks (**d**) within the Adirondack Park, New York

100 m buffer on all roads, which constituted 52 % of stream length the Park.

Of the total 156,561 ha of land area estimated to be impacted by road runoff, 20,474 ha (13 %) was in the forest preserve, including the 100 m buffer which increased the impacted land area to 34,889 ha (Table 4). Of the total 78,153 ha of surface water area estimated to be impacted by road runoff, 67,496 ha (86 %) was in the forest preserve. Of the total 820 lakes estimated to be impacted by road runoff, 334 (40 %) were in the forest preserve. Of the total 5934 km of stream length estimated to be impacted by road runoff, 1152 km (19 %) was in the forest preserve. Within the forest preserve, 12 % of the impacted land area and 26 % of the impacted surface water area were located in lands classified as Wilderness, which are the most protected lands in the Park.

Discussion

Impacted lands and waters

Our results show a high degree of connectivity between paved road networks and lands and waters within the Park. Due to this high connectivity and the basic hydrologic connectivity between water and its watershed (Hynes 1975; Vannote et al. 1980), road runoff pollutants may flow through substantial land and surface water area in the Park. This area may be significantly greater than what we estimated with our 100 m buffer, as field experiments have shown vehicular spray and wind can transport pollutants hundreds of meters from the road (Zechmeister et al. 2005; Bernhardt-Roemermann et al. 2006); thus, our estimates are likely conservative. The results further show that local roads impact substantially more land area than state and federal roads;

Table 4 Land and surface water area, number of lakes, and stream length impacted by surface runoff from paved roads with and without a 100 m buffer located in state-owned forest preserve lands within the Adirondack Park, New York

Landscape parameter	Road category			Total[a]
	State and federal	County	Local	
Land area (ha)	7118	4490	10,470	20,474
Land area (ha) (100 m)	12,360	7196	18,274	34,889
Surface water area (ha)	58,177	57,163	63,530	67,496
Surface water area (ha) (100 m)	58,306	58,426	64,002	68,216
Number of lakes	157	126	254	334
Number of lakes (100 m)	165	135	272	353
Stream length (km)	454	474	860	1099
Stream length (km) (100 m)	477	447	927	1152

[a] Total is not the sum of the land and water impact of each road category as it includes land and water values with overlap

however, the impact of state and federal roads may be more intense. For example, road salt loading is largest for state and federal roads in the Park (Kelting and Laxson 2010) and the loading of heavy metals and PAH pollutants has been shown to increase with road use intensity, giving evidence to a disproportionate impact of high-use roads (Klimaszewska et al. 2007). The impacts of road runoff may be compounded in areas hydrologically connected to multiple road categories, which represents a challenge to monitoring and management as the road categories are managed in different ways. While state and federal roads are all salted, county, town, and local roads in the Park receive a variety of winter road management treatments from salting to just plowing (Kelting et al. 2012). Modeling flow downslope of high-use roads will identify high-risk land areas for effective monitoring regimes.

A substantial area of land downslope of roads may be at risk from decreasing soil fertility and increasing toxins. Road salt (NaCl) is widely used on roads throughout the region for snow and ice control, for example the New York State Department of Transportation applies 28 t of road salt per kilometer of state and federal road annually (NYSDOT 2006). Increased Na concentrations in road runoff have similar base cation depleting effects as acid deposition (Norrström and Bergstedt 2001) and may become a more relevant stressor as public policy has greatly reduced levels of acid deposition in the northeast (Waller et al. 2012). Given that 67,496 ha of land may be impacted by runoff from 1965 km of state and federal roads, we can assume that this land area is at risk of base cation depletion and may also be experiencing the toxic effects of other associated road pollutants such as heavy metals and PAHs. Road salt discharge compounded by the northeasterly gradient of acid deposition in the Park (Ito et al. 2002) can also have implications for soil fertility (Gałuszka et al. 2011), tree physiology (Fan et al. 2014), and forest community dynamics (McEathron et al. 2013).

Aquatic ecosystems and groundwater resources hydrologically connected to paved roads may also be at risk from the toxic effects of road runoff pollutants. Regional salinization of lakes in the Park has already been documented by Kelting et al. (2012), who found state and federal road densities explained 86 % of the variation in both Na and Cl concentrations in lakes. Similar relationships between road density and salt concentration in surface waters were reported for studies in New Hampshire (Daley et al. 2009) and Rhode Island (Nimiroski and Waldron 2002). Now, through this new work, we have estimated that 76 % of surface water area and at least 45 % of stream length (dependent on buffer width) in the Park may be receiving road runoff pollutants such as salt. Salt is one of the most widely studied road runoff pollutants, and the effects of salinization on freshwater ecosystems are well documented; for example, studies have reported reduced aeration and water circulation at depth (Fay and Shi 2012), decreased spotted salamander (*Ambystoma maculatum*) and wood frog (*Rana sylvatica*) survival (Karraker et al. 2008), shifts in community structure (Collins and Russel 2009), reduced copepod density and changes in algal resources (Meter et al. 2011), and decreased productivity at all trophic levels in model freshwater communities (Dalinsky et al. 2014). A significant amount of road salt also enters groundwater where it can accumulate over years and exceed thresholds for potable water and aquatic organisms (Perera et al. 2012).

Given the large regional extent of road runoff and the potential for negative ecological effects at a large scale, the effects of this stressor should be more deliberately evaluated, particularly with its potential to interact with other regional stressors. Road salt, and other runoff pollutants, may confound the recovery of acidified lakes, complicating the assessment of lake management and policies on air pollution (Rosfjord et al. 2007). Jensen et al. (2014) found the biological recovery of an acidified lake receiving road salt discharge lagged behind the biological recovery of a similarly acidified lake not receiving road salt discharge. Thus, the application of road salt may create a mosaic of varying levels of lake recovery from acid deposition in the Park. Additionally, climate change may exacerbate the effects of road runoff pollutants on aquatic ecosystems (Schiedek et al. 2007). Warming spring temperatures break the diapause of aquatic invertebrates (Goddeeris et al. 2001) which corresponds temporally with high stream salt loads in snow melt (Oberts 1994). Also considering that the variability and intensity of spring storms are projected to increase in the northeastern USA (Hayhoe et al. 2007), the likelihood of physiologically active aquatic invertebrates being exposed to road salt discharge and other pollutants must increase (Silver et al. 2009). Our method for identifying areas receiving road runoff is vital to understanding the extent to which these multiple stressors of water quality interact across a large geographic area.

We can conceptualize road runoff from the road network as a geographically extensive stressor with spatially explicit inputs, and from this perspective, the ecological implications of road runoff are largely unknown. Acid deposition and climate change in the northeast are characterized at the regional scale by a smooth distribution of continuous stress inputs over an extensive geographic area, or in other words, the stress inputs of acid deposition and climate change do not vary much from hectare to hectare and thus are monitored at coarse scales (Ito et al. 2002; Hayhoe et al. 2007). Conversely, eutrophication is characterized at the watershed scale by discontinuous, but spatially explicit stress inputs (Agha et al. 2012) and therefore monitored at finer scales. Road runoff shares characteristics of each of the abovementioned stressors. Total runoff from a road network spans the entire region similar to that of acid deposition and climate change, but the stress inputs occur in spatially explicit distributions similar to eutrophication, creating a mosaic of environmental stress defined by topography and road position which is never the same from hectare to hectare.

Application of methods

Our methods create a spatially defined sampling unit that provides the framework for regional monitoring of road runoff as well as provides an estimate of aquatic and terrestrial ecosystems that may benefit from reductions in road salt applications (McDonald 2002; Kilgour et al. 2014). As mentioned above, road runoff pollutants often share similar environmental impacts as other regional stressors and often act synergistically, enhancing their effects on ecosystem function (Palmer and Yan 2013). A solid understanding of the spatial extent of road runoff stress allows researchers to focus regional monitoring efforts as well as develop study designs for experiments investigating the interactions of our significant regional stressors (Fancy et al. 2009).

Our results show that roads are hydrologically connected to a substantial area of land and water in the Park's forest preserve, which includes four categories of land use: wilderness, primitive, canoe, and wild forest. The wilderness category is the most protected land and is managed to protect or restore its natural conditions (apa.ny.gov). Yet, we find that roads connect to over 5000 ha of land and nearly 70 % of surface water in the wilderness category. Road runoff may put at risk substantial forest preserve lands given the multitude of well-studied ecological impacts of runoff pollutants. Our methods may direct management to use alternatives to road salt in these sensitive areas (Jackson and Jobbagy 2005).

There is no New York State-wide monitoring program that chooses monitoring sites based on a single, spatially sound framework. The selection of monitoring sites in the state is predominately based on voluntary cooperation of lake associations (New York DEC 2009). New York State primarily utilizes the Citizen Statewide Lake Assessment Program (CSLAP; www.dec.ny.gov) to produce data for US Environmental Protection Agency Clean Water Act reports. CSLAP monitors sites on the basis of voluntary cooperation of lake associations, although the in-lake sampling protocols used adhere to the literature (Kishbaugh 1988). The New York State Department of Conservation's Rotating Integrated Basin Studies and Lake Classification and Inventory programs monitor specific water bodies, but in a rotation of the State's 17 major basins and the methodologies of choosing sampling sites are based on random and targeted selections, but are not based upon a regional framework and may produce sampling bias

(New York DEC 2009; Larsen et al. 2007). Few lakes are monitored by CSLAP in the Park, though two other programs monitor the Park's water quality: the Adirondack Lake Assessment Program (www.adkwatershed.org), which monitors lakes based on voluntary cooperation similar to CSLAP, and the Adirondack Long Term Monitoring Program (www.adirondacklakessurvey.org), which monitors only minimally impacted water bodies. Given that road runoff is hydrologically connected to the majority of surface water area in the Park and that road salt represents a major external pollutant load in our region (Kelting and Laxson 2010), our methods may provide a model for a useful study design that can answer real questions regarding statewide water quality.

Efficacy of methods

Subresolution variability is an inherent issue with DEM-based hydrologic modeling (Woods 2006), but for identifying impacted lakes and streams in this study area, the 10 m USGS DEMs are of sufficient resolution to meet this channel-based objective in this topographically variable study area (McMaster 2002). The total impacted area values are more subject to resolution biases. Coarser resolutions produce higher accumulation area values (Yang and Chu 2013), possibly overestimating the area impacted by overland flow. Zhang and Montgomery (1994) analyzed the effects of DEM resolution on geomorphic and hydrologic process in a moderate to steep gradient landscape. They found that 10 m resolution DEMs such as those used in this study should be sufficient to model surface flow processes, while DEMs greater than 30 m resolution produce erroneous estimates of hill slope and runoff processes.

This study quantifies total land area downslope of roads impacted by road runoff, but does not differentiate between surface and subsurface flows within this land area. Surface flow is a combination of direct runoff of salt-laden melt waters through road drainage networks (Labadia and Buttle 1996) and Hortonian flow (de Lima and Singh 2002). Hortonian flow is an important process during spring snow melt when soils are saturated and when frozen soils prevent infiltration of melt waters during thaws (Laudon et al. 2004). Surface flow removes about 50 % of road salt annually, and the remainder accumulates and moves through soils and groundwater as subsurface flow (Meriano et al. 2009), though a greater percentage of road salt may enter subsurface flow in the Adirondacks, as the region is dominated by coarse texture sandy soils with high infiltration rates (Sullivan et al. 2006; Kelting et al. 2012). Groundwater salt concentration increases every year with road salting as only a portion of salt entering in subsurface flow will flush annually (Perera et al. 2012), resulting in higher salt concentrations in streams during summer when biological activity is highest and perhaps most vulnerable to negative effects of salt (Jackson and Jobbagy 2005; Corsi et al. 2015). Given the importance of subsurface flow to road runoff impacts, landscape level modeling estimates would be enhanced by partitioning surface and subsurface flow paths when the additional soil and geologic data necessary to make these estimates exists.

Conclusions

The high degree of hydrologic association between paved roads and the lands and waters of the Park strongly suggests that the environmental impacts of road runoff should be evaluated along with other regional stressors currently being studied. Being able to estimate the spatial extent of road runoff is important for designing monitoring programs that can explicitly monitor this stressor while also providing opportunities to understand the interaction of multiple environmental stressors. Our methods provide an efficient way to estimate the spatial extent of road runoff for large geographic areas.

Acknowledgments Support for this research was provided with a grant from the Northeastern States Research Cooperative (nsrcforest.org).

Appendix

Methods

Below are detailed instructions that accompany each subsection in "Methods".

Producing a hydrologically relevant surface

The DEMs were mosaicked into a single raster dataset using the ArcGIS data management tool "workspace to new raster dataset." Missing cells within the mosaicked dataset were identified using the "Is Null" tool. Identified missing cells were then replaced with interpolated elevation values using a conditional statement in ArcGIS. A low pass filter was used to remove DEM artifacts and noise from the appended DEM (Gesch and Wilson 2001).

Producing road disturbance grids

The eight shapefiles (SF, C, L, and SFCL and 100 m SF, 100 m C, 100 mL, and 100 m SFCL) were converted into 10 m cell size raster datasets using the ArcGIS "polyline to new raster" tool and the "polygon to new raster" tool. The new rasters were reclassified so that roads were set to a value of 1 and non-road cells were set to a value of 0. Reclassification was necessary to create a disturbance grid input parameter for the accumulation tool. The final indicator grids were then converted to TIFF as the accumulation tool requires this file format. The disturbance grid indicates the zone of the area of substance supply (runoff) with 1 representing the zone and 0 representing the rest of the domain.

Producing input weights

The effective runoff and decay multiplier grids were created by reclassifying of one of the disturbance area grids to a constant value of 1.

Delineating overland flow grids

TauDEM's "D-Infinity concentration limited accumulation" functionality applies to a situation where an unlimited supply of a substance is loaded at a constant concentration over the cells of a value of 1 in the disturbance grid. For this paper, road runoff was the substance, and roads were the cells with a value of 1 in the disturbance grid. All inputs were clipped to a standard size to fit the needs of the tool parameters. The outputs were eight overland flow grids representing downhill flow originating from SF, C, L, and SFCL roads. These eight outputs were clipped by the extent of the Adirondack Park boundary polygon (apa.ny.gov/gis) to retain park only overland flow.

Quantifying impacted land area

Multiplying the overland flow dataset by open water cells represented by NoData excluded the original values from the output, and multiplying by 1 retained the original values in the output. The park wide open water raster data was created by clipping a NYS hydrography shapefile of lakes and ponds (gis.ny.gov) to the Adirondack Park boundary shapefile used above and then converting the clipped shapefile into a raster dataset using the ArcGIS "polygon to new raster" tool. The new raster dataset was then reclassified so that open water was represented by NoData and land area represented by a value of 1. The cell counts in the eight no-open-water overland flow grids were found and then multiplied by the cell area (100 m^2) to find the total terrestrial area impacted by road runoff. Values were reported in hectares.

Identifying impacted lakes and ponds

Total surface area (hectares) of these impacted water bodies was calculated using the calculate geometry function.

Calculating downstream length of impacted streams

Point shapefiles were created at their intersections by setting the output type in the "intersect" tool to point. The raster dataset was then reclassified so that cells represent intersection as a value of 1 and non-intersection cells as a value of 0. Input effective runoff and decay grids as well as the original D-Infinity flow direction grid were reused. The eight output flow grids were thinned using the ArcGIS "thin" tool, and flow cells over lakes and ponds were removed. Thinning accumulation rasters efficiently identifies stream channels over large geographic areas (Betz et al. 2010). This approach is aimed at sidestepping the process of using flow accumulation thresholds to identify stream channels across the entire park. We found the intersection of overland flow from roads and linear stream data was sufficient for identifying the presence of a stream. Total lengths (kilometers) of these impacted stream channels were calculated using the calculate geometry function.

Quantifying impacted land and waters in APA forest preserve lands

Forest preserve land is represented by 4 of the 15 land use types, wilderness, primitive, canoe, and wild forest. These four land use types were merged, and impacted surface waters and streams for each of the six disturbance categories within the preserve were calculated by ESRI's Select by Location and calculate geometry functions.

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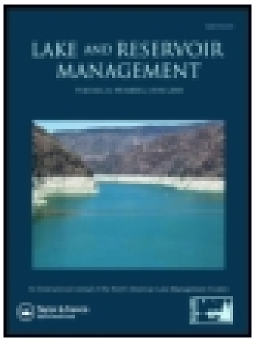

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A reduction in spring mixing due to road salt runoff entering Mirror Lake (Lake Placid, NY)

Brendan Wiltse[a], Elizabeth C. Yerger[b] and Corey L. Laxson[b]

[a]Ausable River Association, Wilmington, New York 12997, USA; [b]Paul Smith's College, Adirondack Watershed Institute, Paul Smiths, New York 12970, USA

ABSTRACT

Wiltse B, Yerger EC, Laxson CL. 2019. A reduction in spring mixing due to road salt runoff entering Mirror Lake (Lake Placid, NY). Lake Reserv Manage. XX:XXX–XXX.

Road salt has resulted in the salinization of surface waters across temperate North America. Increasingly, road salt is recognized as a significant regional pollutant in the Adirondack Park. Here we analyze biweekly limnological data from Mirror Lake (Lake Placid, NY) to understand the role of road salt runoff in an apparent lack of spring mixing in 2017. Water column profile data show notable spatial and temporal variability in chloride concentrations within the lake. Concentrations are highest at the lake bottom during the winter, with increases associated with the onset of road salt application to the watershed. High chloride concentrations in the hypolimnion persisted through the summer of 2017 due to a lack of complete spring mixing as the result of road salt induced density differences within the water column. Water density calculations and Schmidt stability point to an increase in water column stability due to the accumulation of salt at the lake bottom. The incomplete spring mixing resulted in greater spatial and temporal extent of anoxic conditions in the hypolimnion, reducing habitat availability for lake trout. Restoration of lake mixing would occur rapidly upon significant reduction of road salt application to the watershed and improvements in stormwater management.

KEY WORDS
Adirondack Park; Lake Placid; lake trout; Mirror Lake; monomixis; road salt; stratification

Road salt commonly is used across temperate North America to maintain roads free of snow and ice in the winter because of its comparatively low cost and availability (TRB 1991). Most of these are chloride-based salts, primarily sodium chloride, although mixtures of calcium chloride and magnesium chloride also are used commonly (TRB 1991). The use of road salt has been increasing steadily over the past 60 yr. In 2014, 24.2 million metric tons of sodium chloride was applied to roads in the United States (Lilek 2017).

Sodium chloride increasingly is recognized as a significant pollutant in the northern hemisphere. Road runoff has infiltrated both surface water and groundwater, resulting in elevated salinity (Kelting et al. 2012, Dugan et al. 2017, Schuler et al. 2017, Hintz and Relyea 2017a, Kelly et al. 2018, Pieper et al. 2018). Sodium and chloride concentrations in impacted waters have been linked to road density, impervious surface density, and road salt application rates (Kaushal et al. 2005, Kelting et al. 2012). Groundwater with concentrations exceeding US Environmental Protection Agency (EPA) drinking-water guidance values for sodium has been documented in areas receiving runoff from general road application, as well as runoff from salt storage facilities (Kelly et al. 2018, Pieper et al. 2018).

Road salt has the potential to upset ecosystem structure, resulting in undesirable shifts in ecological communities (Hintz et al. 2017). The impact of road salt on aquatic life varies by species, ecosystem, and the type of salt applied (Jones et al. 2015, Hintz and Relyea 2017b). In urban areas, surface waters may exceed the EPA chloride thresholds for chronic and/or acute chloride toxicity to aquatic life (Corsi et al. 2015). However, the EPA

CONTACT Brendan Wiltse brendan@ausableriver.org; wiltseb@gmail.com
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thresholds for acute and chronic toxicity from chloride may not be directly relevant to a particular lake or stream ecosystem. Specifically, taxa in oligotrophic soft-water lakes may be more sensitive to chloride pollution (Elphick et al. 2011, Brown and Yan 2015). The addition of multiple stressors, such as climate change, acid deposition recovery, and eutrophication, further complicates our understanding of how aquatic ecosystems will respond to elevated salinity (Palmer and Yan 2013).

The impact of road salt runoff on the physical limnology of North American lakes is less well known. A few studies have documented meromixis in lakes in urban environments, as a result of salt-induced density gradients (Judd 1970, Novotny et al. 2008, Novotny and Stefan 2012, Sibert et al. 2015, Wyman and Koretsky 2018, Dupuis et al. 2019). The disruption of a physical processes such as lake mixing has indirect effects on both the chemistry and biology of the lake (Judd et al. 2005). Incomplete mixing increases the likelihood of anoxia in the hypolimnion, resulting in the release of phosphorus from sediments, as well as the accumulation of manganese, ferrous iron, sulfide, and methane in the hypolimnion (Wetzel 2001). Prolonged periods of anoxia also can reduce habitat availability for cold stenotherms, such as lake trout (*Salvelinus namaycush*). Lake trout have been declining across their native range in New York State and the state has classified them as a "Species of Greatest Conservation Need" (Carlson et al. 2016). Invasive sea lamprey and climate change are the two major factors affecting lake trout populations in New York State (De Stasio et al. 1996, Schneider et al. 1996). Reductions in habitat availability due to an interruption of lake mixing has the potential to further challenge this species.

Small urban lakes receiving direct stormwater runoff are typically most susceptible to the deleterious effects of road salt pollution (Dupuis et al. 2019, Scott et al. 2019). Mirror Lake in the Village of Lake Placid is the most developed lake in the 6 million acre Adirondack Park. The village is surrounded by New York State Wilderness, resulting in concentrated urban development around Mirror Lake. The goal of this study was to assess the extent to which road salt is responsible for an apparent incomplete spring mixing in Mirror Lake.

Study site

Mirror Lake (44.290°N, 73.982°W) is a small lake in the eastern portion of Essex County in the Adirondack Park of upstate New York. The lake has a surface area of 50 ha, a maximum depth of 18 m, and a volume of 3.42×10^6 m^3. The watershed is 301 ha and is composed of 51% forest, 27% developed, 20% surface water, and 2% wetland. The development is concentrated directly around the lake, with the headwaters of the watershed forested. The lake has one major natural inlet at the north end, but also receives runoff from 22 stormwater outfalls that discharge runoff directly to the lake. The lake outlets at the south end through a culvert that eventually enters the Chubb River. There are 1.1 km of state road and 7.6 km of local road within the watershed. The lake shore is entirely encompassed by road, a combination of both state and local. State roads in the Adirondack Park receive 3 times higher salt loads than local roads (Kelting et al. 2012). Within the watershed there are 28.0 ha of impervious surfaces (9% of watershed area) that may be treated with road salt, with the largest being parking lots (8.3 ha), followed by village roads (5.9 ha), private roads (3.7 ha), driveways (3.5 ha), town roads (2.5 ha), village sidewalks (2.0 ha), state roads (1.2 ha), and private sidewalks (0.9 ha) (Wiltse et al. 2018).

The watershed bedrock material is made up of granite gneiss, migmatite, and olivine metagabbro (Caldwell and Pair 1991). These are common parent materials in the Adirondack uplands and contain very little chloride; as a result, natural chloride concentrations in least impacted Adirondack lakes seldom exceed 1 mg/L (median = 0.24 mg/L; Kelting et al. 2012). The bedrock is overlaid by till and till moraine in much of the watershed, with exposed bedrock in the upper portions of the watershed. Soils consist of the Hermon series directly around the lake and Becket series in the rest of the watershed. These soils are well drained or somewhat excessively drained and commonly occur on glacial till.

The fish community consists of rainbow trout (*Oncorhynchus mykiss*), lake trout (*Salvelinus namaycush*), white sucker (*Catostomus commersoni*), smallmouth bass (*Micropterus dolomieu*),

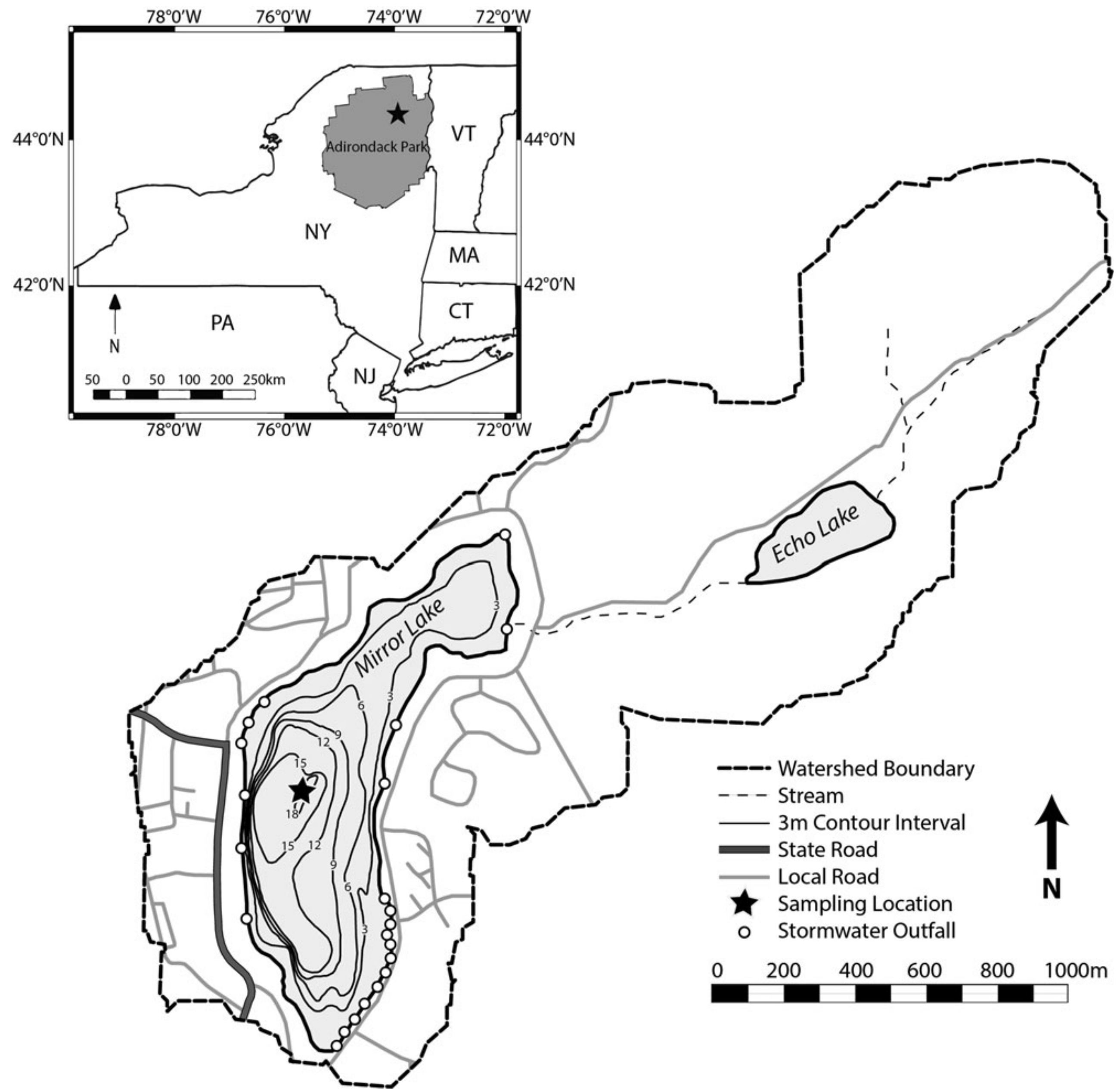

Figure 1. Watershed map of Mirror Lake, including lake bathymetry.

rock bass (*Ambloplites rupestris*), pumpkin seed (*Lepomis gibbosus*), yellow perch (*Perca flavescens*), and brown bullhead (*Ameiurus nebulosus*) (NYSDEC unpublished data). The New York State Department of Environmental Conservation regularly stocks the lake with rainbow trout and lake trout (NYSDEC 2018).

Methods

Mirror Lake was sampled biweekly from 5 December 2015 to 4 January 2018, at the point of maximum depth (18 m, Figure 1). During the period of ice cover, water sampling was conducted monthly by auguring a hole through the ice. Sampling was resumed as soon as possible after ice-out to adequately capture spring mixing. In 2016, sampling resumed 7 d after ice-off, and in 2017 sampling resumed the day of ice-off. Likewise, sampling continued until the formation of ice, to capture the full extent of fall mixing. A YSI Professional Plus multiparameter sonde was used to collected in situ dissolved oxygen, specific conductance, temperature, and pH measurements at 1 m intervals. Water samples were collected at the surface using a 2 m integrated tube sampler and 1.5 m off the bottom (16 m depth) using a 1.2 L stainless steel Kemmerer sampler. Water samples were transferred immediately to acid-washed and field-rinsed sample bottles and transported on ice to the Paul Smith's College Adirondack Watershed Institute for analysis. In the laboratory, samples were passed through a 0.45 µm nylon filter to remove suspended material and were frozen at −30 C until analysis (no more than 28 d). Chloride concentration was determined with chemically suppressed ion chromatography (Lachat Instruments, QC8500, Loveland, CO) following EPA Method 300.1. Quality control measures such as blanks, duplicate samples, and laboratory control samples were analyzed at a frequency of one per batch of 20 or fewer samples and assessed on an ongoing basis. The practical quantitation level of chloride during this study was 0.2 mg/L, and the percent recovery on laboratory control samples (10 mg/L) was 99.8%.

Linear regression was used to develop a lake-specific relationship between specific conductance and chloride. Linear model assumptions were quantitatively assessed using the Global Validation of Linear Models Assumptions (GVLMA) package in R (Pena and Slate 2014, R Core Team 2018). This relationship was used to estimate the concentration of chloride at 1 m intervals through the water column.

Schmidt stability was used to assess changes in the resistance to mixing over the period of the study. Schmidt stability is a measure of the amount of energy required, per unit surface area, to mix the lake to uniform density (Wetzel 2001). Schmidt stability was calculated using a modified version of the rLakeAnalyzer package in R (Winslow et al. 2017). The schmidt.stability() function was modified to use water density equations developed by Chen and Millero (1986). Temperature and practical salinity units (PSU) are used to calculate density. The resulting precision is better than 2×10^{-6} g/cm^3, an improvement over the density calculations used by default in the rLakeAnalyzer package (Millero and Poisson 1981, Martin and McCutcheon 1998). PSU is derived from conductance and temperature measured by the YSI Professional Plus hand-held sonde used to collect water-column profiles (UNESCO 1981). Bathymetric cross-sectional areas were calculated using data provided by the New York State Department of Environmental Conservation (NYSDEC).

Lake trout preferred habitat was estimated using an upper temperature limit of 15 C and lower dissolved oxygen limit of 6 mg/L (Plumb and Blanchfield 2009). The upper and lower limits within the water column were plotted for a visual assessment of shifts in available habitat, and total habitat volume was calculated using bathymetric data.

Results

A linear regression run on all of the paired conductivity–chloride measurements failed to meet the assumptions of a linear regression due to the binomial nature of the data, violating the assumption of a continuous dependent variable. Surface samples had a mean chloride concentration of 44 mg/L, while bottom water samples had a mean of 65 mg/L. To meet the assumptions of a linear regression the 118-sample data set was down sampled to 47 samples to provide a continuous gradient of the dependent variable (chloride). Down sampling was conducted by binning the specific conductance data from 150 to 450 μS/cm in 50 μS/cm increments and then randomly resampling the original data set to obtain a reduced data set with nearly equal frequency across all bins. The data set was reduced sequentially until the resulting regression met the assumption of a continuous dependent variable as evaluated using the GVLMA package in R. The regression results were similar before and after the down sampling ($y = 0.23x - 4.83$, $R^2 = 0.91$, $P < 0.001$; $y = 0.23x - 2.72$, $R^2 = 0.93$, $P < 0.001$, respectively), an indication of the robustness of the relationship and lack of sensitivity to the down sampling (Figure 2). The final regression equation is slightly more conservative in its estimate of chloride.

Temperature at the lake surface ranged from 0.4 C to 24 C, and the lake bottom ranged from 3.1 C to 6.6 C. The lake exhibited a clear pattern of thermal stratification during the summer of

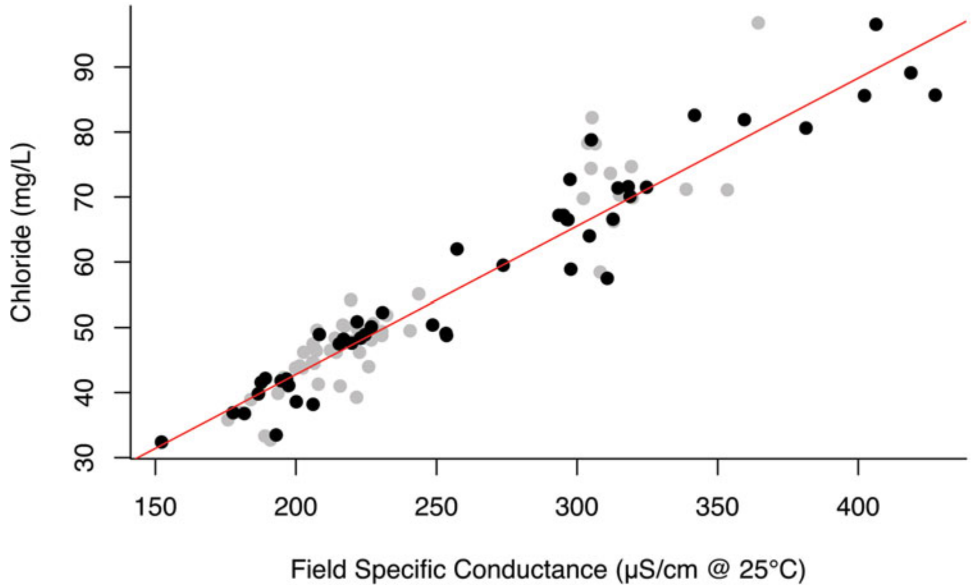

Figure 2. Relationship between conductivity and chloride in Mirror Lake (Lake Placid, NY). $y = 0.23x - 2.72$, $R^2 = 0.93$, $P < 0.001$. Black points are down sampled data; gray points represent data removed through the down sampling process.

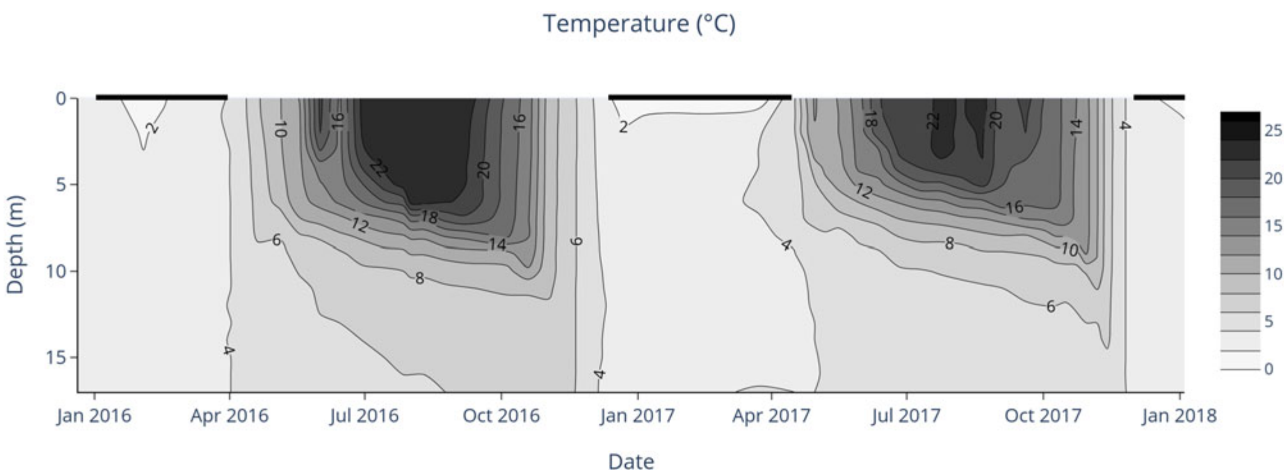

Figure 3. Temperature measured in the Mirror Lake water column. Black bars at the top of the figure indicate periods of ice cover.

both 2016 and 2017, and the lake became isothermal during the spring and fall of each year. The onset of thermal stratification occurred at least 12 d earlier in 2017 than in 2016 (Figure 3).

The spatial and temporal distribution of chloride in the Mirror Lake varied considerably between 2016 and 2017, with concentrations ranging from 19 to 123 mg/L (Figure 4). In both years, concentrations near the lake bottom increased during the winter while salt was being applied to the watershed. In the spring of 2016, concentrations became uniform during spring mixing and remained relatively uniform throughout summer stratification. In the spring of 2017, high concentrations of chloride persisted near the bottom through spring mixing and remained high through summer stratification. Chloride concentrations were uniform during fall mixing in both years. Concentrations at the surface remained fairly constant through 2016 but declined during summer stratification in 2017.

During the winter months, the water column density gradient was driven by elevated salinity near the lake bottom (Figures 4 and 5). The timing and increase in density corresponded with the timing and increase in chloride (Figures 4 and 5). During the summer, the water column density gradient was dominated by temperature. In 2017, when chloride remained high in the

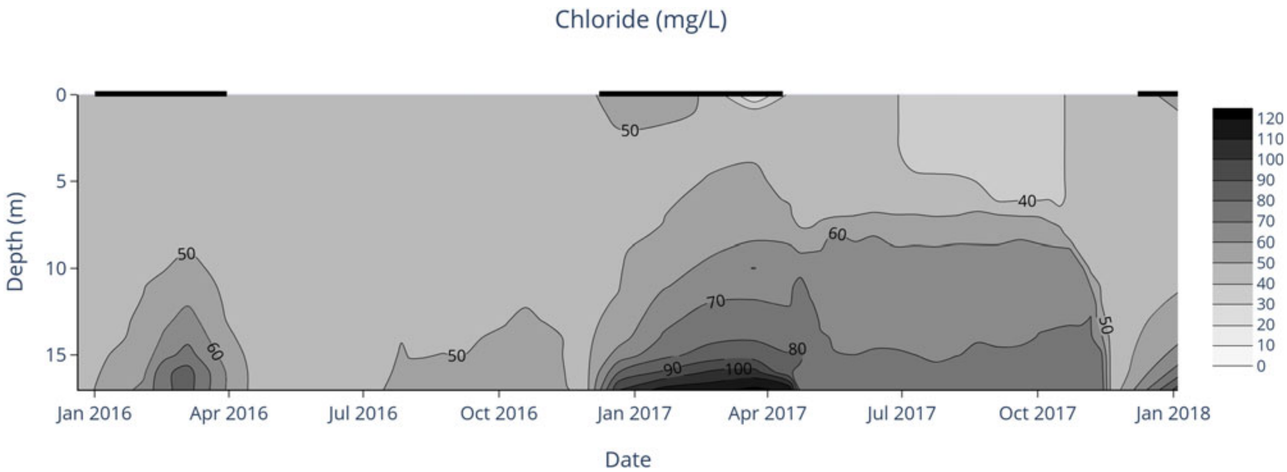

Figure 4. Distribution of chloride in Mirror Lake, modeled from conductivity. Black bars at the top of the figure indicate periods of ice cover.

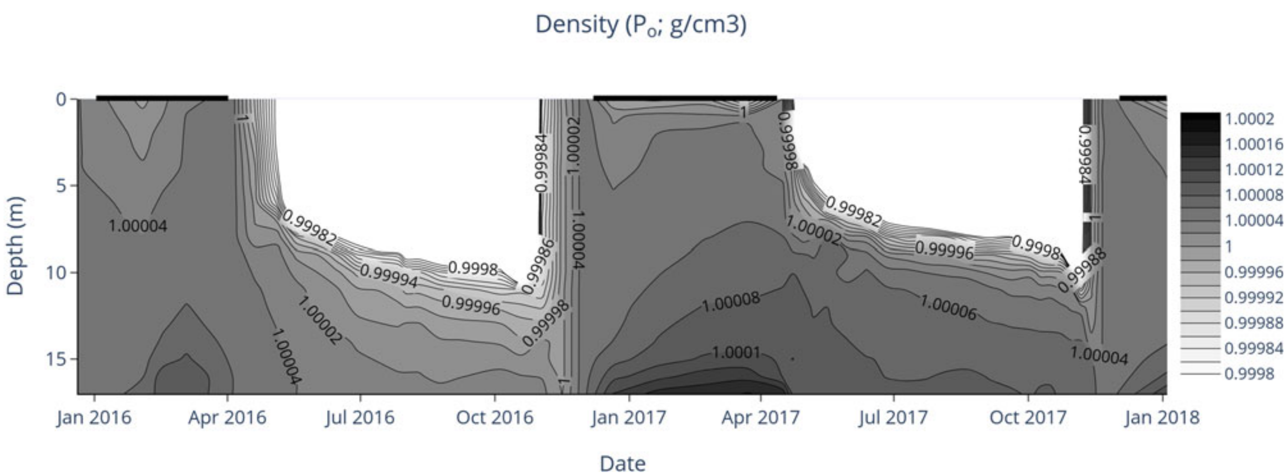

Figure 5. Water density calculated based on temperature and practical salinity units (Chen and Millero 1986). Densities below 0.99980 g/cm^3 are not plotted, for ease of plot interpretation. Black bars at the top of the figure indicate periods of ice cover.

hypolimnion, water density was higher than during the same period in 2016. This is explained partially by a 1 C difference in hypolimnion temperature as well, with hypolimnion temperatures being cooler in 2017.

The hypolimnion was anoxic during summer stratification in both years (Figure 6). The duration and spatial extent of the anoxic conditions were greater in 2017. Anoxic conditions were present at the lake bottom during the winter in 2017, but not in 2016. In both years, a positive heterograde oxygen curve was persistent throughout summer stratification, with the peak dissolved oxygen concentrations occurring at or near the thermocline (Figure 6 and Table 1). Hypolimnetic dissolved oxygen concentrations increased during the spring and fall mixing in 2016 and the fall mixing in 2017. Concentrations increased during spring mixing in 2017, but remained below 6 mg/L (Figure 6).

Temperature was the predominant driver in changes in lake stability (Figure 7). Lake stability dropped by one or more orders of magnitude during the periods of mixing, except during the spring of 2017. Stability during the winter of 2017 was higher than for 2016 because of the salinity-induced water density gradient (Figures 5 and 7). The additional stability in the spring of 2017 because a buildup of salt at the lake bottom increased the stability of the lake to the equivalent of a 3 C increase in surface water temperature.

Lake trout habitat was restricted in both 2016 and 2017 by warm surface water and anoxic conditions in the hypolimnion (Figure 8). Overall habitat volume was similar in both years, showing a marked decline in available habitat once

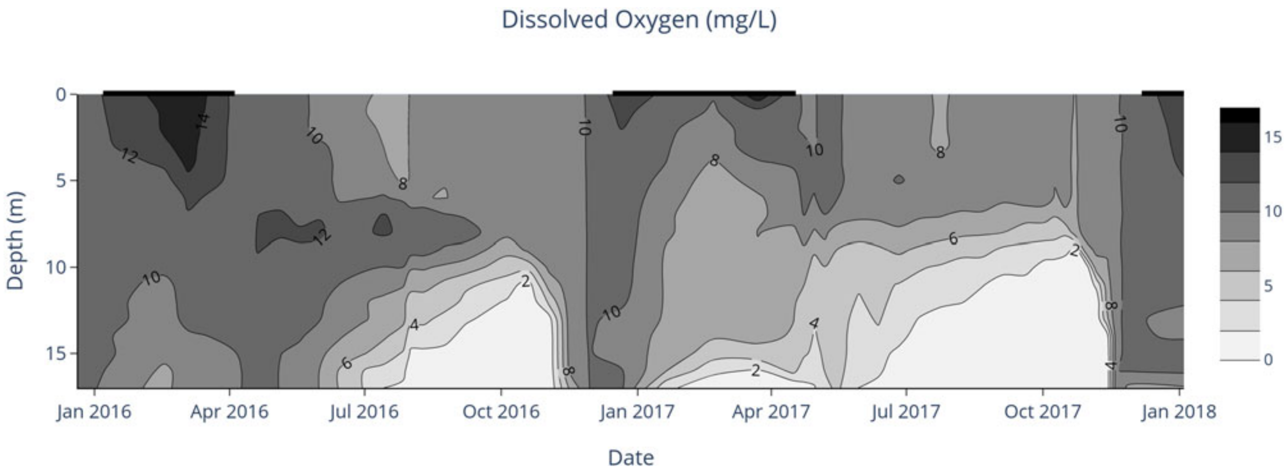

Figure 6. Dissolved oxygen concentrations measured in the Mirror Lake water column. Black bars at the top of the figure indicate periods of ice cover.

Table 1. Historical dissolved oxygen and specific conductance data from previous surveys of Mirror Lake (Lake Placid, NY).

Depth (m)	7 Sep 1954[a]	7 Jul 1967[a]	11 Aug 1971[b]	5 Aug 1974[c]	10 Aug 2001[d]	16 Jun 2003[a]	10 Aug 2016[e]	7 Aug 2017[e]
Dissolved oxygen (mg/L)								
0		10.0	8.7	8.4	8.1		8.1	8.2
1					7.9		8.2	8.3
2					8.0		8.2	8.3
3		8.4			8.1		8.2	8.2
4					8.5		8.2	8.2
5				8.9	10.4		8.2	9.7
6		9.4			12.0		8.0	9.6
7					11.7		11.5	7.5
8			11.2		10.9		11.4	6.4
9	7.8	10.2			7.3		10.9	4.7
10			6.9	7.0	4.0		8.2	3.2
11					3.0		6.9	2.5
12	2.8	8.2			2.0	7.0	4.9	2.1
13					1.1		4.6	0.8
14					0.2		2.8	0.3
15	2.4				0.0		1.7	0.2
16		5.2		0.05	0.0		0.4	0.2
17					0.0	2.0	0.3	0.2
18			1.4					
Specific conductance (μS/cm)								
0				82	158		226	178
1					158		226	178
2					158	216	226	178
3					157		226	178
4					157		226	178
5				88	176		225	198
6					195		225	211
7					217		219	228
8					230		220	252
9					253		221	274
10				92	269		221	285
11					275		222	289
12					279	329	224	291
13					281		226	295
14					285		228	298
15					289		230	301
16				98	297		232	318
17					308		233	326
18								

[a]NYSDEC unpublished data.
[b]Oglesby (1971).
[c]Oglesby and Miller (1974).
[d]UFI (2001).
[e]Current study.

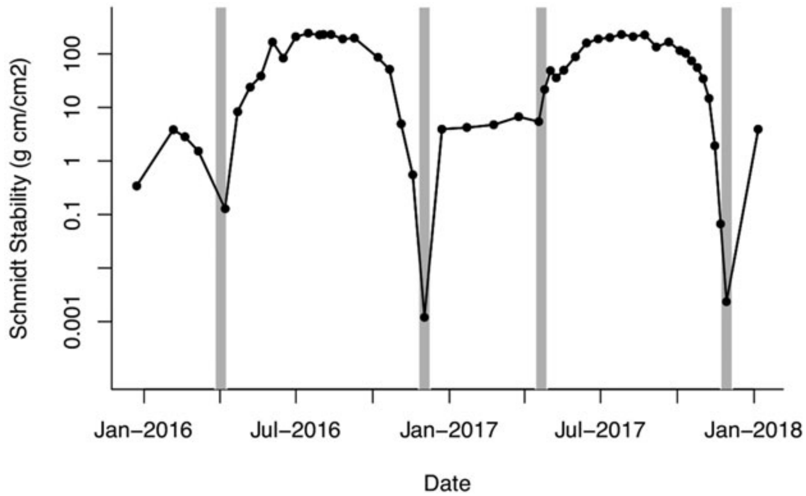

Figure 7. Schmidt stability for Mirror Lake. Gray bars represent expected periods of mixing based on the presence of nearly uniform temperature in the water column.

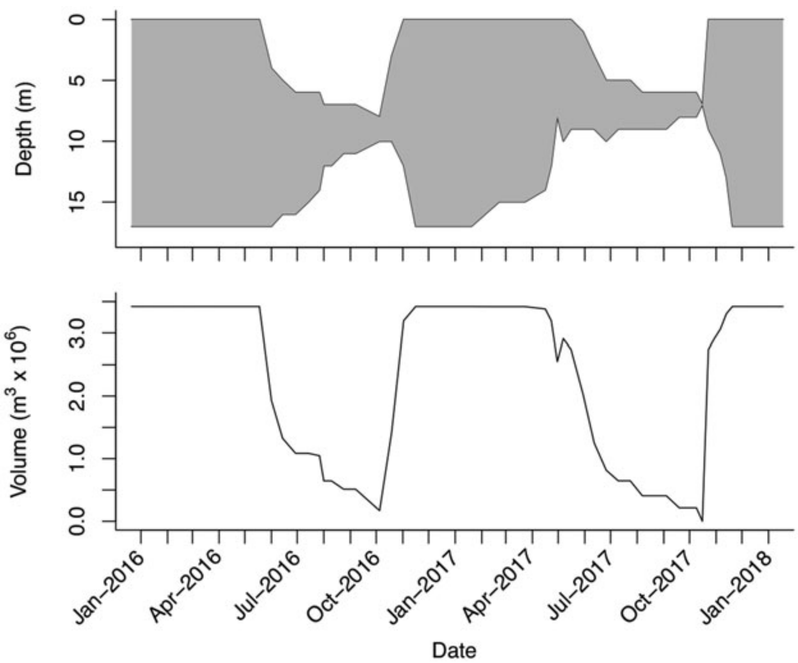

Figure 8. Lake trout habitat area in Mirror Lake (gray shade) as a cross section of the water column in 2016 and 2017 (top panel). Total habitat volume in 2016 and 2017, calculated from habitat suitability in the water column and bathymetric data (bottom panel). Habitat suitability was based on water temperatures less than 15 C and dissolved oxygen concentrations greater than 6 mg/L (Plumb and Blanchfield 2009).

stratification developed. An exception occurred in 2017 when preferred habitat volume briefly reduced to 0 m^3. The greater spatial and temporal extent of anoxia in 2017 was offset by cooler surface water temperatures and a shallower thermocline.

Discussion

Northern temperate lakes of sufficient depth are typically dimictic, turning over twice a year, once in the fall and once in the spring (Wetzel 2001). This is a process critical to the health of a lake, especially one that supports cold stenotherms such as lake trout and rainbow trout. Mirror Lake completely mixed seasonally as expected in all but the spring of 2017. During this period hypolimnetic chloride was much higher than epilimnetic concentrations, resulting in distinct salinity-driven density differences even though the water column was isothermal (Figures 3–5). The

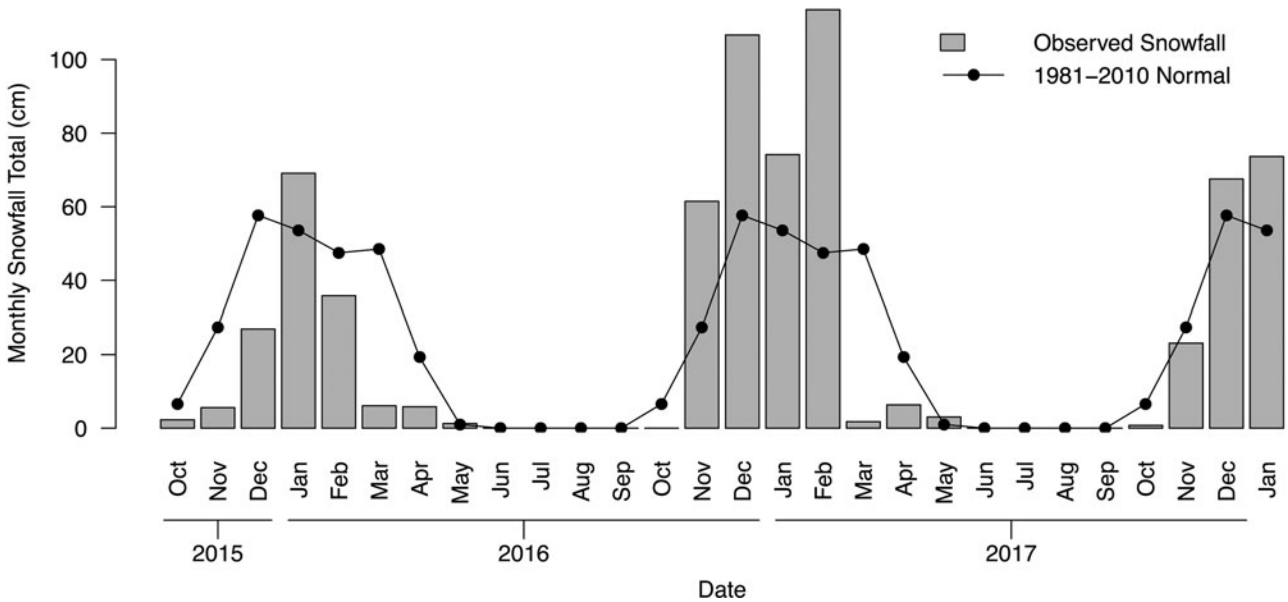

Figure 9. Monthly snowfall in Lake Placid, NY from October 2015 through January 2018. Data pulled from USHCN (Station ID: US1NYES0001).

density gradient due to salinity in the spring of 2017 caused the lake to maintain enough stability to prevent complete mixing of the water column (Figure 7). The persistence of elevated chloride (Figure 4) and low dissolved oxygen (Figure 6) at the lake bottom before and after spring mixing further support this observation.

Twenty-two stormwater outfalls discharge directly into Mirror Lake (Figure 1). These outfalls drain the urban landscape around the lake, including the state road, the village sidewalks, and many parking lots found within the Village of Lake Placid. Discrete sampling of the outfalls during the winter frequently yielded chloride concentrations entering the lake ranging from 500 to 2500 mg/L (Wiltse et al. 2017). These concentrations are 2500 to 12,500 times greater than runoff from least impacted streams in the Adirondack Park (Kelting et al. 2012). The resulting densities for this stormwater range from 1.00085 to 1.00435 g/cm^3, higher than the average density of the lake water at this time (~1.00000 g/cm^3). The highest concentration and volume of this runoff enter the lake along its western shore, where a steep gradient exists along the lake bottom (Figure 1). The timing of increased chloride concentrations at the lake bottom, density of stormwater, and lake bathymetry together point to a mechanism by which the higher salinity stormwater flows along the lake bottom and accumulates at depth.

Spring mixing in 2016, but not 2017, can be explained by the difference in accumulation of salt at the lake bottom (Figure 4). The winter of 2015–2016 was mild, total snowfall was 152.9 cm, 58% of the 1981–2010 normal of 261.6 cm, and 42% of the 2016–2017 total of 367.0 cm (Figure 9). In the nearby (98 km SSE) Lake George, NY, watershed, a 30–40% reduction in road salt application was observed during the 2015–2016 season (Sutherland et al. 2018). Additionally, several rain events occurred during this winter, diluting stormwater chloride concentrations. With lower chloride in the hypolimnion, lake stability was lower during spring mixing in 2016 as compared to 2017 (Figure 7). Also, ice-off in 2016 was the third earliest in the 114-year ice-record, occurring 25 d earlier than the mean ice-off date (Wiltse and Stager 2018). As a result, the period between ice-off and the establishment of the thermocline in 2016 was 39 d, compared to 7 d in 2017. Therefore, the combination of a lower salt load, dilution from rain events, and prolonged spring mixing allowed for the lake to completely mix in the spring of 2016.

The incomplete mixing in the spring of 2017 resulted in a longer duration and spatial extent of anoxic conditions in the hypolimnion, which have consequences for both the chemistry and biology of the lake. The lack of oxygen in the hypolimnion shifts the redox chemistry, causing mobility of phosphorus, manganese, iron, and other ions from the sediments and particulates in the water column. This has the potential to lead to toxic concentrations of trace metals, sulfide, and ammonia in the hypolimnion, combined with anoxia, which can result in declines in biodiversity as well as fish kills (Wetzel 2001, Koretsky et al. 2012). Prolonged periods of anoxia also may lead to increased internal phosphorus loading in the lake (Wetzel 2001). Precambrian shield lakes, such as Mirror Lake, typically have low rates of internal phosphorus loading, but also tend to be phosphorus limited (Orihel et al. 2017). Therefore, increases in internal phosphorus loading of these lakes can be ecologically significant (Healey and Hendzel 1980, Lean and Pick 1981).

Lake trout are the only native cold-water lake-dwelling top predatory fish in the Adirondack Park (Carlson et al. 2016). Narrow habitat requirements (cold, well-oxygenated water), slow growth, and late sexual maturity make lake trout particularly vulnerable to climate change, eutrophication, and other stressors. In small upland lakes, like Mirror Lake, lake trout habitat is likely to decline with increasing temperatures (De Stasio et al. 1996). Mirror Lake has demonstrated a significant increase in the ice-free period over the past 114 years, with the lake currently experiencing an average of 24 more days being ice-free than in the early 1900s (Wiltse and Stager 2018). Climate-driven effects on the lake trout population in Mirror Lake will be further exacerbated by the lack of complete mixing in the spring.

A simple model of preferred lake trout habitat reveals a constriction in habitat in 2017 when the lake did not completely mix in the spring (Figure 8). It is important to note that conditions outside of the thresholds used here may not be lethal to lake trout. Outside of these values it is likely that the fish are experiencing greater physiological stress, which can have impacts on growth, reproduction, and long-term populations dynamics. Juvenile lake trout appear to have less tolerance for conditions outside of this preferred range of temperature and dissolved oxygen, which could influence recruitment into the population (Evans 2005).

Interannual variability in thermocline depth and long-term changes in lake thermal structure also will play an important role in the volume of suitable habitat. This is evident in the comparison of overall habitat volume between 2016 and 2017. Cooler temperatures and a shallow thermocline in 2017 shifted the suitable habitat up in the water column, providing habitat volumes comparable to those of 2016, despite the lack of mixing. Notably, late-season habitat volume in 2017 dropped to 0 m^3 as fall mixing caused a temporary increase in water temperature at the thermocline and the anoxic zone continued to expand (Figure 8). Conditions in the lake did not become lethal to lake trout during this period, and no fish kill was noted, but greater physiological stress was likely.

The small amount of historical data for Mirror Lake suggests that incomplete mixing is not a natural occurrence for the lake. NYSDEC provided limited data from 7 September 1954 and 7 July 1967. These data show dissolved oxygen concentrations higher than at comparable times of year (2.4 mg/L at 15 m and 5.2 mg/L at 17 m, respectively) in our current data (Table 1, NYSDEC unpublished data). This suggests that Mirror Lake was experiencing greater mixing and/or decreased respiration at depth during those years. Data from NYSDEC on 16 June 2003 show dissolved oxygen concentration at 17 m was 2.0 mg/L, while specific conductance was 216 μS/cm at 2 m and 329 μS/cm at 12 m (Table 1, NYSDEC unpublished data). These data are consistent with our observations during the spring of 2017, when the lake experienced incomplete mixing.

Other historical data suggest that incomplete mixing is a more recent phenomenon for the lake. Surveys of the lake in August 1971 and August 1974 generally found hypolimnetic dissolved oxygen concentrations and specific conductance consistent with greater mixing (Table 1, Oglesby 1971, Oglesby and Mills 1974). Despite the low dissolved oxygen in 1974, specific conductance measurements from that day show a 16 μS/cm difference from the surface to the bottom, indicating that there was little or no chemical stratification. A survey on 10 August 2001 found

0 mg/L of dissolved oxygen at 15 m and a difference in specific conductance from surface to bottom of 150.5 μS/cm, consistent with a lack of spring mixing (Table 1, UFI 2002).

Current and historical data indicate that anoxic conditions are likely normal for Mirror Lake. Natural anoxia is common in relatively shallow lakes where the volume of the hypolimnion is small in relation to the sediment surface area (Fulthorpe and Paloheimo 1985). The lack of spring mixing in Mirror Lake has resulted in an increase in the spatial and temporal extent of the anoxic zone in the hypolimnion, which has the potential to threaten the ability of the lake to support lake trout. Further research should be conducted to study the lake trout population and assess the impact of incomplete mixing. The lake currently is stocked with an average of 450 6–7 inch lake trout annually, but no data are available to assess how long these fish survive in the lake and whether there is natural recruitment in the population.

When the interruption in lake mixing began is difficult to establish. The data available in 2001 and 2003 show elevated conductivity in the hypolimnion during the summer, which is consistent with an accumulation of sodium and chloride, and incomplete spring mixing (Table 1). It is possible that this has been an annual or semiannual occurrence at least since that time period. Unfortunately, the paucity of preimpact data and limited continued monitoring of the lake make it impossible to determine when the lake first experienced incomplete mixing and how frequently it occurs.

We believe that restoration of the lake to dimictic conditions would occur during the spring following a substantial reduction in salt load. The lake has demonstrated an ability to overcome the salt-induced density gradient during fall mixing, and without the establishment of a new gradient the following winter, the lake would mix in the spring. The winter of 2015–2016 offers a glimpse at the restoration potential for the lake. Following a winter with less snowfall and a lower salt load, the lake completely mixed in the spring. Substantial reductions in the application of road salt within the watershed, coupled with stormwater improvements to retain stormwater runoff before entering the lake, would provide marked improvements in the water quality of Mirror Lake.

Funding

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Regional pollution of groundwater by road salt in the Adirondack Park

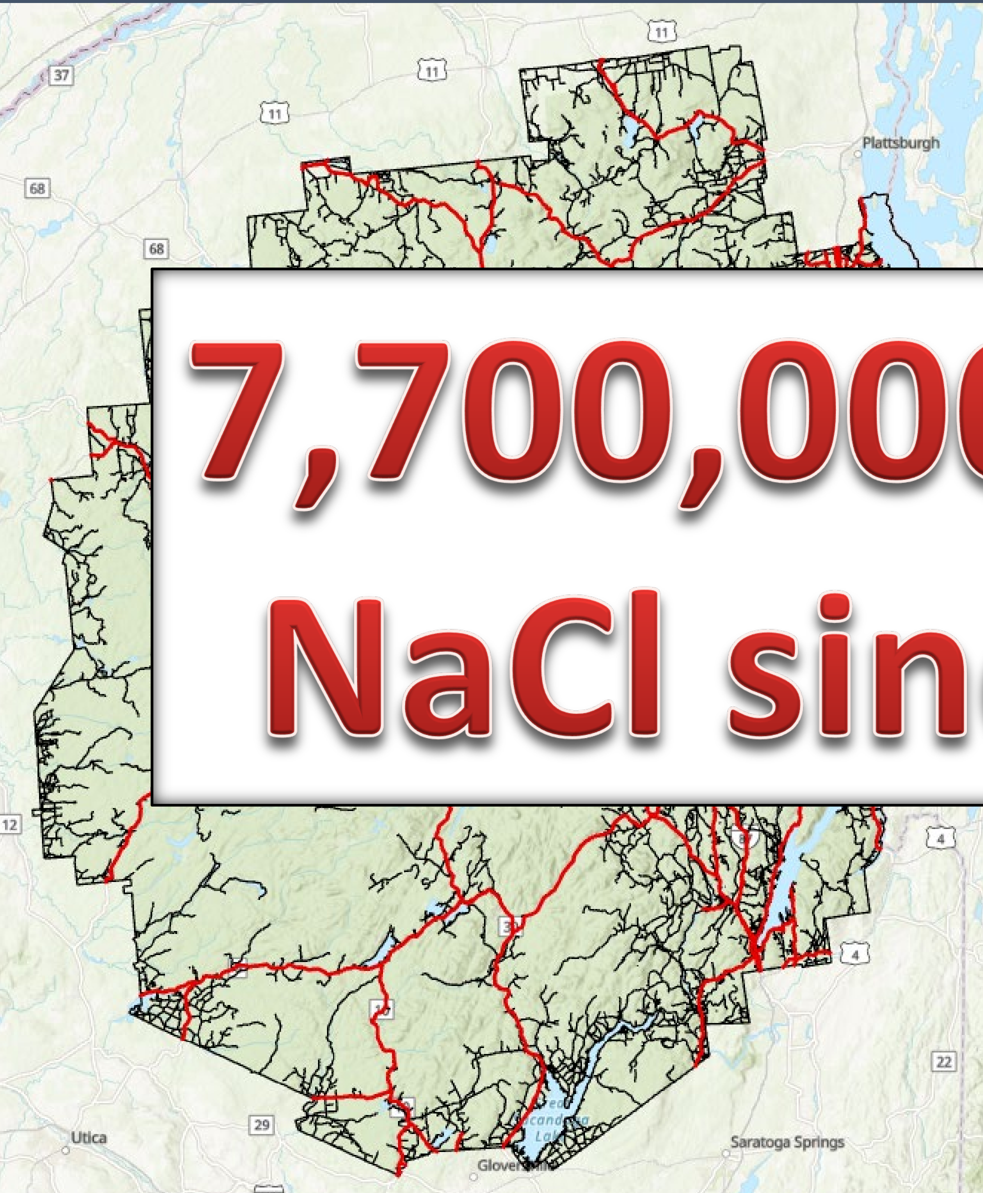
Dan Kelting

Executive Director

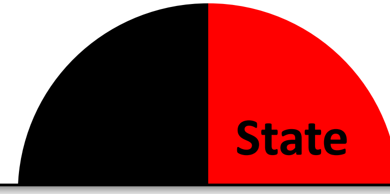
(dkelting@paulsmiths.edu)

PAUL SMITH'S COLLEGE
ADIRONDACK WATERSHED INSTITUTE

Road Salt (NaCl) Use in the Adirondacks



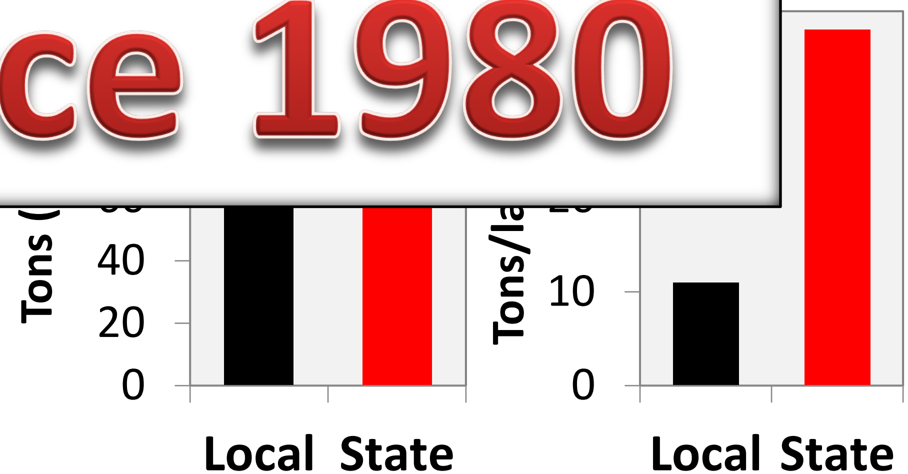
10,555 lane-miles of paved roads



- 2,830 lane-mile
  State & US highways
  Interstate 87

iles

ons



7,700,000 tons of NaCl since 1980

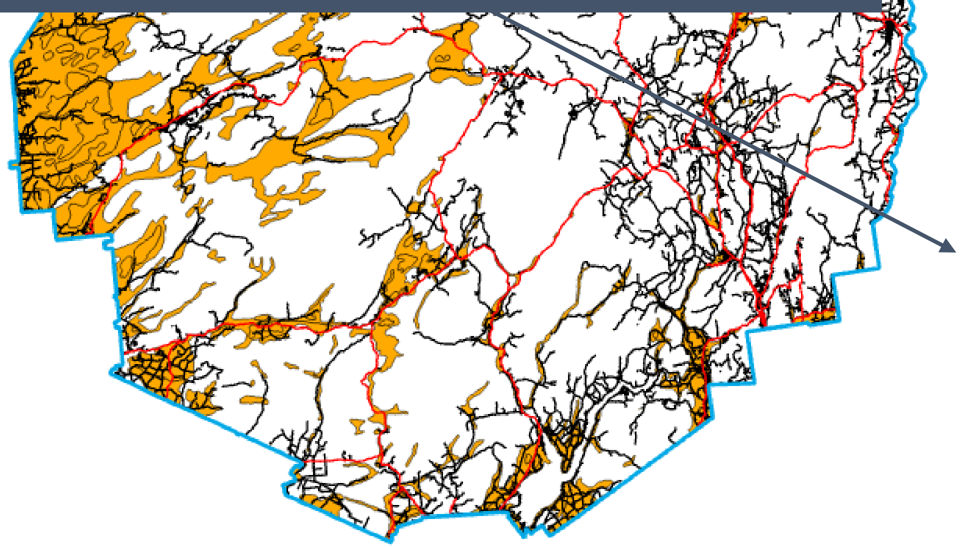
- **State uses 2.5× more salt per lane-mile**

Groundwater Contamination by Road Salt?

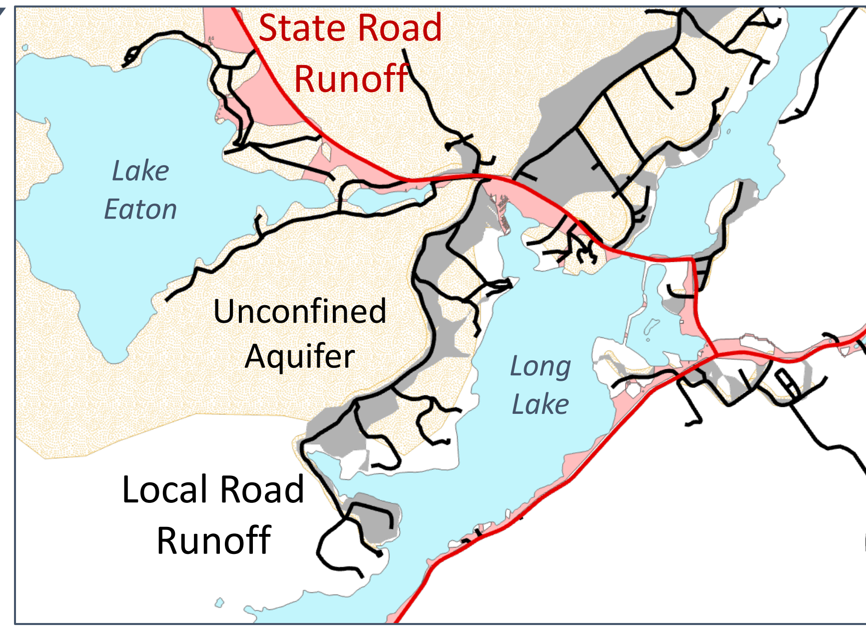
Regional Groundwater Contamination?

Impacts:

- **Human Health**
- **Homeowner Expenses**
- **Property Values**



- 1,600 square miles of **unconfined aquifers**[1], many receiving **runoff from paved roads**



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[1]NYSAPA, 2016

Groundwater Survey Approach

Solicitation
1. Cold calls to database
2. Emails to contact lists
3. Flyers in mailboxes

THE FUND *for* LAKE GEORGE

Online Survey
1. Mailing & well address
2. What year was your well installed?
3. How deep is your well?
4. How was your well constructed?
5. What kind of formation does your well tap into?
6. What water treatment systems do you have?
7. How would you describe your well's water quality?

Hamilton College

AdkAction

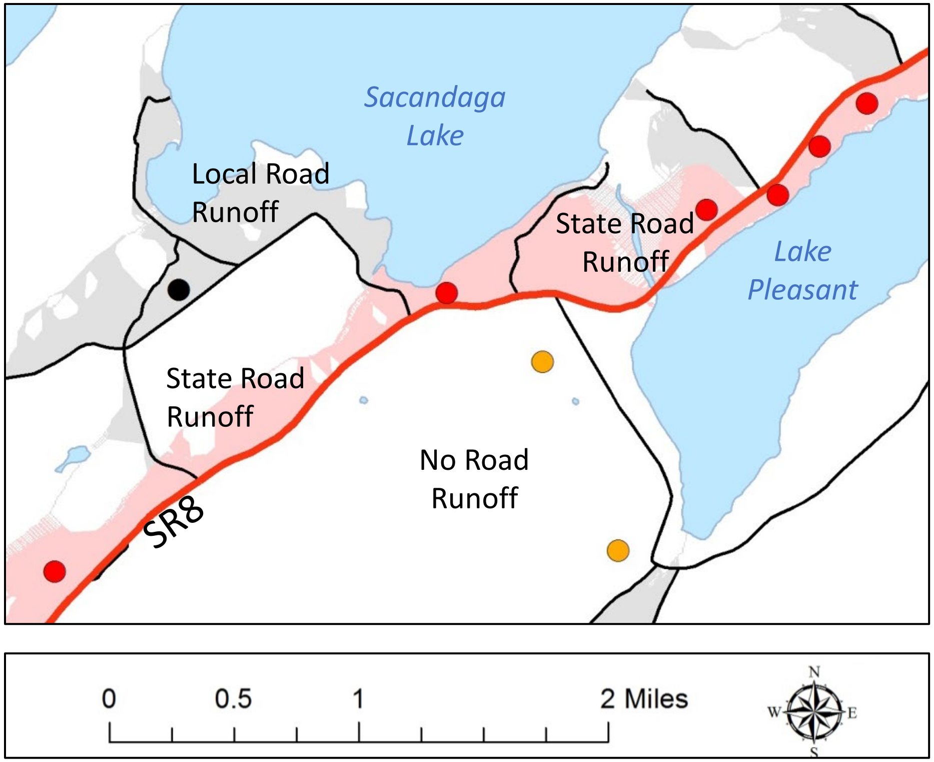
PAUL SMITH'S COLLEGE
ADIRONDACK WATERSHED INSTITUTE

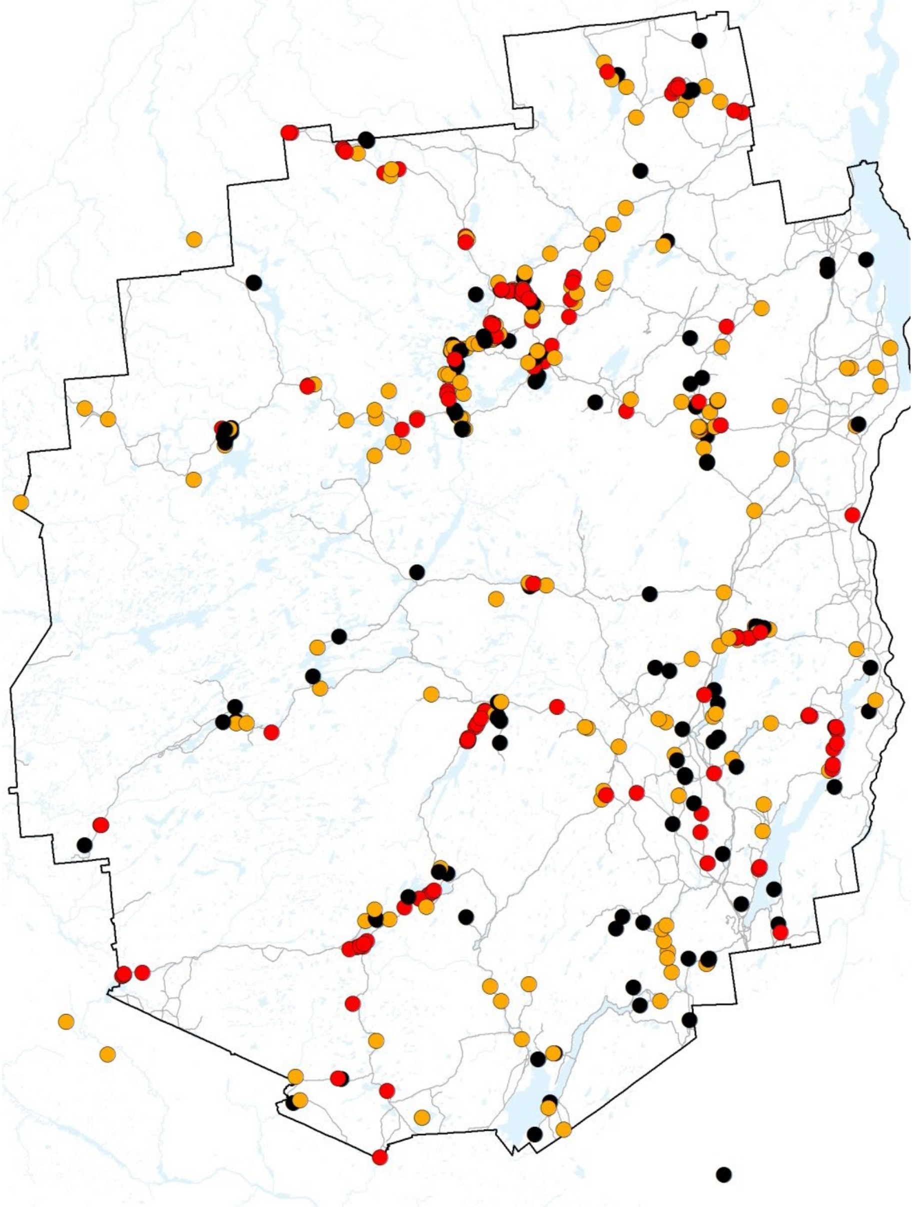
Sampling Kit
1. Prepaid return mailer
2. Sample bottle
3. Gloves
4. Directions

Private Drinking Water Well Study

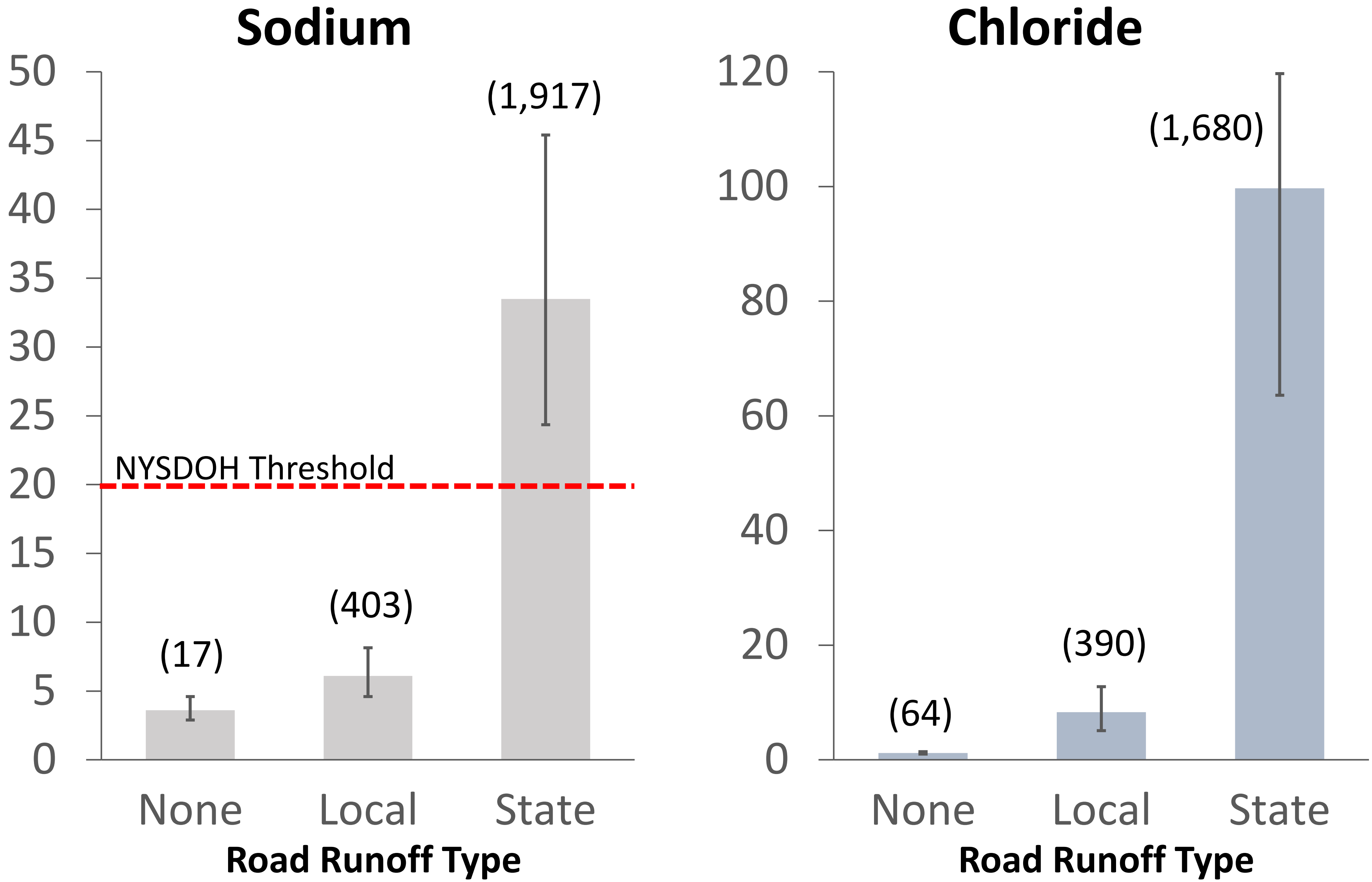
529 private wells

- 213 no road runoff = None
- 143 local road runoff = Local
- 173 state road runoff = State

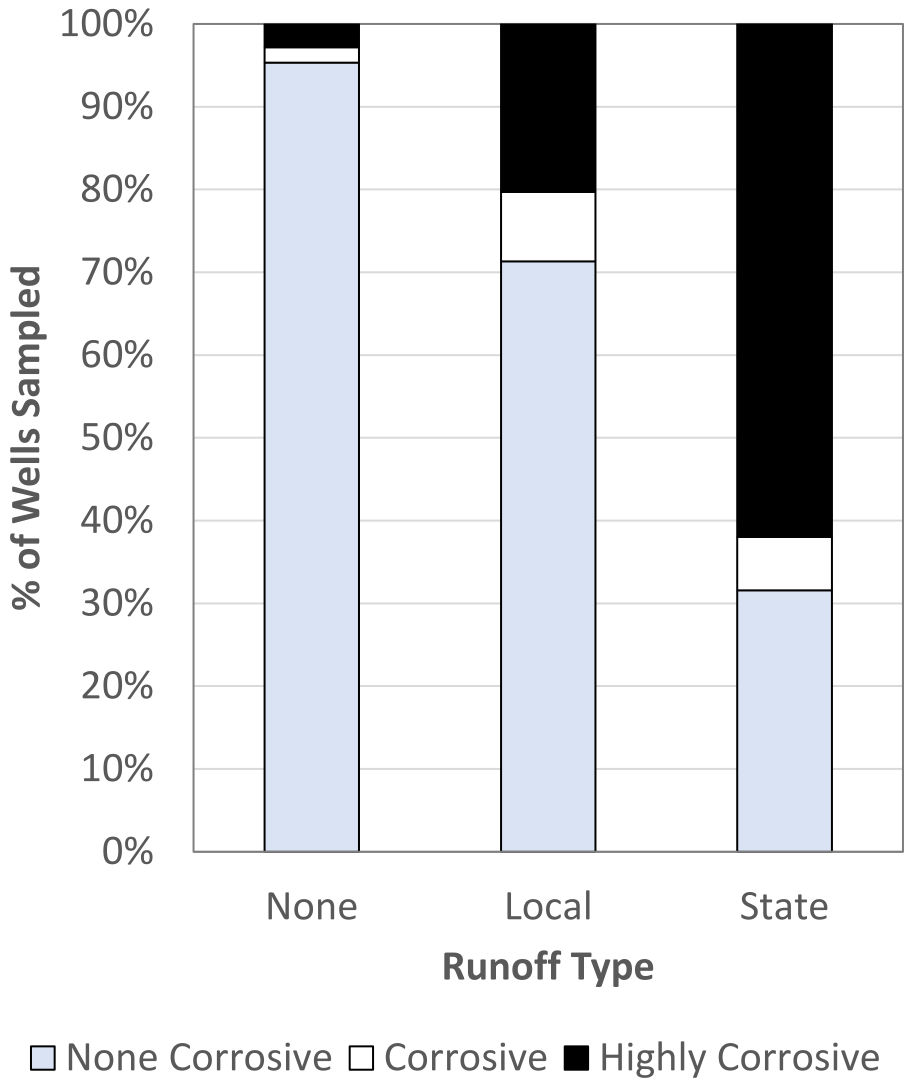




Median Sodium & Chloride (mg/L)

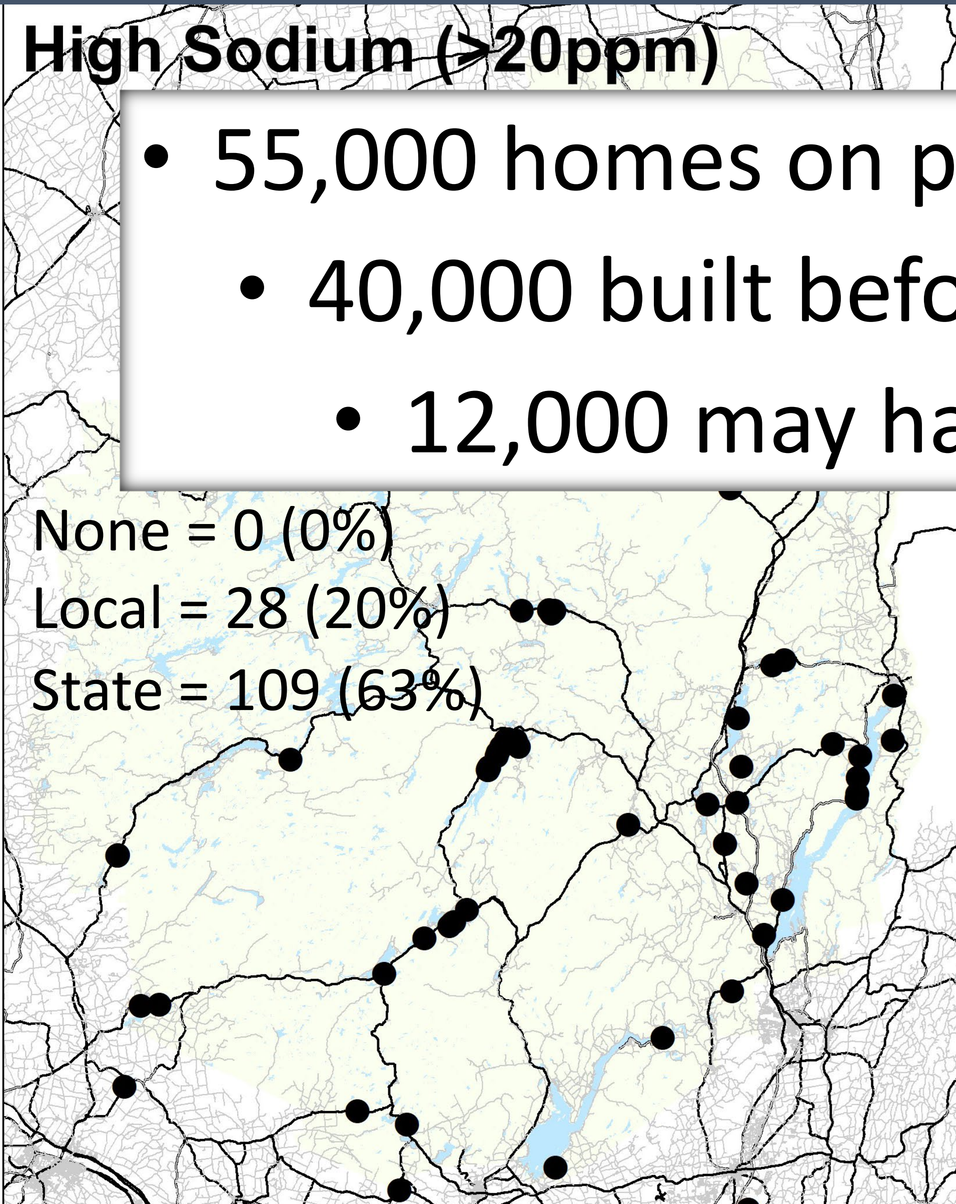


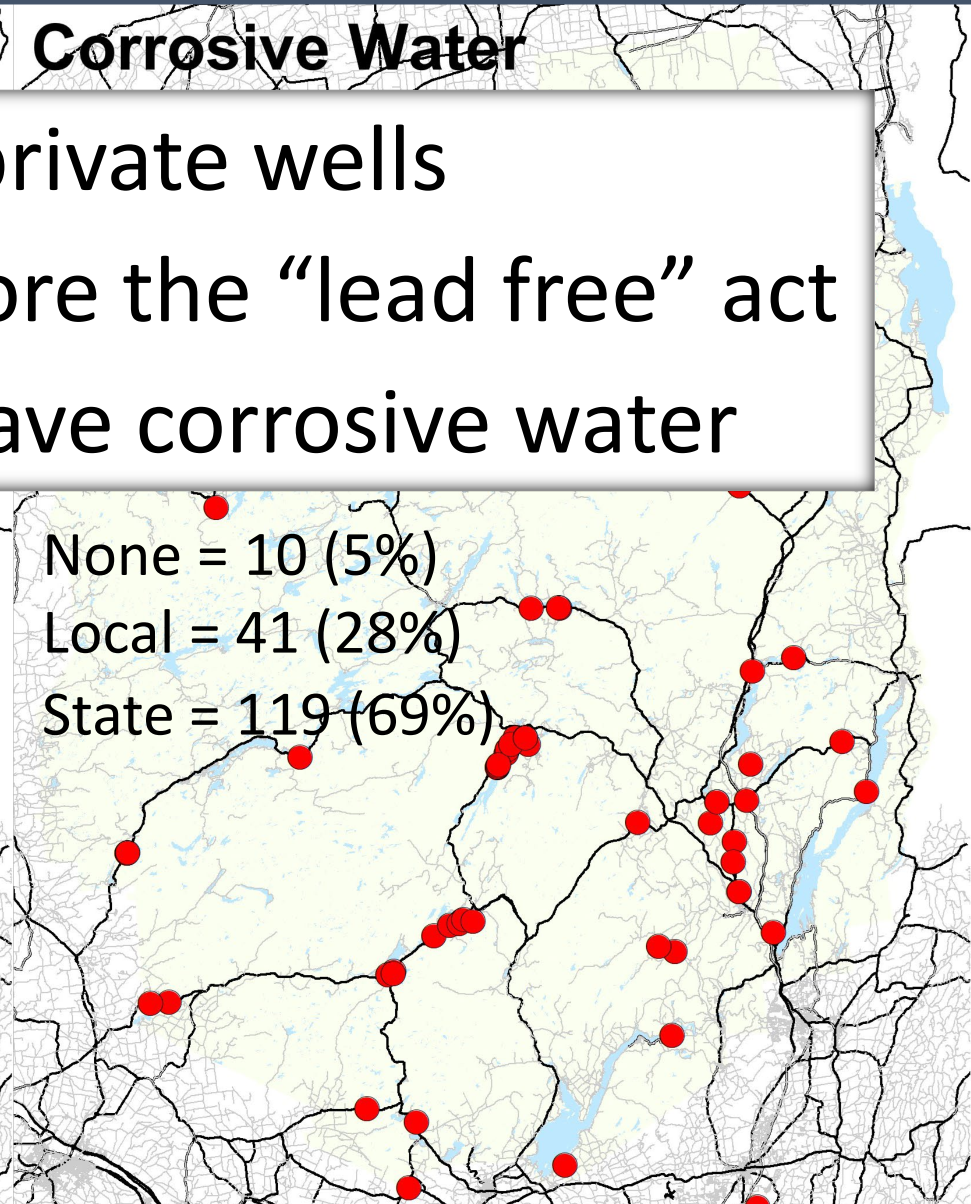
Corrosivity of Well Water



Distribution of Wells Exceeding Thresholds

- 55,000 homes on private wells
  - 40,000 built before the "lead free" act
    - 12,000 may have corrosive water





Conclusions

- Road salt runoff has resulted in widespread groundwater pollution
- Groundwater located downslope of state roads is most polluted
- Many private wells exceed Na drinking water guidelines and have highly corrosive water
- Corrosive water causes damage and may leach lead in older homes and pose a risk to human health

Regional pollution of groundwater by road salt in the Adirondack Park

Dan Kelting

Executive Director

(dkelting@paulsmiths.edu)

PAUL SMITH'S COLLEGE
ADIRONDACK WATERSHED INSTITUTE

Lake Champlain Basin Program

TECHNICAL REPORT NO. 107

Quantifying the De-icing Salt Pollution Load to Mirror Lake and the Chubb River

October 2023

Prepared by:
Brendan Wiltse
Paul Smith's College Adirondack Watershed Institute

For:
The Lake Champlain Basin Program and
New England Interstate Water Pollution Control Commission

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The Lake Champlain Basin Program has funded more than 100 technical reports and research studies since 1991. For complete list of LCBP Reports please visit:
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NEIWPCC Job Code: 0100-328-002

FINAL REPORT

NEIWPCC Job Code:	0100-328-002
Project Code:	L-2019-085
Contractor:	Ausable River Association
Prepared By:	Brendan Wiltse, Senior Research Scientist, PSC AWI
Project Period:	10/4/2019 to 12/31/2022
Date Submitted:	December 2022
Date Approved:	March 2023

QUANTIFYING THE DE-ICING SALT POLLUTION LOAD TO MIRROR LAKE AND THE CHUBB RIVER

CONTACT INFORMATION

Paul Smith's College Adirondack Watershed Institute

7777 NY-30, PO Box 265, Paul Smiths, NY 12970

518-327-6460, bwiltse@paulsmiths.edu

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This project was funded by an agreement awarded by the Great Lakes Fishery Commission to the New England Interstate Water Pollution Control Commission in partnership with the Lake Champlain Basin Program. NEIWPCC manages LCBP's personnel, contract, grant, and budget tasks and provides input on the program's activities through a partnership with the LCBP Steering Committee. The viewpoints expressed here do not necessarily represent those of NEIWPCC, the LCBP Steering Committee, or GLFC, nor does mention of trade names, commercial products, or causes constitute endorsement or recommendation for use.

EXECUTIVE SUMMARY

De-icing salt is an important regional pollutant identified as a contaminant of concern in the Lake Champlain Basin Program's (LCBP) *Opportunities for Action*. It is of particular concern in areas of dense urban development where runoff from roads, parking lots, sidewalks, and driveways can contain high concentrations of salt. This is true for Mirror Lake and the Chubb River, located in the headwaters of the West Branch Ausable River subwatershed. In the case of Mirror Lake, direct stormwater discharge to the lake has resulted in a reduction in spring mixing due to salt-induced density differences within the water column. The primary objectives of this three-year LCBP-funded project were to 1) establish a continuous water quality monitoring program capable of quantifying de-icing salt pollutant load to Mirror Lake and the Chubb River, 2) estimate the de-icing salt pollutant load to Mirror Lake from direct stormwater runoff, 3) estimate the total amount of de-icing salt applied within the Chubb River watershed, and 4) educate the public about the effects of de-icing salt on the environment and BMPs for de-icing salt reduction.

Three continuous monitoring stations were added in June 2020 at the inlet and outlet of Mirror Lake and the outlet of Lake Placid to complement existing stations located in the Upper Chubb River watershed and near the outlet of the Chubb River before it enters the West Branch Ausable River. These stations include data loggers that record stage, conductivity, and temperature every 30 minutes from June 2020 to June 2022. Long-term bi-weekly monitoring of Mirror Lake, begun in 2016, was continued for the project's duration. Discrete monitoring of stormwater entering the lake was also conducted during this period and drew upon a delineation of the watershed area that drains to each stormwater outfall entering Mirror Lake. De-icing salt application within the watershed was estimated through municipal salt tracking devices, and a salt survey was distributed to businesses and residents. Finally, education and outreach campaigns were executed, including installing interpretive signage around Mirror Lake.

Utilizing data from the continuous monitoring system with the Chubb River, we estimate that over 1,000 metric tons per annum of chloride were exported from the Chubb River subwatershed from anthropogenic sources, primarily road de-icing salt. For Mirror Lake, we estimate that approximately 90 metric tons of chloride from anthropogenic sources are exported from the watershed annually. Utilizing municipal salt tracking data gathered by our study, previously published estimates of salt application to state roads, and considering lake chloride retention, we estimate that a minimum of 15-16% of the annual de-icing salt load comes from local roads, 33-35% from state roads, 3-4% from sidewalks, and 42-49% is unaccounted for. The unaccounted salt likely originates from commercial and private applications to sidewalks, driveways, and parking lots. This underscores the importance of addressing these sources of de-icing salt in urban environments. The stormwater data indicates that the highest loads of chloride entering the lake come from the western side. The current data suggest that de-icing salt reduction practices and stormwater improvements in the watershed are working. Still, the lake would benefit from further reductions in de-icing salt application.

CONTENTS

1. Project Synopsis

Increasingly, de-icing salt is acknowledged as an important regional pollutant, and it is also listed as a contaminant in the narrative of the LCBP Opportunities for Action (LCBP 2022). In the Adirondacks, the pollutant load from de-icing salt since the 1980s has been nearly six times the total sulfate and nitrate deposition due to acid deposition (Kelting 2017). The result is the regional salinization of Adirondack waters, affecting both surface and groundwater in the region. Emerging research on impacts on biota further elevates concerns about de-icing salt (Coldsnow et al. 2017; Hintz et al. 2017; Hintz & Relyea 2017a; Hintz & Relyea 2017b; Schuler et al. 2017). Perhaps the most pressing concerns are regional contamination of groundwater supplies and the human health implications of contaminated private drinking water wells (Kelly et al. 2018; Pieper et al. 2018).

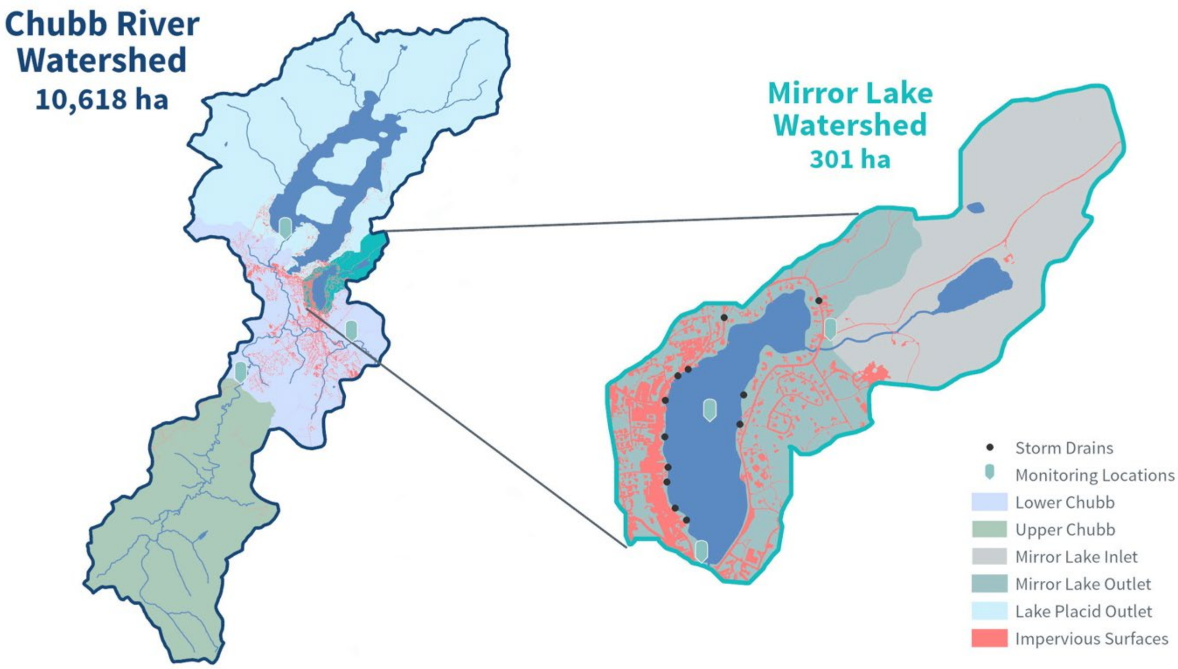

Figure 1. General overview of the Chubb River and Mirror Lake watersheds and the impervious surfaces within them. Stream, lake, and stormwater sampling locations are noted. Data sources: USGS, LCBP.

The Ausable River watershed has the second highest load of salt applied in the Adirondack Park, second only to the Lake George basin (Kelting 2017.). Within the Ausable River watershed, the Chubb River sub-watershed (Figure 1; HUC12) is the most developed and urbanized (Tucker & Treadwell-Steitz 2016; Wiltse et al. 2017). The Chubb River watershed is in the headwaters of the Lake Champlain basin, and heavy urban development is relatively uncommon in this part of the basin. The high density of development, in a small watershed, particularly around Mirror Lake, makes this a model location to study the influence of de-icing salt on Lake Champlain Basin surface waters.

The project directly addresses LCBP FY18 technical funding priority one: Research or innovative demonstration projects that reduce pollution to Lake Champlain, especially nutrients, de-icing agents, and other emerging contaminants of concern. It builds on ongoing work by the

Ausable River Association (AsRA) related to de-icing salt pollution in and around the Village of Lake Placid. AsRA, in partnership with the Paul Smith's College Adirondack Watershed Institute (AWI), regularly monitors Mirror Lake and maintains two continuous monitoring stations on the Chubb River.

In addition, direct runoff from de-icing salt is inhibiting spring turnover in Mirror Lake. Lake water chloride concentrations (40-120 mg/L) range from 160- to 480-times the median concentration observed (0.24 mg/L) in Adirondack lakes not impacted by paved roads (Wiltse et al. 2020; Wiltse et al. 2018; Kelting et al. 2012). Much of the stormwater runoff in the Village of Lake Placid discharges directly to either Mirror Lake or the Chubb River. Understanding the sources and movement of de-icing salt is essential to identifying solutions and monitoring their effectiveness.

With the support of LCBP, this project added continuous monitoring stations to the outlet of Lake Placid, as well as the inlet and outlet of Mirror Lake, completing a continuous monitoring network in the Chubb River subwatershed (Figure 1). Each station consists of a level logger and a temperature/conductivity logger. These, coupled with regular discharge measurements and water samples, allow stage-discharge and conductivity-chloride relationships to be established at each location. Combined with the current work on the Chubb River and Mirror Lake, this network will enable us to quantify the chloride export from the Village of Lake Placid. Stormwater sampling, coupled with LiDAR-derived stormwater runoff models, was used to characterize the salt load to Mirror Lake for individual stormwater outfalls. Additionally, a survey of salt use was conducted to understand private and commercial salt use within the Mirror Lake watershed.

The project included purchasing and installing data loggers on the town and village trucks applying de-icing salt. Resulting data established baseline application rates within the Chubb River watershed at the municipal level. These data helped local road crews optimize and reduce de-icing salt application rates throughout the Village of Lake Placid. Preliminary estimates of salt loads based on the fine-scale delineation of parking lots, sidewalks, and roads, coupled with industry average application rates, indicated that 52% of the lake's chloride load might be coming from parking lot and sidewalk application, 16% from local road application through salt mixed with sand, and 32% from state road application within the Mirror Lake watershed. Applying data loggers and vehicle tracking devices to two town plow trucks, one village plow truck, and the village sidewalk sweeper/drop spreader, along with previously published NYS DOT data, has allowed us to determine a more accurate total salt load and partition that across various application categories. Our preliminary estimates revealed that the parking lot and sidewalk application is potentially a significant load in urban environments and is an area where there has not been much regional focus on implementing BMPs. This project allowed us to develop better measures of the contribution of parking lot and sidewalk salting to the total salt load in urban areas.

The Chubb River watershed size and characteristics present a great opportunity to closely monitor and estimate de-icing salt application rates and monitor how that pollutant moves through an urban lake and a river. A critical component of this project is linking the application rates to the water quality conditions measured in Mirror Lake and the Chubb River. These two aspects of the project will aid in better defining needed reductions in de-icing salt pollutant load, informing target reductions needed through implementing BMPs.

Finally, an essential component of this project has been education and outreach within the Village of Lake Placid and surrounding communities on the impact of de-icing salt on our waterbodies. Public watershed meetings, business outreach, interpretive displays and signage, and school and summer programs were designed and offered to maximize awareness among residents and visitors. A portion of this work was provided as match through established AsRA programs.

This project sought to develop loading estimates for all de-icing salt use in the highly developed Chubb River subwatershed. To date, much of the regional focus has been on state and, to a lesser degree, local road applications. Comparatively little is known about contributions from municipal sidewalk applications, and commercial and private applications. The relatively small area of this watershed, along with existing monitoring programs that have documented significant water quality impairments (lack of turnover in Mirror Lake), offers an excellent opportunity to study and quantify the de-icing salt pollutant load from various of sources. This data collection effort, and the systems established through it, will continue to collect the necessary information to identify, implement, and assess the effectiveness of best management practices to reduce de-icing salt impacts. The outcomes of this work have broad implications regarding understanding de-icing salt pollution in the Lake Champlain basin and understanding how optimization can reduce de-icing salt pollution while still maintaining safe roads, parking lots, and sidewalks. The Chubb River watershed is a good model system for much larger and harder-to-study urban areas in the basin.

This project was organized into three primary objectives: 1) establish a continuous water quality monitoring program capable of quantifying de-icing salt pollutant load to Mirror Lake and the Chubb River, 2) estimate de-icing salt pollutant load to Mirror Lake from direct stormwater runoff, 3) estimate total amount of de-icing salt applied within the Chubb River watershed, and 4) educate the public about the effects of de-icing salt on the environment and best management practices for de-icing salt reduction.

Task #	Task Title	Objective	Deliverable or Output	Timeline
1	Develop a QAPP	Describe quality assurance procedures that will maintain project performance.	QAPP approval	April 2020
2	Install Stations	Purchase equipment and install continuous stream monitoring stations.	Three continuous monitoring stations installed.	June 2020
3-6	Monitoring 1-4	Collect water samples, establish stage-discharge curves, establish conductivity-chloride relationship. Continue Mirror Lake data collection effort.	100% of continuous data on sodium & chloride.	June 2020 – June 2022

7	Stormwater System Mapping	Ground truth stormwater system pour points and outfalls.	Map of stormwater pour points and outfalls.	July 2020 – April 2021
8	Storm-watershed Delineation.	Delineate storm-watersheds for each outfall.	LIDAR based stormwater runoff model.	June 2021 – November 2022
9	Install Stormwater Monitoring	Purchase equipment and install in stormwater outfalls.	Data loggers installed in two outfalls.	June 2020
10-13	Stormwater Monitoring 1-4	Collect and analyze stormwater samples for ~28 runoff events.	100% of data on chloride concentrations in stormwater runoff.	June 2020 – June 2022
14	Distribute Survey	Design and distribute survey.	Survey developed and distributed to area businesses and residents.	March 2022
15	Survey Follow-up	Follow-up with businesses as necessary and compile survey results.	Estimate of private contractor/businesses and residential salt load	March - June 2022
16	Purchase and Install Municipal Salt Tracking Equipment and Plows	Coordinate installation of fleet tracking and data logging equipment on municipal vehicles.	Salt tracking equipment and Live Edge plows installed on municipal vehicles.	April 2020 – September 2020
17-20	Municipal Salt Tracking 1-4	Coordinate training on calibration; ensure ongoing data collection.	100% of municipal salt use data collected.	September 2020 – June 2022
21	First Water Quality Workshop	Host a water quality workshops; incorporate de-icing salt monitoring in one youth program.	Workshop and youth program held.	NA

22	Second Water Quality Workshop	Host a water quality workshops; incorporate de-icing salt monitoring in one youth program.	Workshop and youth program held.	NA
23	Interpretive Displays	Work with designer to develop interpretive displays.	Final displays developed and printed.	October 2022
24	Data analysis and reporting	Compile and analyze project data, write up results into final report.	Quarterly & final LCBP reports	June 2022 – December 2022

2. Tasks Completed

Objective 1: Establish a continuous water quality monitoring program capable of quantifying de-icing salt pollutant load to Mirror Lake and the Chubb River.

Task 2-6: Continuous Monitoring Stations and Lake Monitoring.

Before starting this project, AsRA & AWI maintained two continuous water quality monitoring stations on the Chubb River. One station is above Averyville Road (Upper Chubb), which primarily drains New York State Forest Preserve, and the other is located near the mouth

Figure 2. AWI Research Associate, Lija Treibergs, teaching an AWI Seasonal Research Technician and Paul Smith's College undergraduate student how to measure discharge using an acoustic Doppler velocimeter at the Lower Chubb River site.

of the Chubb River (Lower Chubb) before entering the West Branch Ausable River. Three more stations were added to this network; Mirror Lake Inlet, Mirror Lake Outlet, and Lake Placid Outlet. Each station was outfitted with a Solinst Levelogger to measure stage height and HOBO conductivity logger to measure conductivity. Throughout the project, the sites were visited to collect discrete water samples and discharge measurements to relate stage height to discharge and conductivity to chloride (Figure 2). These stations allow us to monitor water quality entering Mirror Lake and discharging from Mirror Lake to the Chubb River and the contribution of de-icing salt to the Chubb River from Lake Placid. Collectively, these stations, along with those already established, allow us to isolate the chloride load coming from the Village of Lake Placid.

All field equipment was manually inspected for invasive aquatic species and organic matter before use. Equipment was cleaned and dried within the manufacturer's specifications to prevent the spread of invasive species. To the extent possible, equipment was explicitly designated for Mirror Lake (integrated tube sampler, Secchi disc, sample bottles, etc.). A designated canoe was used to sample Mirror Lake and kept at a resident’s house.

Objective 2: Estimate de-icing salt pollutant load to Mirror Lake from direct stormwater runoff

Task 7-8: Storm-watershed Delineation.

To estimate the de-icing salt loading to Mirror Lake from the stormwater system, we need to better understand the drainage area for each stormwater outfall that enters the lake. This involved taking the mapped stormwater system from the Village of Lake Placid, ground truthing and updating it where necessary, and delineating storm watersheds for all entry points for each outfall. We used GIS analysis of recently captured LiDAR data to delineate storm watersheds for the entire stormwater system that drains into Mirror Lake.

This task was complicated because the Village of Lake Placid began a multi-year construction project to completely redesign the stormwater system and replace water and sewer lines under Main St. in 2021. The changes to the stormwater system presented challenges with ground truthing storm drain locations, as many were moved or replaced during this process. The data shown represents the stormwater system before its redesign.

Task 9-13: Stormwater Monitoring

In addition to delineating storm watersheds, we monitored the stormwater runoff entering Mirror Lake for sodium and chloride. Water samples and discharge estimates were collected during runoff events, and winter/spring runoff events were the highest priority for sampling. A small fraction of the total sampling effort occurred in the summer and fall to assess the relative contribution of de-icing salt, or lack thereof, during these times of the year. In addition, conductivity data loggers were installed in two outfalls to monitor stormwater discharge continuously.

Objective 3: Estimate the total amount of de-icing salt applied within the Chubb River watershed.

Task 14-15: Private Contractor/Business and Residential Salt Survey

We distributed a survey to private contractors, residences, and businesses within the Chubb River watershed that helped establish a greater understanding of salt used by the private sector. The survey focused broadly on de-icing and winter maintenance, whether chemical de-icing agents were used, and questions that could help estimate application rates. The largest applicators were followed up with to request annual purchase records for de-icing salt to help better estimate the total load of salt applied.

AsRA has good personal relationships with many members and leaders in the Lake Placid business community, including connections to the Lake Placid Business Association, which helped get survey responses. The survey was delivered in electronic and paper format through various sources, including distribution through local utility bills, and personal follow-up calls or visits were conducted to encourage participation.

Task 16-20: Municipal Salt Tracking

Fleet tracking and data logging devices were added to four municipal vehicles used to maintain the roads and sidewalks in the Chubb River watershed, specifically focusing on Mirror Lake. This included: one tandem plow truck from the Town of North Elba, one tandem and one small plow truck from the Village of Lake Placid, and the sidewalk sweeper/drop spreader used by the Village of Lake Placid to maintain the sidewalk around Mirror Lake. Applying fleet tracking and data logging devices to the entire fleet of vehicles used by the Town of North Elba and the Village of Lake Placid was not feasible within this project's scope. The data collected were used to estimate the salt load on all road and sidewalk surfaces maintained by the Town and Village.

Objective 4: Educate the public about the effects of de-icing salt on the environment and BMPs for de-icing salt reduction.

Task 21-23: Water Quality Workshops and Displays

Two water quality workshops were initially planned as part of this project but had to be canceled due to COVID restrictions and challenges. In place of these workshops, AsRA led a working group that convened local government, businesses, NYS DOT, and watershed groups to discuss de-icing salt reduction strategies within the Village of Lake Placid. This group met regularly throughout the project to discuss progress, share updates, and identify opportunities for further salt reduction.

A briefing of the project's achievements and takeaways was offered to the community in the summer of 2022. Held at the Uihlein Foundation's Heaven Hill Farm, the briefing included presentations by Brendan Wiltse and Leanna Thalmann. Attendees included residents and town administrative personnel.

A critical aspect of addressing de-icing salt pollution holistically is to foster an engaged and informed public. Coupled with the working group, we designed interpretive signs around Mirror Lake. These displays tell the story of the Mirror Lake ecosystems, the challenges associated with stormwater runoff and de-icing salt, and the work being done to address these challenges.

3. METHODOLOGY

Task 2-6: Continuous Monitoring Stations and Lake Monitoring.

Each station consists of a Solinist Levelogger and a HOBO Conductivity logger. This equipment records stage, temperature, and conductivity at 30-minute intervals year-round. In addition, discharge measurements were taken using an acoustic doppler velocimeter every three weeks on average. Sampling visits were targeted to capture the full range of discharges experienced at each station. Finally, water samples were collected from each station during each visit to measure discharge. These samples were analyzed at the Adirondack Watershed Institute for the following parameters: Lab pH, Specific Conductance, Apparent Color, Chlorophyll-a, Total Phosphorus, Nitrate+Nitrite, Alkalinity, Chloride, Calcium, & Sodium using standard protocols (APHA, APHA 2510 B, APHA 2120 C, APHA 10200 H, APHA 4500-P H, APHA 4500-NO_3 I, EPA 301.2, EPA 300.0, EPA 200.7).

Figure 3. AsRA Water Quality Associate, Leanna Thalmann, collecting a water sample from Mirror Lake.

The discharge data was coupled with stage measurements to develop stage-discharge curves for each station (Figure A1). The logarithm of both stage and discharge were plotted, and a linear regression was used to predict discharge based on stage for each site. These models were used to estimate stream discharge on a 30-minute basis based on the continuous data collected from the Levelogger. A linear relationship between conductivity and chloride was also established for each station (Figure A1). The continuous discharge estimates were multiplied by the continuous chloride estimates to develop a continuous estimate of the chloride mass moving past each gaging station during the study period. Data were plotted continuously and summed from June 15 of one year to June 14 of the next to get annual estimates of the chloride export and yield (export standardized to the watershed area) for each station. The June-to-June window was used to coincide with the project period and fully capture when de-icing salt is applied each year.

To estimate the total chloride mass exported from the Chubb River watershed that exceeds the natural chloride yield, we used the three reference sites to calculate a continuous average chloride yield for the study period. This time series was then applied to the entire Chubb River watershed to estimate the total export of chloride from the watershed attributed to natural sources. The difference between the estimated natural export and the observed export at the lower Chubb site is assumed to be attributed to anthropogenic sources, primarily de-icing salts.

Mirror Lake was sampled bi-weekly during the project, except when it was unsafe to sample from the ice (Figure 3). Sampling included the collection of a 2-meter integrated surface water sample, a discrete sample one meter above the lake bottom, and profiles of temperature, dissolved oxygen, specific conductance, and pH at 1-meter intervals through the entire water column. A Secchi reading will also be recorded during each visit during the open water season.

Task 7-8: Storm-watershed Delineation.

One-meter digital elevation models (DEMs) for the Mirror Lake watershed were downloaded from USGS. The elevation data is of bare-earth surface and derived from high-resolution light detection and ranging (LiDAR) data captured at one-meter or higher solution. Five DEM tiles were needed to cover the Mirror Lake watershed, combined into a mosaic to create one continuous DEM for the watershed. Sinks in the DEM were then filled to facilitate accurate flow accumulation modeling. Flow direction was then determined for each raster cell using the D8 method. Finally, continuous flow accumulation was calculated from the flow direction raster. The flow accumulation raster was used to visually identify flow paths that led to individual stormwater drains. Individual storm drains were tied to outfalls discharging the lake using existing stormwater systems maps and ground truthing. Watersheds draining to each storm drain for a given outfall were delineated and combined to represent the total area draining to a specific outfall. All processing was completed in ArcGIS Pro 3.0.2.

Task 9-13: Stormwater Monitoring

All pipes discharging water to the surface of Mirror Lake were sampled during runoff events throughout the project period. During each sampling event, a water sample was collected, and field measurements of temperature, conductivity, dissolved oxygen, and pH were made. If possible, discharge was measured through a timed collection of water into a container of a known volume. When possible, three consecutive measurements were taken to measure discharge, and their values were averaged. Load and yield (based on the stormwater delineation) of chloride at each outfall were calculated for each sampling trip.

Task 14-15: Private Contractor/Business and Residential Salt Survey

The residential and commercial salt survey was delivered electronically using a Google Form, and paper forms were distributed to local businesses. A notice and link to the survey were also included in Village of Lake Placid utility bills to reach individuals who were not online or associated with a business. A summary of survey responses was developed and used to help guide outreach efforts around de-icing salt reduction practices.

Task 16-20: Municipal Salt Tracking

Salt tracking data was imported into ArcGIS, and points were labeled based on whether they were inside the Mirror Lake watershed. The total material applied within the watershed was summed by month and vehicle over the project period. For the three vehicles spreading a sand salt mixture, we assumed the salt constituted 9% of the mixture by mass based on information provided by the local highway departments. To determine the chloride mass, we multiplied the total salt applied by 0.61, which is the proportion of sodium chloride's mass attributed to the chloride ion. This gave us the total mass of chloride measured through the municipal salt tracking efforts.

4. Quality Assurance Tasks Completed

All quality assurance tasks were completed as outlined in the QAPP, except for an oversight on the collection of field blanks during the project's first year. Due to a change in an internal SOP that needed to be updated in a timely manner due to challenges associated with COVID, staff sick leave, and onboarding new personnel, field blanks were not being collected at the frequency described in the QAPP. This was addressed during a site visit with LCBP staff in July 2021 and subsequently determined to be a minor deviation by LCBP and NEIWPCC staff. The SOP was updated in the QAPP and followed for the rest of the project.

5. Deliverables Completed

Installation of three new continuous monitoring stations in the Chubb River sub-watershed with two full years of data collection.

All three stream stations (Mirror Lake Inlet, Mirror Lake Outlet, and Lake Placid Outlet) were installed and brought up to operational status in June 2020 and the two existing stations (Lower Chubb, Upper Chubb) were continued to be maintained. All scheduled field data collection, including discharge measurements and discrete water sampling, was conducted as outlined in the project workplan. Each station had a greater than 90% completeness for all records (Table A4). Gaps in the record were filled with a linear regression for the purposes of future analysis.

Rating curves for each site were developed using a linear fit to log-transformed data for both stage and discharge. All curves had significant relationships (alpha = 0.05) and explained between 86% and 99% of the variation in discharge (Table A3, Figure A1.). These curves were applied to the level (stage) data recorded by the data loggers. Discharge at all five sites exhibited seasonal variation typical for the region, with the highest flows recorded during periods of spring runoff and the lowest flows during summer and early fall (Figure 4).

Conductivity chloride relationships were developed at each site using linear regression. Three of the five sites had significant relationships (alpha = 0.05). Mirror Lake Inlet and Mirror Lake Outlet did not have a significant relationship between conductivity and chloride due to the low chloride concentration at these sites and a limited range of variation (Table A3, Figure A1). The linear regression was used at these sites despite the non-significance because it better approximates the anticipated chloride variation than applying a mean or median. Additionally, the regression slope at the Mirror Lake inlet is close to zero, which means the fit line approximates the global mean at this site. And at the Lake Placid outlet, a trend is observed in the data even though the relationship is deemed non-significant at an alpha of 0.05. Given the low concentrations of chloride at these sites, using the regressions or an estimate of central tendency would keep the overall conclusions of this analysis the same.

The chloride concentrations at the five sites vary from less than 1 mg/L to 60 mg/L. The Lower Chubb, Upper Chubb, and Mirror Lake inlet sites show clear dilution patterns during high-flow events. The Lake Placid Outlet and Mirror Lake Outlet do not display this trend as clearly because the retention capacity of the respective lakes dominates these sites. Therefore, large runoff events at these sites do not translate as directly to reduced chloride concentrations (Figure 5).

The chloride export from each site was largely driven by changes in discharge. The exception is the Lower Chubb site which had more short-period variation in chloride concentration and export than the other sites. Exports ranged from less than 1 to over 120 g/s across the five sites. Because export is primarily a function of watershed area and discharge, the data were standardized by watershed area through the calculation of yield (g/s/ha). Yields at the reference sites (Mirror Lake Inlet, Lake Placid Outlet, Upper Chubb) were all much lower than the impacted sites (Mirror Lake Outlet, Lower Chubb; Figure 6).

Chloride yields from the three unimpacted sites were used to estimate the total background or natural chloride export from the Chubb River subwatershed (Figure 7). This equaled 53.26 metric tons over the 2020-2021 season and 61.40 metric tons over the 2021-2022 season. Subtracting these values from the total observed export at the Lower Chubb site provides an estimate of the additional chloride loading in the watershed due to anthropogenic sources, primarily road de-icing salts. Over the 2020-2021 season, 1,140.56 metric tons of excess chloride were exported from the watershed, and over the 2021-2022 season, 1,019.70 metric tons of excess chloride were exported.

The same process outlined above was used to estimate the background export from the Mirror Lake watershed. This equaled 0.70 metric tons over the 2020-2021 season and 0.92 metric tons over the 2021-2022 season. The additional export estimates for Mirror Lake are 29.20 metric tons for the 2020-2021 season and 37.88 metric tons for the 2021-2022 season. It is important to note that these estimates are only for export through the lake outlet and do not account for export to groundwater. Based on the mass of chloride in the lake and the retention time of the lake, we estimate the total net export from the lake to be 94.45 metric tons over the 2020-2021 season and 89.45 metric tons over the 2021-2022 season.

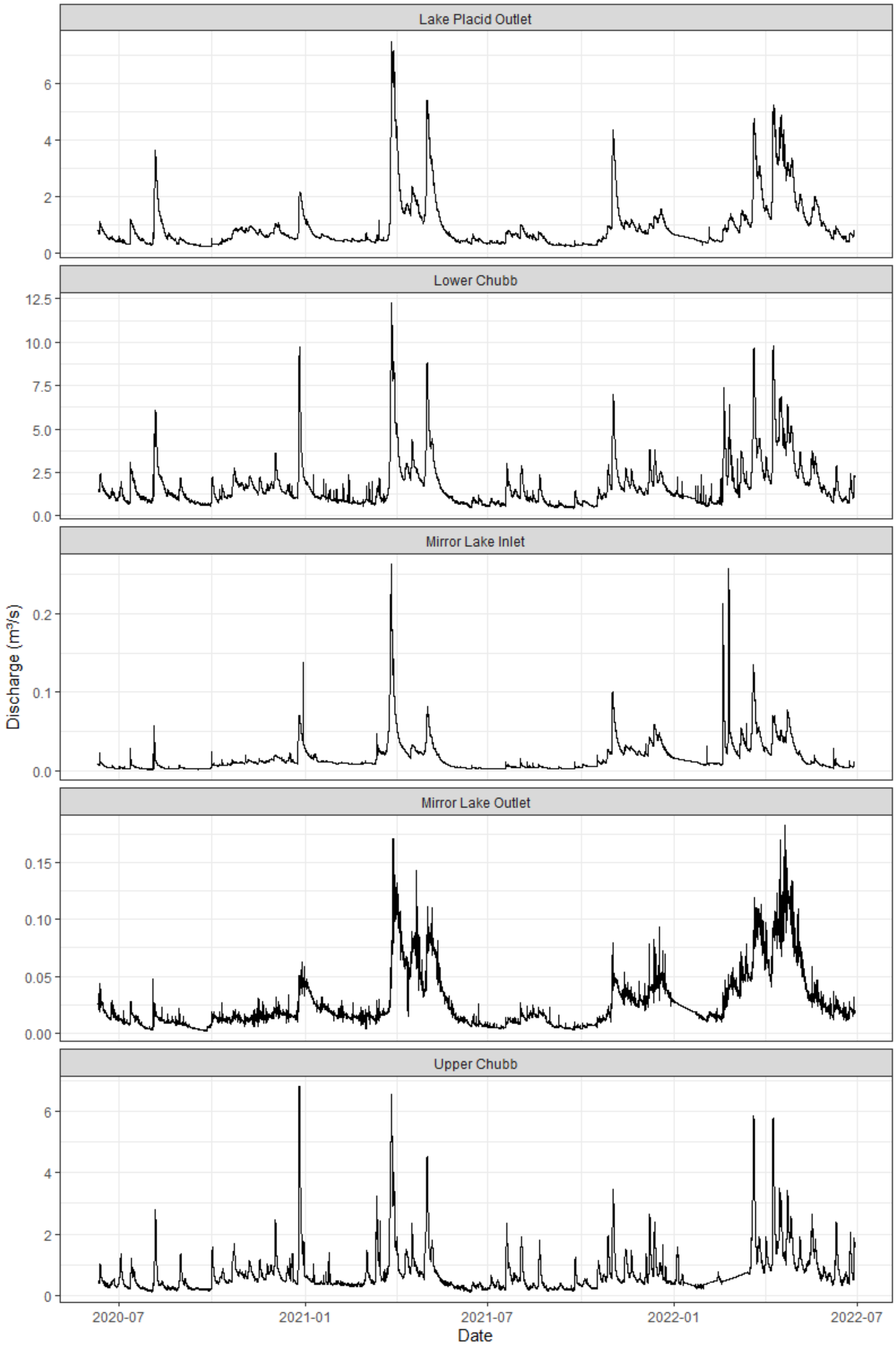

Figure 4. Discharge for the five stream sites in the Chubb River subwatershed.

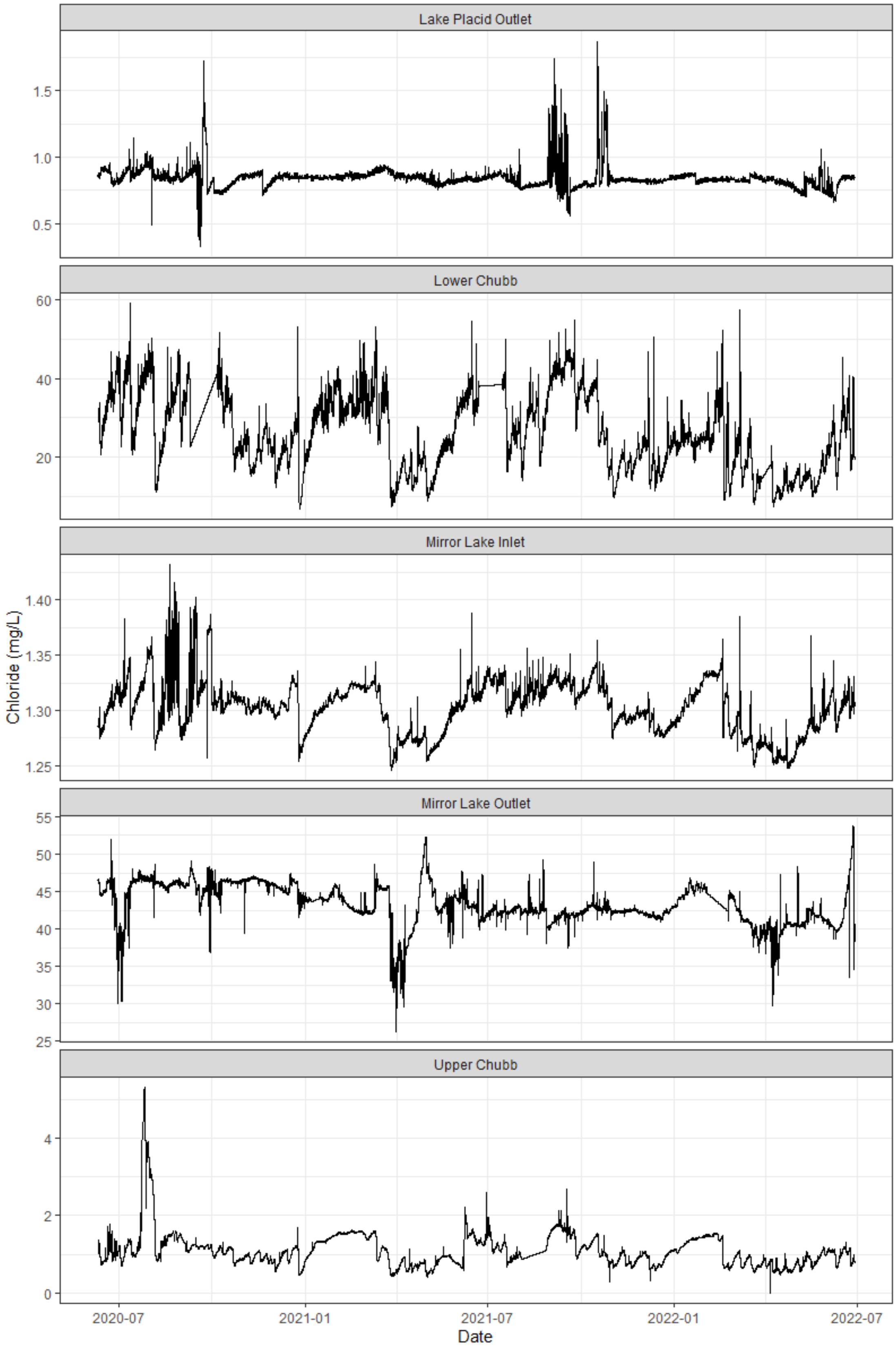

Figure 5. Chloride concentration for the five stream sites in the Chubb River subwatershed.

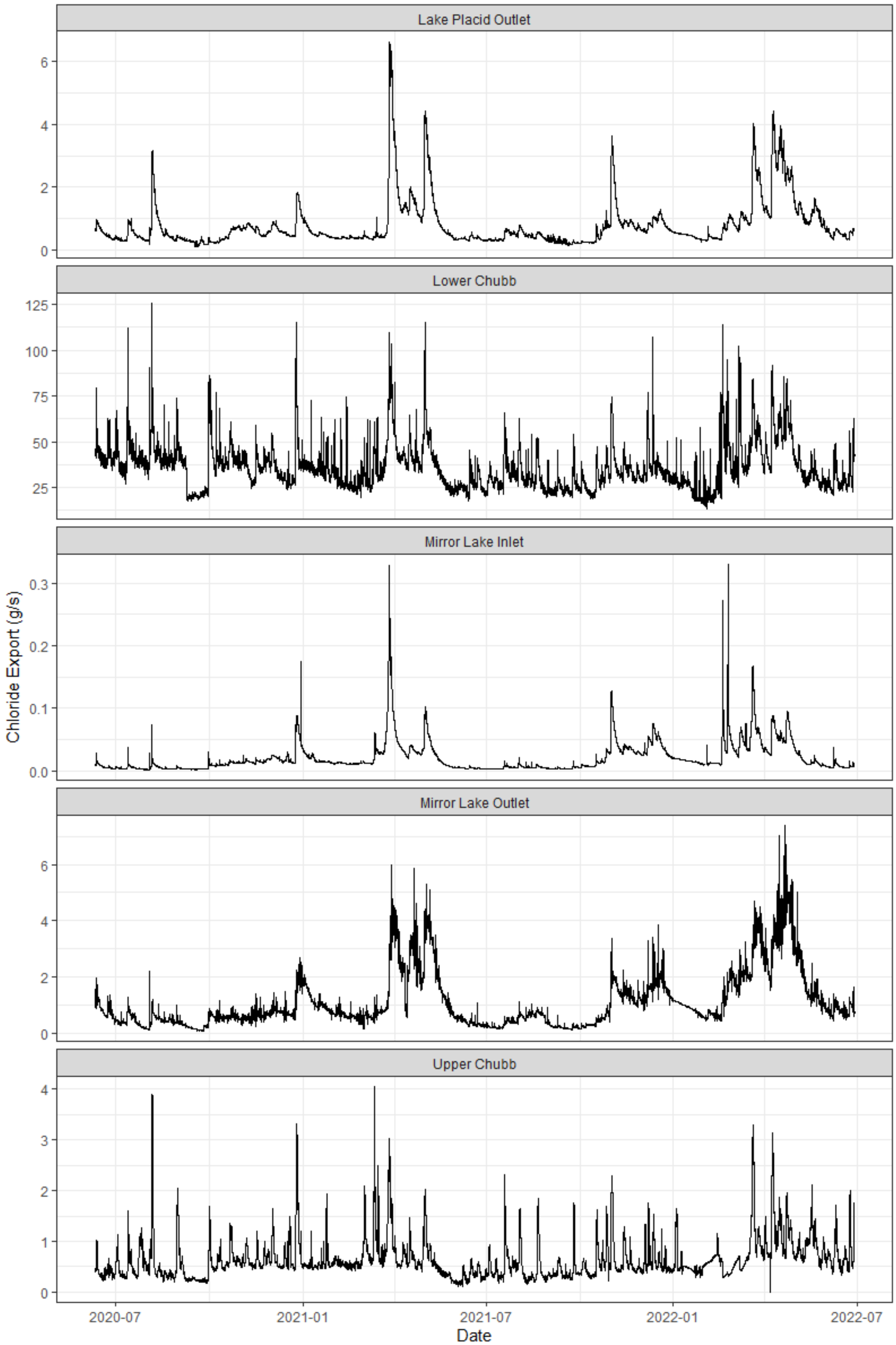

Figure 6. Chloride export for the five stream sites in the Chubb River subwatershed.

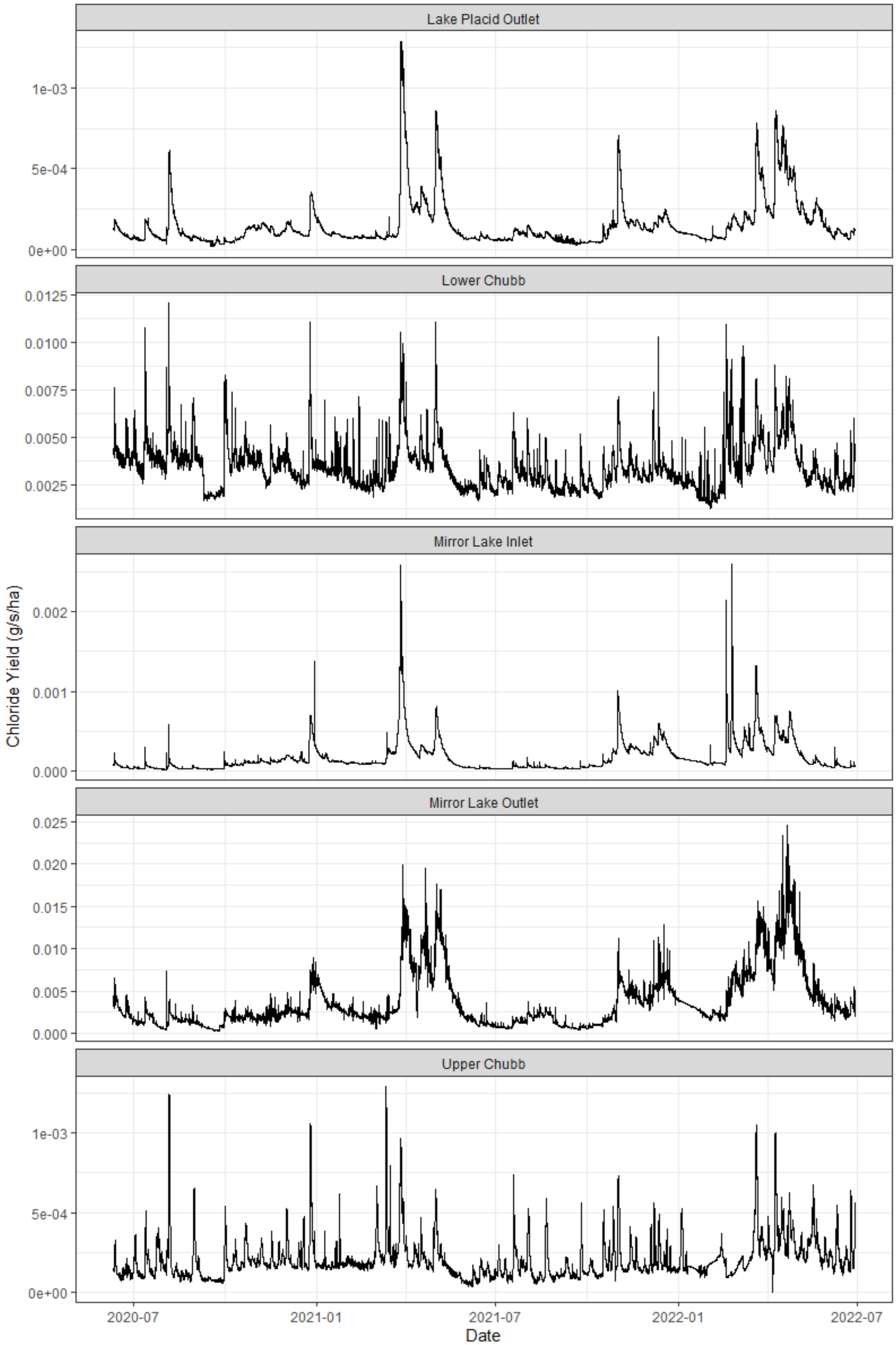

Figure 7. Chloride yield for the five stream sites in the Chubb River subwatershed.

Vertical conductivity profiles and the conductivity-chloride relationship published by Wiltse et al. (2020) were used to develop vertical chloride profiles. The concentration of chloride in Mirror Lake varies through both space and time. Historically, strong vertical gradients of chloride existed in Mirror Lake, driven by the elevated density of stormwater runoff with high salt concentrations. These density differences reduce spring mixing, which has a variety of implications for the physics, chemistry, and biology of the lake. Over the winter of 2019-2020, we saw a noticeable reduction in chloride concentrations at the lake bottom. This season preceded the execution of this grant but coincided with the grant award announcement, which helped motivate local government, businesses, and residents to start exploring salt reduction strategies in the watershed. Further reductions were seen over the 2020-2021 and 2021-2022 winter seasons. The lake completely turned over, or mixed, in the spring of 2020 and 2022, something that had not occurred since 2016 following an abnormally mild winter (Figure 8).

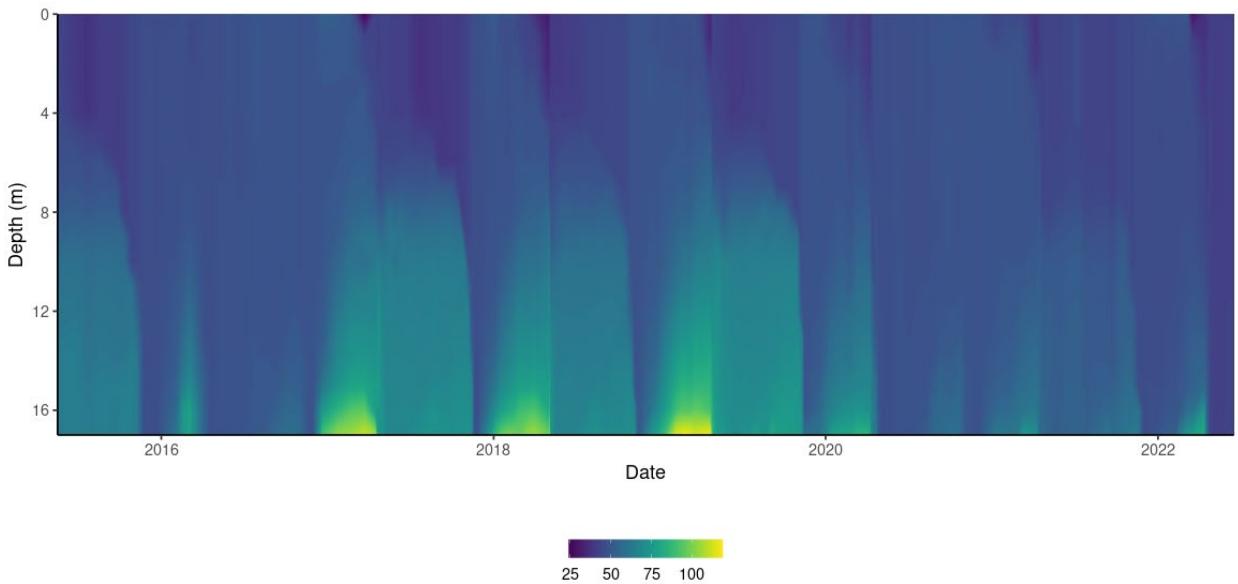

Figure 8. Heatmap plot of chloride concentration in Mirror Lake from May 2015 through July 2022.

The in-lake chloride data was combined with bathymetric data for the lake to determine the total mass of chloride retained. From 2016 through 2019, the chloride retention in the lake followed a distinct seasonal pattern, with retention increasing during the winter while de-icing salt application was occurring in the watershed and retention decreasing during the spring when elevated runoff is occurring. Overall retention in the lake was also trending upward during this period. Beginning with the winter of 2019-2020, this pattern changed, with a much-reduced increase in retention occurring in the winter. An overall decline in retention started at this time as well. This is tied to a combination of changes in de-icing salt practices and stormwater improvements, as are discussed in later sections of this report. Though, the onset of this decline in retention pre-dates the stormwater system changes, suggesting that the best management practices implemented positively affect Mirror Lake (Figure 9).

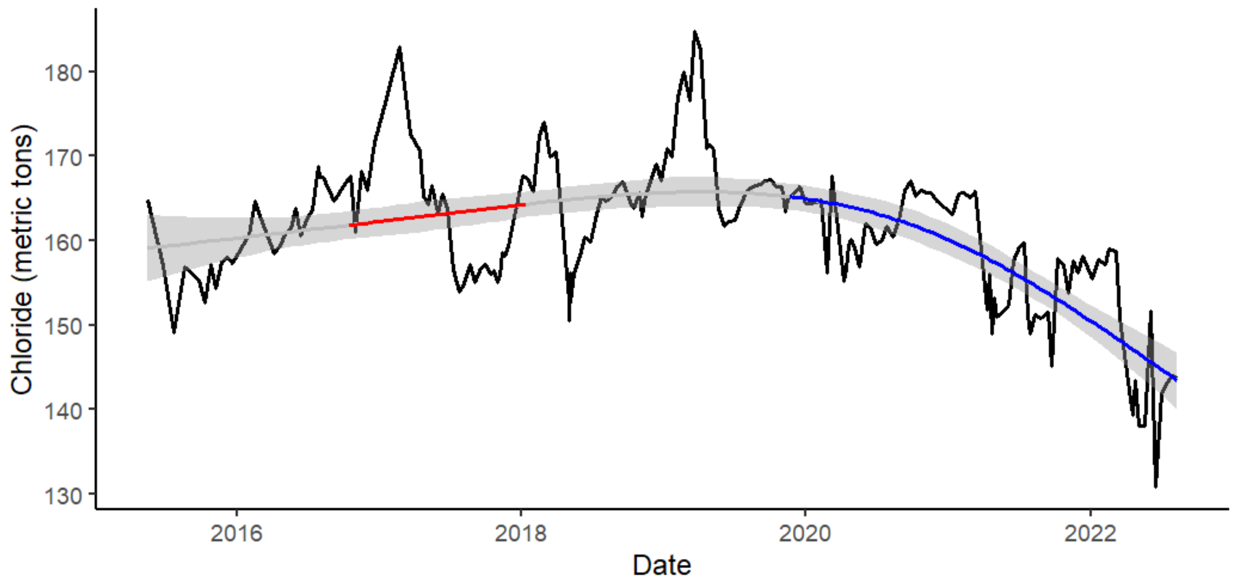

Figure 9. Chloride retention in Mirror Lake from May 2015 to July 2022. The fit line is a generalized additive model (GAM). The red line represents periods when the first derivative (slope) of the GAM is positive, and the blue line represents where the first derivative is negative.

LIDAR based stormwater runoff model

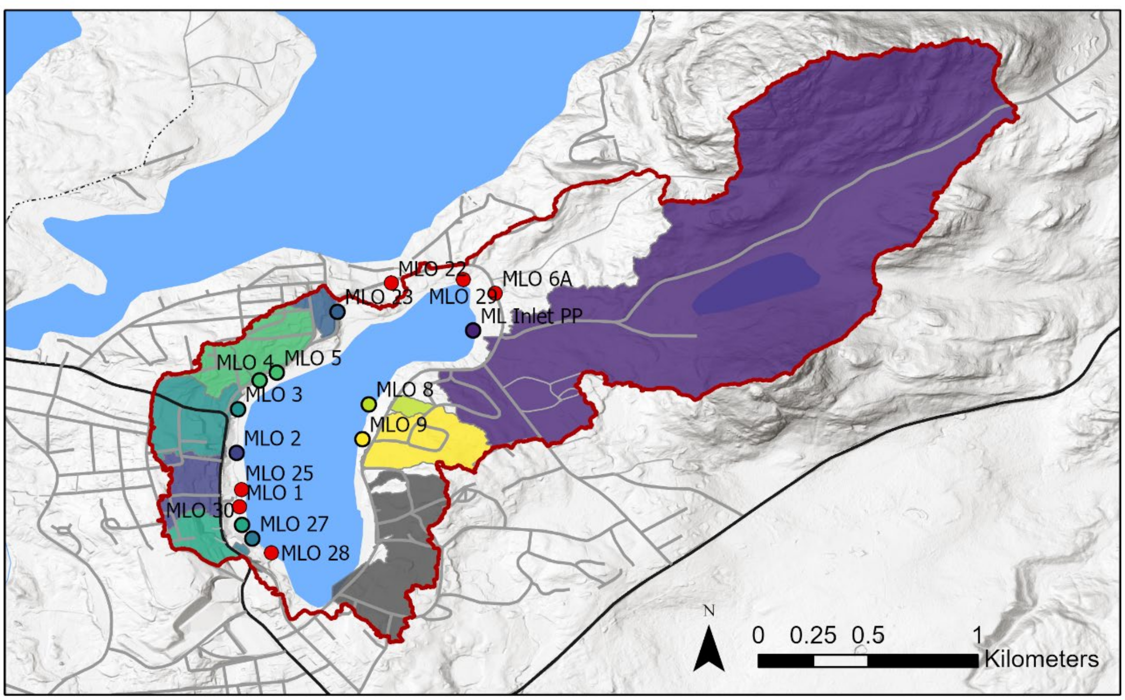

Figure 10. Map showing the areas of the Mirror Lake watershed draining to specific stormwater outfalls. Colors correspond to the associated outfall points. Red points indicate outfalls that were either not in the watershed or determined not to be part of the stormwater system. The grey area drains to storm drains that discharge to a small pond on the Lake Placid Club golf course that is outside of the Mirror Lake watershed. Note that the inlet pour point (Inlet PP) is different from the inlet stream station, which is located upstream. The inlet of Mirror Lake combines with the stormwater system at Mirror Lake Drive before flowing to the lake.

The LiDAR-based stormwater runoff model identifies areas of the Mirror Lake watershed draining to specific stormwater outfalls flowing to the lake (Figure 10). During the project execution, the Village of Lake Placid made substantial changes to the stormwater system along Main Street. The map produced here represents the system before the recent changes. We anticipate that the current system is similar, except that the flows along Main St. now enter underground retention basins and will only flow directly to the lake when runoff exceeds the capacity of those basins to retain runoff.

While sampling outfalls entering the lake, several were identified that appeared not to be tied to the stormwater system, or the runoff model identified them as not being in the Mirror Lake watershed. Many of these were smaller diameter PVC pipes not commonly used as part of a stormwater system and are likely connected to gutters, French drains, or sump pumps. Additionally, a series of outfalls along the lake's southeastern shore are no longer connected to the stormwater system due to prior improvements. This area is shaded grey in the stormwater runoff map and is now draining to a system that routes discharge underground to a pond on the Lake Placid Club golf course outside the Mirror Lake watershed. This creates an interesting and complex hydrological situation in this portion of the watershed. Surface runoff in the grey area is no longer part of the Mirror Lake watershed, but infiltration in this area likely is.

Stormwater Conductivity Data Loggers

The data loggers installed in two stormwater outfalls entering the lake revealed noticeable differences in conductivity at the two sites. The logger installed in the outfall that drains only local roads, parking lots, and sidewalks exhibited episodic spikes in conductivity throughout the winter season of both years associated with the de-icing salt application. The logger installed in the outfall that drains local roads, parking lots, sidewalks, and state roads exhibited higher and more frequent spikes in conductivity. The logger regularly exceeded the operating specifications during the 2020-2021 winter season. A noticeable change in the conductivity occurred at this outfall in the second season of deployment (2021-2022) that reflect changes in the stormwater system leading to the outfall. This outfall was one of several on Main St. with large underground retention basins installed within the stormwater system before the outfall. A marked decline in conductivity spikes occurred, demonstrating the effectiveness of the new stormwater system in reducing direct discharge to the lake (Figure 11).

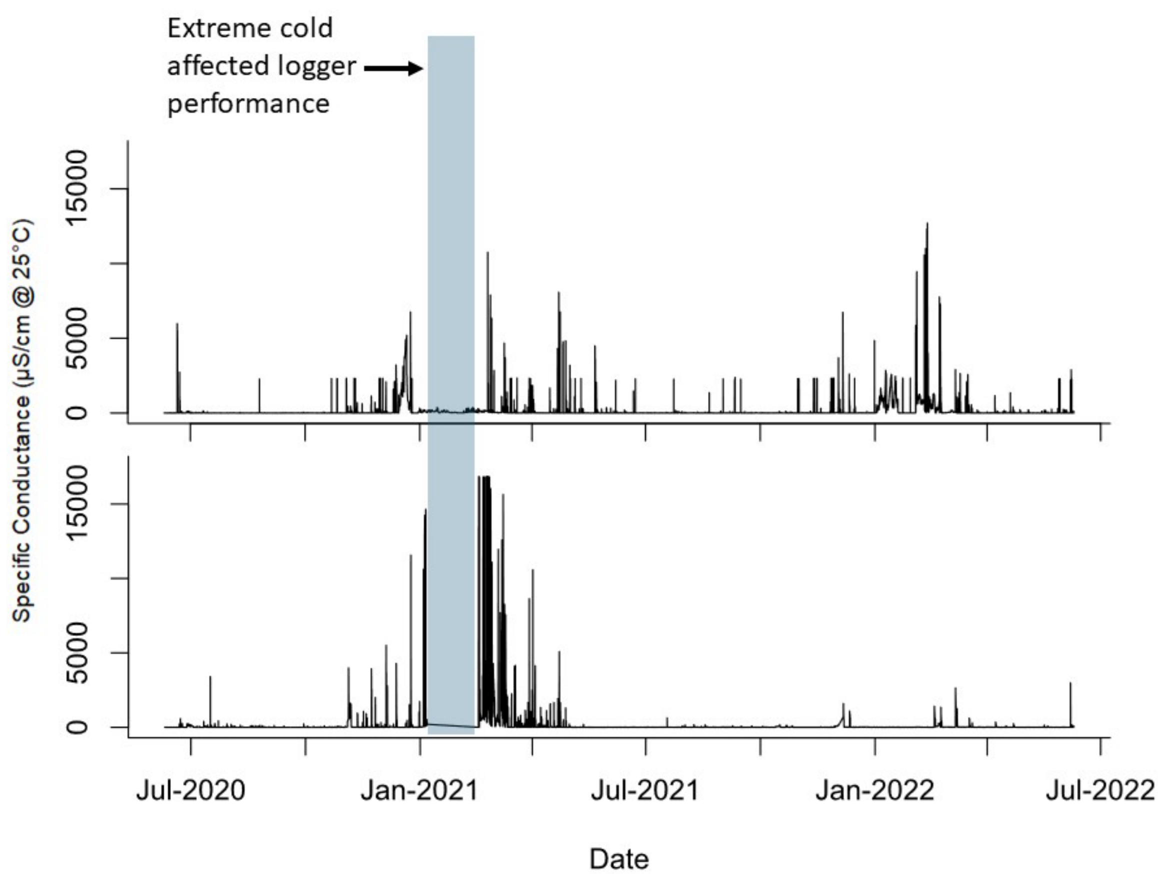

Figure 11. Continuous conductivity logger data for two stormwater outfalls. Top panel: MLO 4, bottom panel: MLO 3. The top panel drains an area that includes local roads, parking lots, sidewalks, and driveways. The bottom panel drains all the same areas plus state road.

Stormwater Data

Stormwater samples were collected from any pipe or outfall flowing into Mirror Lake. Upon completion of the stormwater runoff model and review of the data collected, it was determined that several of these sites are discharging water other than stormwater; this may be from a French drain, roof runoff, sump pumps, etc. Additionally, some outfalls were either difficult or impossible or impossible to sample because they discharge below or partially below the lake's surface (ex. MLO 30). In some cases, water samples could be collected from an outfall, but the discharge was not because the discharge pipes were too close to the lake surface to fit a bucket beneath to measure discharge volume (ex. MLO 2 & MLO 27). Finally, these site-specific challenges varied by time of year based on ice, snow, and water level. Here we are presenting data from the nine outfalls that were mapped as part of the runoff model (Figure 10).

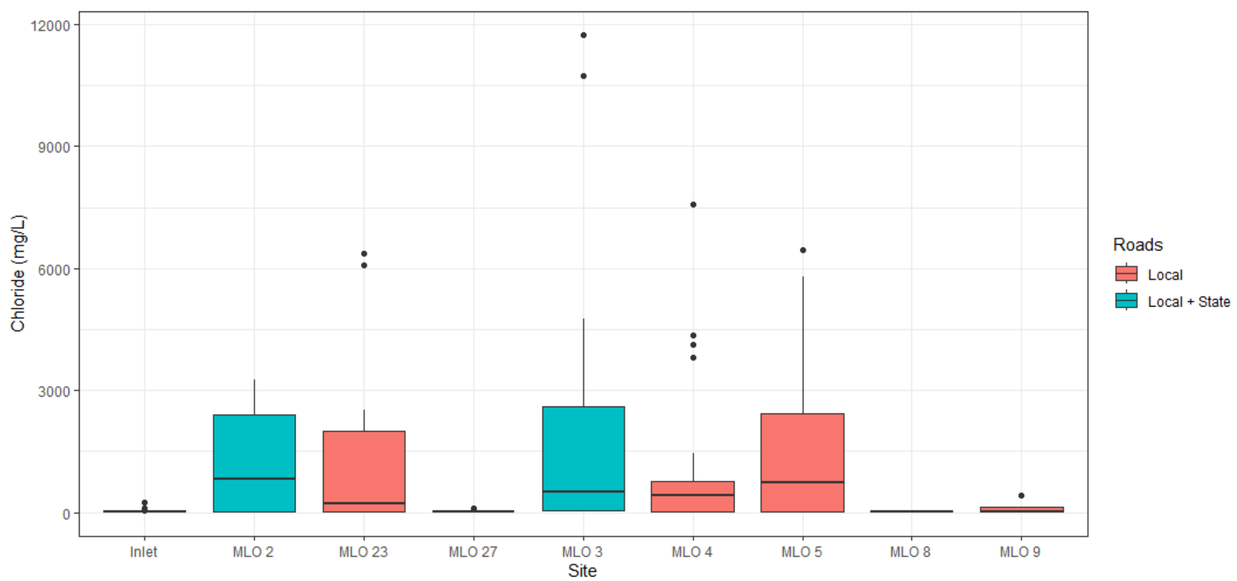

Figure 12. Box plot of chloride concentrations for outfalls discharging directly to Mirror Lake.

Chloride concentrations ranged from a minimum of 2.98 mg/L at MLO 8 to 11,738 mg/L at MLO 3. In general, we saw higher concentrations at outfalls along the western side of the lake and similar concentrations among the western outfalls regardless of whether they were draining only local roads or local and state roads. One exception to this is MLO 27 which enters the lake on the southwestern shore of the lake and is only draining runoff from the Golden Arrow Lakeside Resort parking lot (Figure 12).

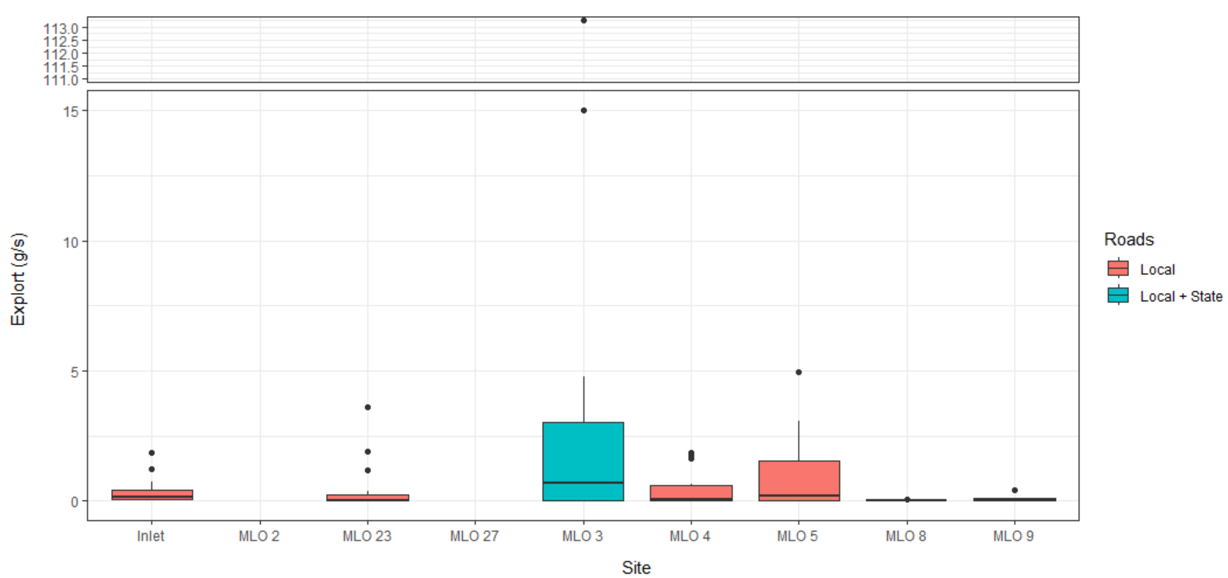

Figure 13. Box plot of chloride export for outfalls discharging directly to Mirror Lake. Note the axis break associated with the outlier for MLO 3.

Chloride export ranged from 2.9 x 10^{-6} g/s at MLO 23 to 113 g/s at MLO 3. The median export at MLO 3 (0.706 g/s) was 4.9 times higher than the next highest outfall (MLO 5). This outfall drains a large portion of the state road that runs through the watershed, including a stretch of road that is on a steep hill going down to the lake. Unfortunately, discharge data could not be collected at MLO 2, another outfall that drains Main St. and state road. MLO 3 had generally higher chloride export than any other outfall. However, it is worth noting that this outfall was part of the stormwater redesign and had a significant reduction in discharge as a result. This is the outfall with a conductivity logger that showed a reduction in conductivity spikes over the 2021-2022 season (Figure 13).

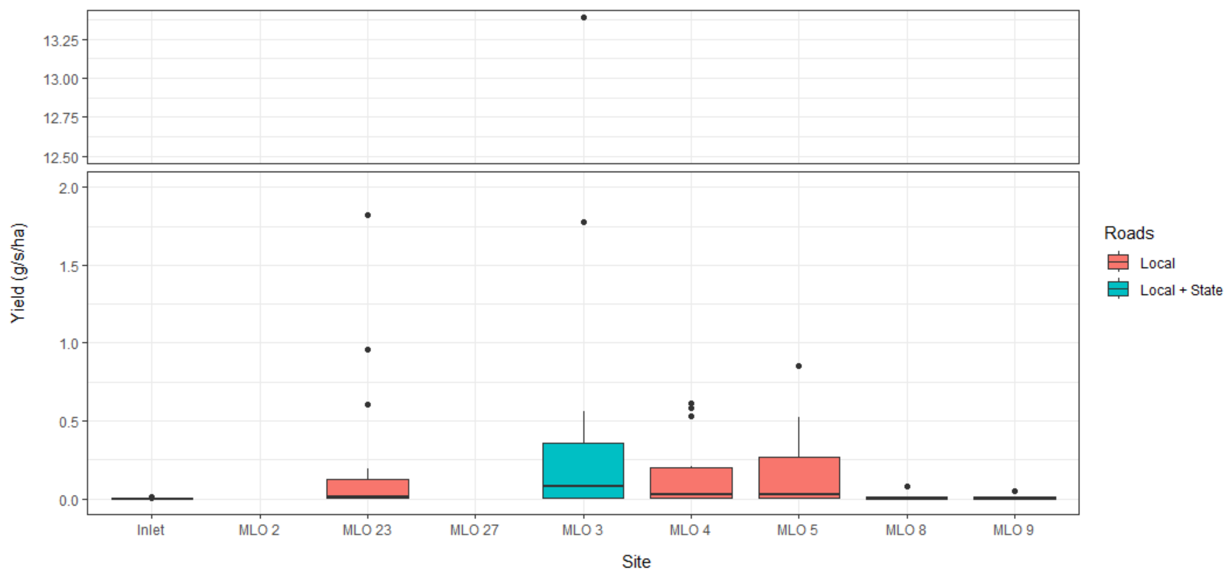

Figure 14. . Box plot of chloride yield for outfalls discharging directly to Mirror Lake. Note the axis break associated with the outlier for MLO 3.

Utilizing the storm-watershed delineation, we can calculate the chloride yield from each outfall. This standardizes the chloride mass exported to the watershed area, allowing better comparisons between sites. The minimum yield was observed at the inlet outfall at 5.7 x 10^{-7} g/s/ha, and the maximum was observed at MLO 3 at 13.4 g/s/ha. The highest median yield was also observed at MLO 3 (0.71 g/s/ha). Similar to the pattern seen with the concentration data, the highest yields were observed at outfalls along the western side of the lake, regardless of road type draining to the outfall (Figure 14).

Municipal Salt Application Data

Salt use data were collected from the Town of North Elba and the Village of Lake Placid within the Mirror Lake watershed. At the start of the project, two salt tracking units were installed on the Town of North Elba's trucks, one unit on the Village of Lake Placid's truck, and one unit on the village's sidewalk sweeper. It was later discovered that one of the two Town of North Elba trucks was not operating in the Mirror Lake watershed and one village truck without tracking equipment was. The tracking hardware from the North Elba truck was moved to the village truck to address this deficiency in December 2020.

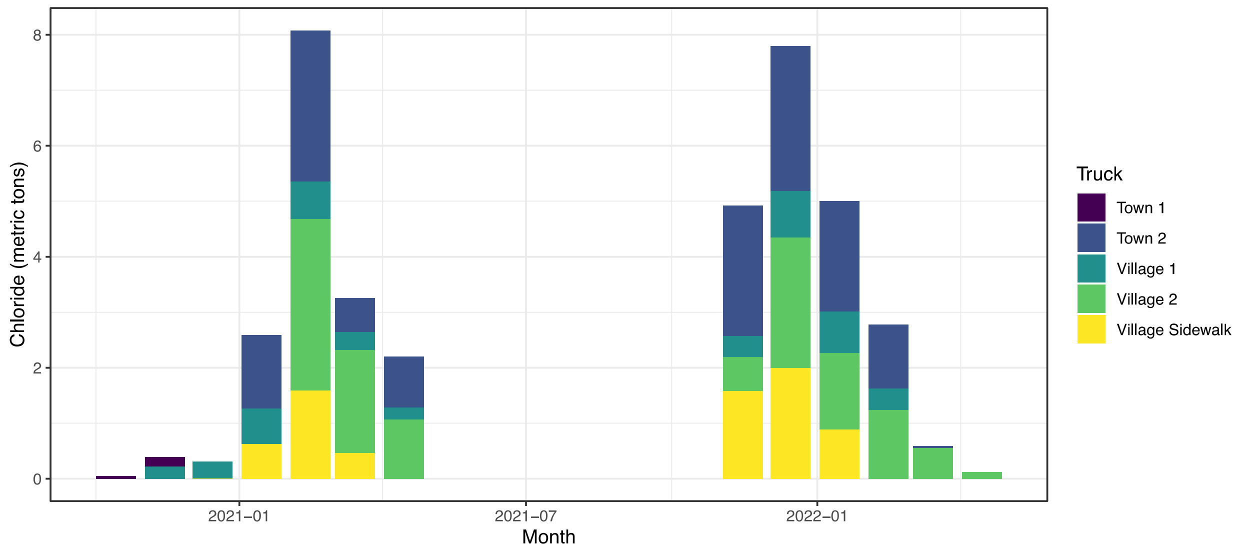

Figure 15. Estimate of chloride load from municipal application to road and sidewalks within the Mirror Lake watershed.

The salt tracking effort faced several challenges throughout the project. One major initial challenge was outfitting the village sidewalk sweeper with the appropriate data loggers. The technology depends on measuring the rotations of the material auger to monitor material application. Installing this technology on a small sidewalk sweeper required unanticipated modifications to the equipment, given that this is a non-standard application of the technology. There were also instances of vehicles being out of service for maintenance and repairs. It was challenging to fully document these issues due to the complex and seasonally demanding nature of local highway department operations. The most consistent and complete record exists for the 2021-2022 season. All data should be viewed as minimum material applied.

In total, we measured 16.86 metric tons of chloride applied to the watershed over the 2020-2021 season and 21.22 metric tons over the 2021-2022 season. Of this, 14.18 & 16.76 metric tons were from road application, and 2.68 & 3.46 metric tons were from sidewalk application over these two seasons, respectively. State salt application data was not applied to be obtained in time for this report due to a requirement to go through a lengthy freedom of information law request process. Estimates of the average salt application rates for state roads indicate that the average chloride load from the state road would be 31.58 metric tons annually (Kelting & Laxson 2010). This brings the total estimated chloride load to the watershed to 48.44 & 52.8 metric tons annually over the two winter seasons. The increase in estimated chloride load between the two seasons may not be real, given the uncertainty in the salt tracking data. In fact, it is possible that the increase is simply a reflection of better data collection in the second season (Figure 15).

Throughout this project, several changes in road and sidewalk management occurred that complicated our understanding of the deicing salt pollution load to Mirror Lake and the Chubb River. Both municipalities and the state were implementing and assessing various best management practices; these include the use of “live edge” plows supported through this grant, targeted reduction of material application within the watershed, a reduction in overall material application rates, changes in sidewalk sweeping and salt application practices, road and stormwater improvements, and other practices. Therefore, application data collected as part of this project is likely not representative of the historical practices that resulted in the pollution of Mirror Lake. While we do not have data to quantify the reduction in de-icing salt application that has occurred over the past several years, the in-lake chloride data suggests that these practices have been effective because the decline in chloride retention began before the stormwater improvements made in 2021.

*Table 1. Estimate of the chloride load to Mirror Lake from different sources over the project period. *State road application is estimated from Kelting & Laxson (2010). **Unaccounted is the difference between the total and accounted-for sources (local roads, sidewalks, state roads).*

	2020-2021		2021-2022	
Source	Chloride (mt)	Percentage	Chloride (mt)	Percentage
Local Roads	14.18	15%	16.76	19%
State Road*	31.58	33%	31.58	35%
Sidewalk	2.68	3%	3.46	4%
Unaccounted**	46.01	49%	37.65	42%
Total***	94.45	100%	89.45	100%

Salt Survey

We received 116 responses to the salt survey sent to businesses, contractors, and residents. 101 of the responses to the salt survey were in the target area. The salt survey was broken down by maintenance areas, including sidewalks, driveways, and parking lots.

Sidewalks

Of the 101 responses, 66 of the respondents maintain a sidewalk. Winter maintenance practices used on sidewalks varied. Of the respondents that maintained a sidewalk, all used a shovel, and 13 people responded that they used salt (Figure 16). Respondents were also asked what level of service they hoped to achieve with their sidewalk maintenance practices. 26 of the respondents aimed for hardpack snow that may have ice present (Figure 17). Survey takers were given the option to respond with “other,” and those who did emphasized their concern with safety. Fourteen of the responses used a sidewalk or entryway deicing product (Table A6). Of the responses, 13 out of 14 used a chloride-based deicing product. One responder used an alternative to chloride. Only three people used two products, and one person used three different deicing products (Table A6). These additional products were all chloride-based.

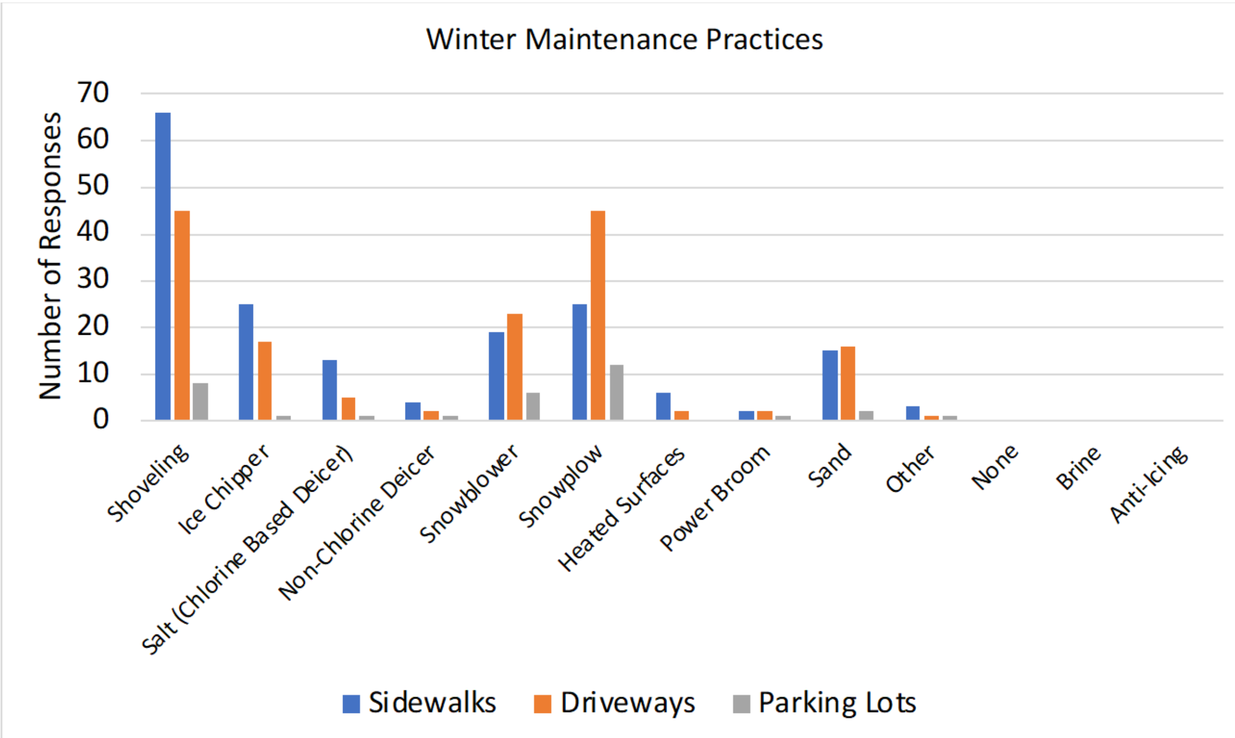

Figure 16. Winter maintenance practices summary by sidewalk, driveway, or parking lot.

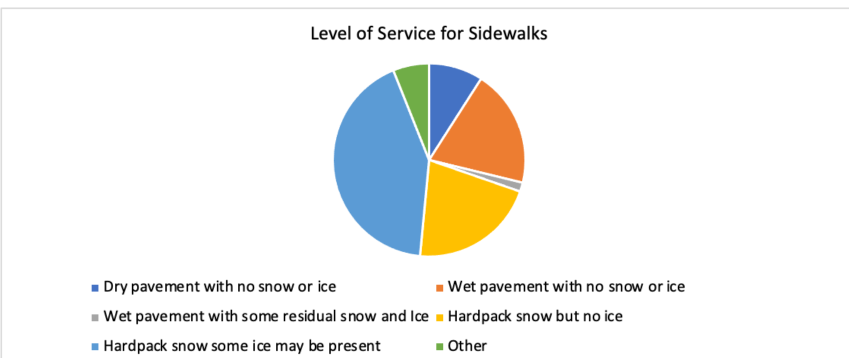

Figure 17. Level of service aimed to achieve in winter sidewalk maintenance.

Driveways

Sixty-four people responded that they maintained a driveway. Of those, 64 people primarily used a snowplow or blower, 16 used sand, five used salt, and two used a non-chloride-based de-icer (Figure 16). Most respondents aimed to achieve a hard snowpack with some ice present, but 12 respondents did aim to achieve wet pavement with residual snow and ice present (Figure 18). Of the 64 responses for driveways, five people responded yes to using a deicing product (Table A7). For the survey, most of the salt used was on driveways.

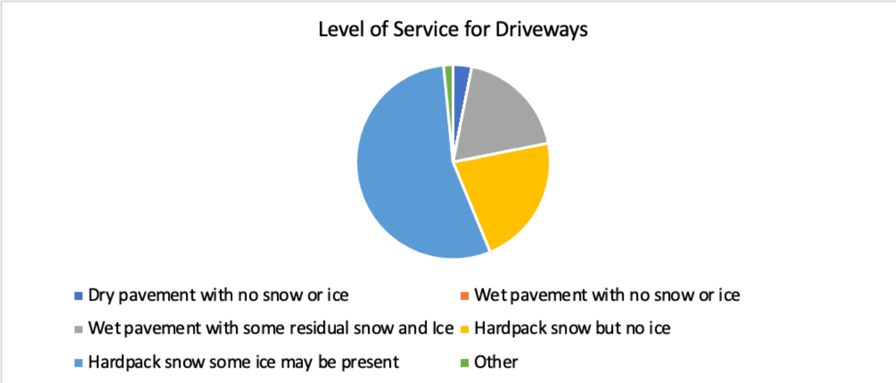

Figure 18. Respondents who maintained a driveway aimed to achieve this level of service.

Parking Lots

Twelve people who completed the survey maintained a parking lot. All responded that they use a plow, half use a snow blower, and eight use a shovel (Figure 16). Although one person said they used salt, and one used a non-chloride de-icer, no one responded that they use a deicing product on their parking lot. Attempts were made to reach these responders for more details, but product names and types were not collected. For the level of service question, most people aim for hardpack snow that may have some ice present (Figure 19).

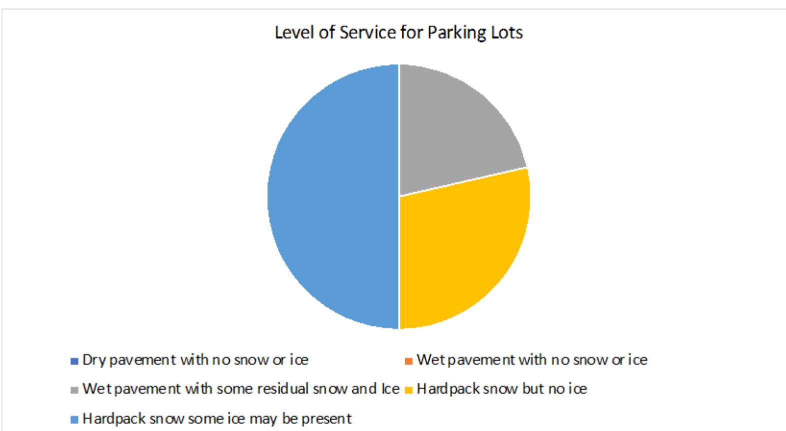

Figure 19. Parking lot level of service.

Education & Outreach

Four colorful and accessible interpretive signs were installed in early October 2022 (Figure 20). They provide information on the aquatic food web, the watershed, de-icing salt impacts, and monitoring efforts on Mirror Lake. The signs were installed by the Village of Lake Placid Highway Department. A visitor can walk around Mirror Lake and see all four signs. The signs are located around Mirror Lake at Mid's Park, Brewster Parkette, the public beach, and at the boat launch easement. A press release was picked up by local news outlets and businesses in the area. A blog was produced and added to the Ausableriver.org website and was featured in the Ausable River Association's e-newsletter.

Press Release Link:https://www.adirondackalmanack.com/2022/10/new-educational-signs-installed-around-mirror-lake.html

Blog Link: https://www.ausableriver.org/blog/new-educational-signs-installed-around-mirror-lake

Figure 20. Visitors reading an interpretive sign next to Mirror Lake installed with the support of LCBP.

AsRA and AWI staff held fourteen events that reached 282 people over the course of this project. These education and outreach efforts include outreach to businesses, municipal partners, high school classes, college classes, and community groups (Table 2). Presentations to regional groups and academic institutions represent an interest in using Mirror Lake as a case-study for salt reduction efforts. Additionally, both AWI and AsRA engaged with local school to get high school aged youth involved with water quality monitoring and salt reduction efforts.

Table 2. Summary of education and outreach events held over the course of this project.

Date	Event	Summary	# of Attendees
August, 2020	Adirondack Water Week	Presentation given at Adirondack Water Week 2020.	75
April, 2021	Salt Use Reduction Initiative Working Group Meeting	Working group meeting with the Village of Lake Placid, the Town of North Elba, highway departments, and local leaders to discuss salt use reduction in Lake Placid.	13
July, 2021	Mirror Lake Watershed Monthly Meeting	The Mirror Lake Watershed Association holds monthly meetings. An update on Mirror Lake work was given.	10
September, 2021	Lake Placid High School	Presentation given to Lake Placid High School students on de-icing salt best management practices to aid in their school project.	8

November, 2021	University of Albany	Presentation given to University of Albany Adirondack Environment class on de-icing salt impacts to the region with emphasis on Mirror Lake as a case study.	22
February, 2022	Salt Use Reduction Initiative Working Group Meeting	Working group meeting with the Village of Lake Placid, the Town of North Elba, highway departments, and local leaders to discuss salt use reduction in Lake Placid.	15
February, 2022	Paul Smith's College	Presentation given to Paul Smith's College Environmental Chemistry class on de-icing salt and impacts to Mirror Lake.	15
March, 2022	Skaneateles Lake Municipal Partnership	Presentation given to the Skaneateles Lake Municipal Partnership as a case study for addressing de-icing salt pollution.	43
April, 2022	Lake Placid Rotary Club Meeting	Leanna led a presentation on the Mirror Lake grant work with emphasis on the Salt Survey.	15
August, 2022	Mirror Lake Public Meeting	We discussed the work on Mirror Lake, findings of the study, and outreach efforts.	16
September, 2022	Northwood School	Leanna gave a lesson to Marcy Fagan's biology class. We discusses nonpoint, and point source pollution, de-icing salt, and delineated the Mirror Lake watershed.	15
September, 2022	Shipman Youth Center Event at Lake Placid Outlet	Leanna led a field based activity. The students sampled the Lake Placid outlet for chloride, phosphorus and nitrogen.	5
October, 2022	Mirror Lake Watershed Monthly Meeting	The Mirror Lake Watershed Association holds monthly meetings. An update on Mirror Lake work was given.	10
October, 2022	Lake Placid Lions Club Meeting	Leanna led a presentation on Mirror Lake, de-icing salt, the salt use reduction initiative, and watershed science.	20

6. Conclusions

This project captured data essential to understanding the de-icing salt pollution load to the Mirror Lake and Chubb River watersheds. The stream, in-lake, and stormwater data provide an important reference for chloride yields and loads from urban areas within the Lake Champlain basin and underscore the importance of de-icing salt best management practices and stormwater management. Importantly, this work determined that de-icing salt application to roads accounts for approximately half the total salt load in urban environments. This finding highlights the importance of working with commercial and private applicators in these areas. This project also highlights the challenges of quantifying the de-icing salt pollution load using salt tracking equipment on municipal and state trucks. Better and improved standard practices around municipal salting tracking are needed if this technology is going to be used to estimate pollutant loads. From a practical perspective, ecosystem monitoring was more effective and reliable in understanding the de-icing salt load in the Mirror Lake and Chubb River subwatershed. Future work in the basin should focus on understanding external factors influencing de-icing salt application, such as weather and climate. While at the same time, it is critical that long-term monitoring efforts capable of quantifying the chloride load to lakes, streams, and rivers continue to track progress toward salt reduction.

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8. Appendices

Appendix 1: Stream Regressions Statistics & Completeness Report

Table A3. Rating curve and conductivity chloride relationship regression equation and statistics for each site.

	Rating Curves			Conductivity Chloride Relationship		
Site	Equation	R^2	p-value	Equation	R^2	p-value
Lake Placid Outlet	y = 2.8x + 0.94	0.99	<0.001	y = 0.0x + 0.18	0.04	0.25
Lower Chubb	y = 2.0x + 1.02	0.93	<0.001	y = 0.2x + 2.14	0.69	<0.001
Mirror Lake Inlet	y = 4.0x + 0.39	0.89	<0.001	y = 0.0x + 1.18	0.00	0.71
Mirror Lake Outlet	y = 3.8x + 0.81	0.86	<0.001	y = 0.2x + 1.30	0.58	<0.001
Upper Chubb	y = 1.8x + 1.05	0.97	<0.001	y = 0.0x – 0.03	0.54	<0.001

Table A4. Summary of the total duration of missing and quality control flagged points within each record.

	Level Logger		Conductivity Logger	
Site	Days	% of Record	Days	% of Record
Lake Placid Outlet	56.5	7.7	0.0	0.0
Lower Chubb	11.3	1.5	62.5	8.6
Mirror Lake Inlet	20.4	2.8	0.0	0.0
Mirror Lake Outlet	20.4	2.8	39.9	5.5
Upper Chubb	40.8	5.6	25.1	3.4

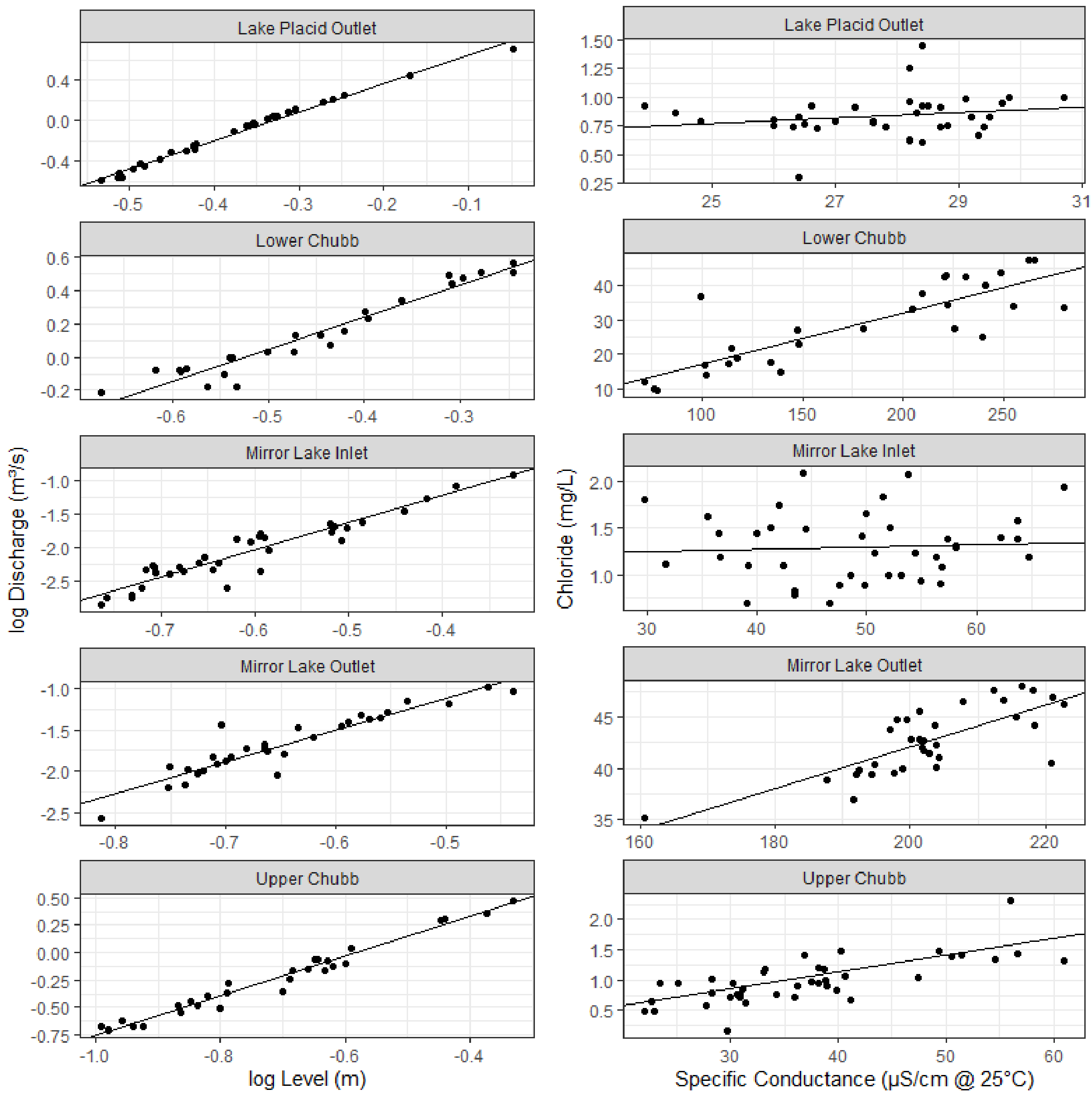

Figure A21. Stage-discharge and conductivity-chloride relationships for the five continuous stream monitoring sites.

Appendix 2. Mirror Lake Profile Data

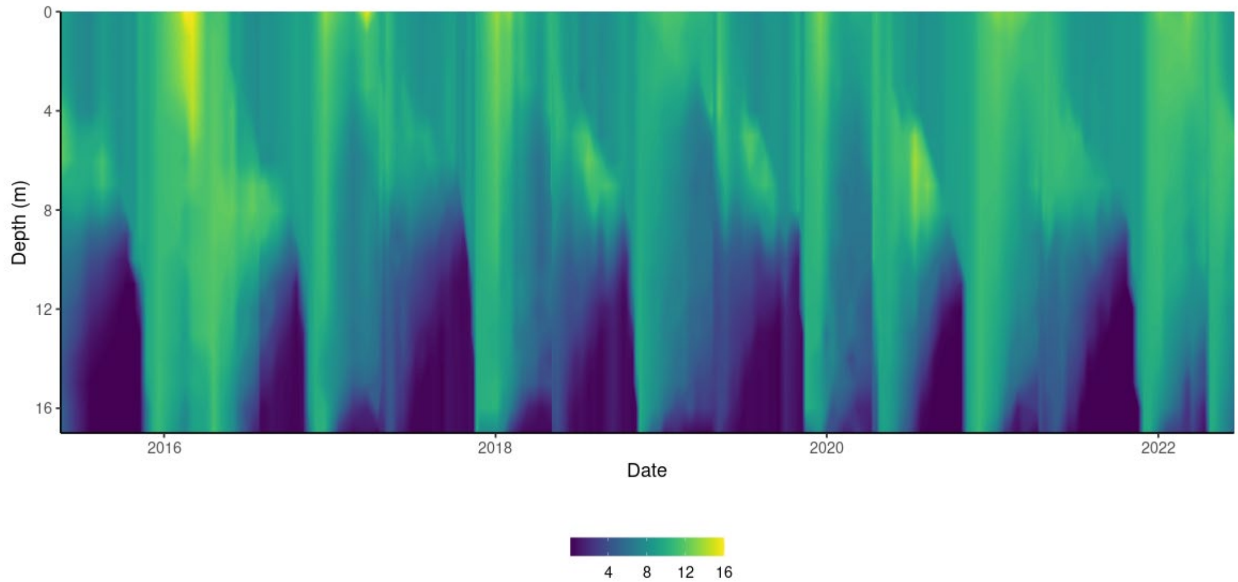

Figure A22. Dissolved oxygen concentraiton in Mirror Lake from May 2015 to July 2022.

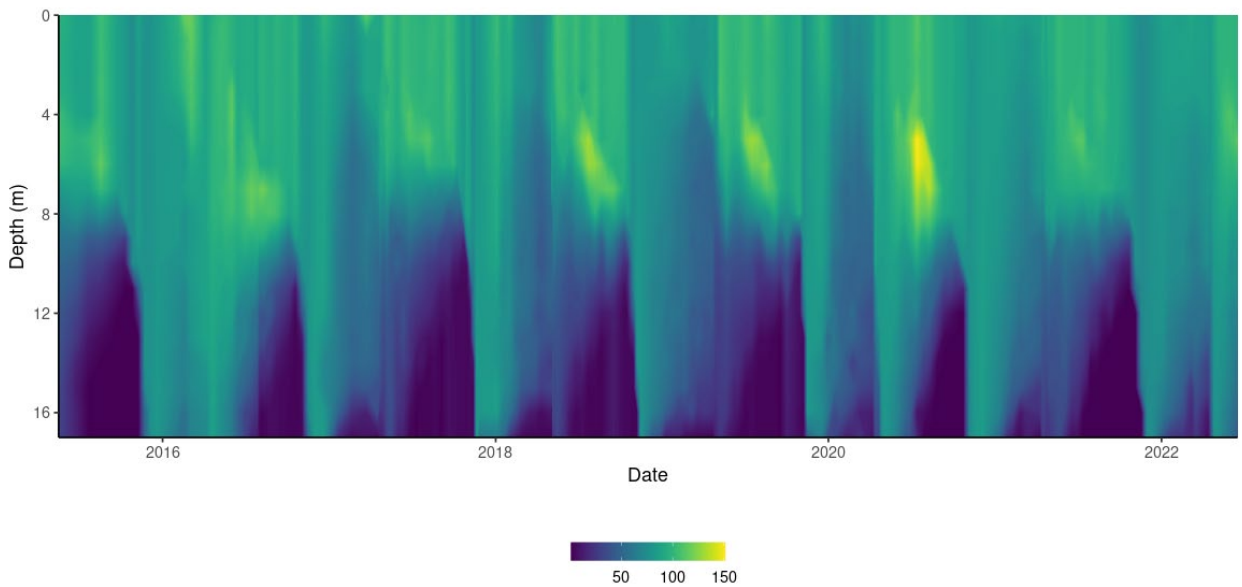

Figure A23. Dissolved oxygen saturation in Mirror Lake from May 2015 to July 2022.

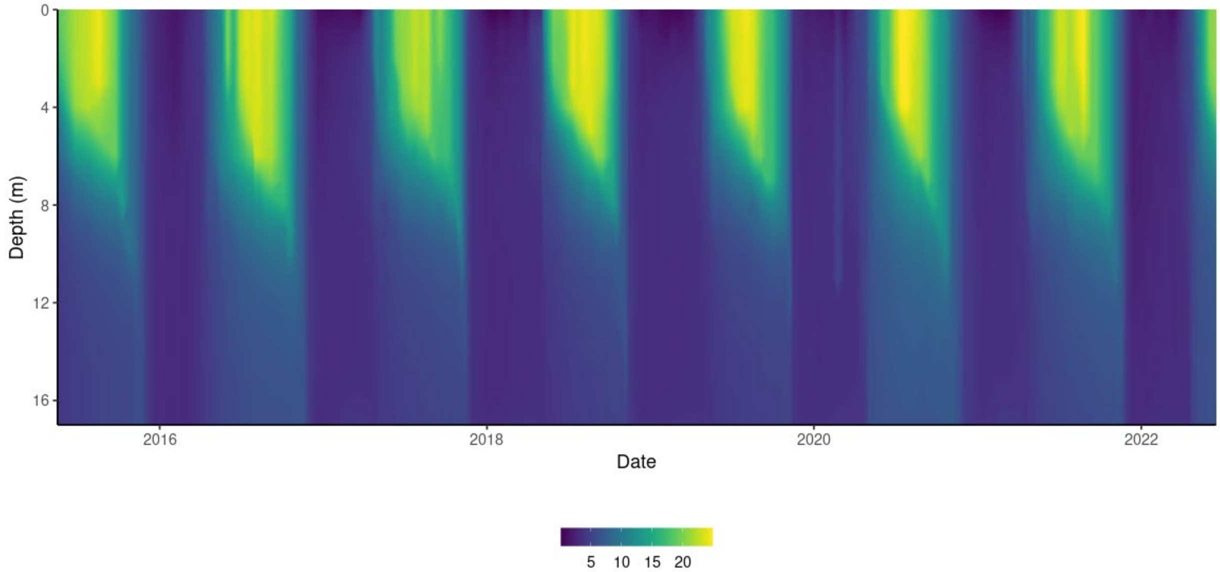

Figure A24. Temperature in Mirror Lake from May 2015 to July 2022.

Appendix 3. Salt Survey Responses

Table A5. Deicing products used on sidewalks from survey responses.

Product Manufacturer	Product Name	Unit Quantity	Number of Units
SWT	Ice melt	40 lb tub/bucket	1
SWI	Miracle melt	50 lb	2
Uline	Ice melt	50 lb	2
Various	Various	Only when the snow turns to all ice	as little as possible
Prestone	Driveway Heat	20 lb	1
Earth Innovations Inc.	eco traction	40 lb	40
Salted sand 80/20	Salted Sand	200 lbs	5 gallon buckets
Vaporizer	Pet safe ice melt	20 lbs	2
Paw Safe	Paw safe ice melt	10 lb	1
Uline	Ice melt	50 lb	2
North American Salt	Safe Step Sure Paws	20 lb	4
North American Salt	Safe Step Sure Paws	50 lb	One bag
North American Salt	Safe Step Sure Paws	20 lb	3
Uline	Ice melt	50 lb	1

Table A6. Supplemental products used by responders.

	Product Manufacturer	Product Name	Unit Quantity	Number of Units
2nd Product	Uline	Pet safe ice melt	20 lb	6
2nd Product	Uline	Ice melt	50 lb	10
2nd Product	SWI	Ice melt	25 lb	1
3rd Product	Staples SWI road runner salt	Ice melt	50 lb	5

Table A7. Deicing products used on driveways from survey responses.

Product Manufacturer	Product Name	Unit Quantity	Number of Units
Safe Paws	Safe Paws	Very little each season	Very little each season
Various	Various	Small amounts only when driveway turns to all ice in spring	Small amounts
Trudeau Sand and Gravel (10% Salt)	Sand Salt Mix	1 Ton	7
Salted Sand (6% Salt)	Salted Sand	18.23 tons	1
Uline	Ice melt	50 lbs	2

Trudeau Sand and Gravel (10% Salt)	Sand Salt Mix	1 Ton	3

Appended Documents:

Press Release AsRA – LCBP Award.pdf – Press release issued announcing the grant award.

Wiltse – ADK Water Week 2020.pdf – Presentation given at Adirondack Water Week 2020.

Wiltse – Lake Placid High Schoool 2021.pdf – Presentation was given to Lake Placid High School students on de-icing salt best management practices to aid in their school project.

Wiltse – Mirror Lake Public Meeting.pdf – Public meeting held to present work related to this project.

Wiltse – PSC Env Chem 2022.pdf – Presentation given to Paul Smith's College Environmental Chemistry class on de-icing salt and impacts to Mirror Lake.

Wiltse – SLMP 2022.pdf – Presentation given to the Skaneateles Lake Municipal Partnership as a case study for addressing de-icing salt pollution.

Wiltse – UAlbany 2021.pdf – Presentation given to University of Albany Adirondack Environment class on de-icing salt impacts to the region with emphasis on Mirror Lake as a case study.

Photos:

Photo 01.jpg – AWI Research Associate Lija Treibergs drilling through the ice on Mirror Lake for winter sampling.

Photo 02.jpg – Reference gage installed along an old bridge abutment at the Lower Chubb site.

Photo 03.jpg – Reference gage installed next to a large boulder at the Upper Chubb site.

Photo 04.jpg – Close-up of the reference gage at the Upper Chubb site.

Photo 05.jpg – Reference gage, stilling well, and conductivity logger installed in the outlet of Mirror Lake.

Photo 06.jpg – Reference gage installed in the Lake Placid outlet.

Photo 07.jpg – AWI staff downloading loggers at the Upper Chubb site.

Photo 08.jpg – AWI Research Technician Connor Vara collecting a water sample using a Kemmerer sampler on Mirror Lake.

Photo 09.jpg – Close-up of stormwater discharge into Mirror Lake.

Photo 10.jpg – AWI and AsRA staff collecting stormwater samples and data during a winter runoff event.

Photo 11.jpg – AWI education staff visiting AsRA & AWI staff while out sampling Mirror Lake.

Photo 12.jpg – AWI & AsRA staff sampling Mirror Lake.

Photo 13.jpg – AWI Research Technician Connor Vara measuring discharge at the outlet of Lake Placid using an acoustic Doppler velocimeter.

Photo 14.jpg – Same as Photo 13.jpg

Photo 15.jpg – AsRA Water Quality Associate Leanna Thalmann sampling Mirror Lake with a Kemmerer sampler.

Electronic Data:

Laboratory and Discrete Data

- LCBP Mirror Lake and Chubb River Data.xlsx – Laboratory and discharge data for discrete chemistry data.
- Mirror Lake Raw Data.csv – Profile data for Mirror Lake.

Municipal Salt Tracking Data

- Lake Placid 2022.csv – Municipal salt tracking data for the Village of Lake Placid over the 2021-2022 winter season.
- Lake Placid and North Elba 2021.csv – Municipal salt tracking data for the Town of North Elba and Village of Lake Placid over the 2020-2021 winter season.
- North Elba 2022.csv – Municipal salt tracking data for the Town of North Elba over the 2021-2022 season.

Salt Survey

- Lake Placid Salt Use Survey (Responses).xslx – Private and contractor salt survey responses.

Stormwater Runoff Model

- Mirror Lake Storm Watersheds.shp – Shapefile of the storm watersheds within the Mirror Lake watershed.

Stream and Stormwater Continuous Data

- Lake Placid Outlet.xlsx – Continuous stage, discharge, conductivity, and chloride data for the Lake Placid outlet.
- Lower Chubb.xlsx - Continuous stage, discharge, conductivity, and chloride data for the Lower Chubb site.
- Mirror Lake Inlet.xlsx - Continuous stage, discharge, conductivity, and chloride data for the Mirror Lake inlet.
- Mirror Lake Outlet.xlsx - Continuous stage, discharge, conductivity, and chloride data for the Mirror Lake outlet.
- MLO 3.csv – Continuous conductivity and temperature data for the MLO 3 stormwater outfall.
- MLO 4.csv – Continuous conductivity and temperature data for the MLO 4 stormwater outfall.

- Upper Chubb.csv - Continuous stage, discharge, conductivity, and chloride data for the Upper Chubb site.