REGIONAL ASSESSMENT OF THE SOURCES AND EFFECTS OF ACIDIC DEPOSITION ON LAKE CHEMISTRY IN ALPINE AND SUBALPINE WATERSHEDS OF NATIONAL PARKS IN THE ROCKY MOUNTAINS, UNITED STATES

by

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Regional Assessment of the Sources and Effects of Acidic Deposition on Lake Chemistry in Alpine and Subalpine Watersheds of National Parks in the Rocky Mountains, United States

Thesis directed by Professor Mark W. Williams

Acidic deposition can adversely affect sensitive aquatic habitats of alpine and subalpine watersheds. Studies were conducted to improve understanding of important controls on sources and effects of acidic deposition on lake chemistry across a regional scale in the United States Rocky Mountains. Relations between basin characteristics, acidic deposition, and lake concentrations of acid-neutralizing capacity (ANC), nitrate (NO_3), sulfate (SO_4), and base cations were evaluated in five National Parks. Multivariate logistic regression models for ANC and NO_3 had the best statistical outcome, with over 93% of lakes in the validation data correctly predicted. Modeling results indicate that elevation had the greatest influence on lake chemistry, followed by bedrock type, steep slopes, aspect, and high NO_3 and SO_4 deposition. The most sensitive lakes to acidic deposition are located in the Southern Rockies. Over 33% of lakes in Colorado National Parks have a high probability for elevated NO_3 and low base cation concentrations and are coincident with areas that have increasing rates of inorganic nitrogen deposition.

The significant correlation ($p < 0.01$) between lake NO_3 concentrations and atmospheric NO_3 deposition was evaluated using NO_3 isotopes at 37 lakes and 7 precipitation sites. Lake NO_3 concentrations ranged from detection to 38 μeq/L, $\delta^{18}O$ (NO_3) values ranged from -5.7 to +21.3 permil, and $\delta^{15}N$ (NO_3) values ranged from -6.6 to +4.6 permil. $\delta^{15}N$ (NO_3) in precipitation and lakes overlap; however $\delta^{15}N$

(NO_3) precipitation is more depleted than $\delta^{15}N(NO_3)$ lakes, ranging from -5.5 to -2.0 permil. Regional patterns indicate that NO_3 concentrations and $\delta^{15}N$ (NO_3) values are more enriched in lakes and precipitation from the Southern Rockies and at higher elevations compared to the Northern Rockies and lower elevations. The correspondence of high NO_3 and enriched $\delta^{15}N$ (NO_3) in precipitation with high NO_3 and enriched $\delta^{15}N$ (NO_3) in lakes, suggests that deposition of inorganic N in wetfall may affect the amount of NO_3 in lakes through a combination of direct and indirect processes such as enhanced nitrification.

Findings and modeling approaches presented in this dissertation may be used to improve long-term monitoring designs of alpine and subalpine watersheds in the Rocky Mountains and may be transferable to other remote mountain areas of the United States and the world.

DEDICATION

To my husband, best friend, and trusted academic and professional sounding-board, Jason, for his endless love, support, and encouragement. To my daughter Talia for the love and joy she brings into my life. To my parents for their unwavering love and support.

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CHAPTER 1

INTRODUCTION

This dissertation presents research from three related studies conducted across five National Parks in the Rocky Mountains of the United States. The objectives of these investigations were to advance approaches for analysis and improved understanding of important controls on sources and effects of acidic deposition on solute chemistry in alpine and subalpine lakes across a regional scale. The first part of the research focused on identifying lakes that are sensitive to acidic deposition. Next, regional patterns in NO_3 sources to watersheds were evaluated using NO_3 isotopes. The ultimate goal of the final phase of the research was to evaluate regional patterns in the relation between GIS-based landscape and acidic deposition attributes and identify the processes that control the spatial distribution in lake concentrations of NO_3, SO_4, and sum of base cations across the Rocky Mountains. Chapter 1 presents an introduction that includes a background discussion of the relations between basin characteristics and lake solute chemistry, the evaluation of sources and effects of DIN deposition in high-elevation lakes, tools for relating basin characteristics and solute chemistry, and an organizational outline of each chapter.

In the western United States (US), energy generation, transportation, industry, and agriculture produce anthropogenic emissions of NO_x (nitrogen oxides), NH_3(ammonia), and SO_2 (sulfur dioxide) that contribute to deposition of dissolved inorganic nitrogen (DIN = nitrate (NO_3) + ammonium (NH_4)) and sulfate (SO_4) to high-elevation watersheds (Baron et al., 2000; Williams and Tonnessen, 2000). Over the last decade, deposition data indicate that DIN in wetfall has increased steadily over much of the Rocky Mountains (Baumgardner et al., 2002; Fenn et al, 2003; Nilles and Conley, 2001) due to increases in motor vehicle emissions and ammonia emissions (NADP, 2007) that have offset reductions in NO_x emissions from fossil fuel burning industries (USEPA, 2000). The percent of DIN in wet deposition contributed by NH_4 has increased from 1992-1996 to 2002-2006, and is now 50% or greater of measured DIN in over half the National Atmospheric Deposition Program (NADP) National Trends Network (NTN) sites in the Rocky Mountains (NADP, 2007), including sites located near national parks. SO_4 deposition has decreased in recent years (Lynch et al. 1996). On a global scale, data from over 4000 lakes in 42 regions of Europe and North America, show a significant correlation between increased wet DIN deposition and elevated lake DIN concentrations (Bergstrom and Jannson, 2006).

Alpine and subalpine ecosystems in national parks of the Rocky Mountains are particularly vulnerable to natural and human-induced stressors (Williams et al., 2002). Deposition of acidic solutes to high-elevation lakes has the potential to change the nutrient balance of aquatic ecosystems, increasing the possibility of episodic acidification and change in nutrient status (Baron, 2006; Williams et al., 1996). Previous work in US National Parks of the Rocky Mountains suggests that high-elevation lakes are showing a response to atmospheric inputs of NO_3 and SO_4 (Clow et al., 2002).

Study Area

Five national parks in the Rocky Mountains, including Glacier National Park, Montana; Grand Teton National Park, Wyoming; Great Sand Dunes National Park and Preserve, Colorado; Rocky Mountain National Park, Colorado; and Yellowstone National Park, Wyoming were selected for this research for several reasons. They are home to diverse wildlife and vegetation and are the largest and most visited national parks in the region. Four of the five parks are located along the Continental Divide and therefore have a large number of alpine and subalpine lakes (n=769), that range in elevation from less than 1,000 m in Glacier National Park to over 3,500 m in Rocky Mountain National Park. The areas under investigation form the headwaters of most of the major rivers in the western United States, and their airsheds extend across state and national political boundaries. These national parks are in glaciated mountainous terrain (Madole, 1976; Richmond and Fullerton, 1986). Dominant bedrock types for each park are as follows: granitic rocks in Grand Teton National Park (U.S. Geological Survey, 1992 a) and Rocky Mountain National Park (U.S. Geological Survey, 1990), sedimentary rocks in Glacier National Park (U.S. Geological Survey, 1992 b), volcanic rocks in Yellowstone National Park (U.S. Geological Survey, 1988), and granitic and sedimentary rocks in Great Sand Dunes National Park and Preserve (National Park Service, 2004).

Average annual precipitation in the Rocky Mountains increase as a function of elevation and latitude. Mountainous areas above 3,000 m elevation generally receive at least 800 mm precipitation per year, most of which accumulates in a seasonal snowpack (Spatial Climate Analysis Service, 2000). Average annual atmospheric deposition maps show regions of high inorganic nitrogen and sulfate deposition within Rocky Mountain National Parks (Nanus et al., 2003). Within the study area, mean annual atmospheric deposition ranges are as follows: hydrogen ion deposition ranges from 0.03 to 0.2 kg/ha/yr hydrogen ion, 0.2 to 4.3 kg/ha/yr inorganic nitrogen, and 0.7 to 12 kg/ha/yr sulfate (Nanus et al., 2003).

Fifty-eight lakes were randomly selected for water-quality sampling and are spatially distributed within each of the national parks. Many of the lakes are located in alpine and subalpine terrain where soils are poorly developed, vegetation is sparse, and growing seasons are short. Surface waters were collected from the outflow of each lake during the low-flow period from August through September. Lakes with a surface area greater than 1 hectare (ha) were used to avoid inclusion of small tarns and ponds.

Background

Relations between basin characteristics and lake solute concentrations

The important hydrologic and biogeochemical processes that influence alpine and subalpine lake sensitivity to acidification from atmospheric deposition of acidic solutes will be discussed in this section. In pristine mountain ecosystems, mineral weathering, cation exchange, and biologic processes in soils can affect water chemistry (Clow and Sueker, 2000). In the Rocky Mountains of the Western US, high-elevation lakes with low acid-neutralizing capacity (ANC) concentrations (less than 100 μeq/L) are the most sensitive natural resource to nitrogen inputs (Williams and Tonnessen, 2000). ANC is a measure of the buffering capacity of water to acidic inputs and a measure of the concentration of solutes. It results from the presence of bicarbonate, carbonate, organic acids, and alumino-hydroxy complexes in the water (Kanciruk et al., 1987). The acidification of lakes can be described as a loss of alkalinity (ANC) over time, which can be related to the increase of acidic anions (NO_3 and SO_4) and a decline in bicarbonate (Brakke et al., 1989). These lakes also have typically low sum of base cations (Brakke et al., 1989) that increases their vulnerability to acidic deposition.

High-elevation watersheds in the Rocky Mountains have physical characteristics that make them particularly susceptible to acidic deposition, including steep topography, thin and rocky soils, sparse vegetation, a short growing season, and the storage and release of pollutants in snowmelt runoff from deep snowpacks (Turk and Spahr, 1991 Williams et al., 1996, Baron and Campbell 1997, Clow and Sueker, 2000). Residence time of water in a catchment can influence water quality because many biogeochemical reactions are time-dependent (Hornberger et al., 1998; Burns et al., 2004). Lake sensitivity to acidification increases during periods of high influx of water that have shorter hydraulic residence times and rapidly transport atmospherically derived chemicals through or over the catchment into lakes without interacting with geologic weathering products that could buffer the acidity (Stoddard, 1987, Landers et al., 1987, Clow et al., 2002).

Geochemical processes, including mineral weathering and cation exchange, play an important role in neutralizing acidic compounds because they are the main source of base cations and ANC. Lake ANC concentrations are typically higher in alpine basins with carbonate lithology because limestone and carbonate rocks are very effective in neutralizing acidity (Berg et al., 2005, Clow and Sueker, 2000). Minerals such as quartz dissolve extremely slowly, feldspars and mafic minerals at intermediate rates, and carbonates very rapidly (Drever, 1997). Bedrock geology can be reclassified to geochemical ranking based on potential buffering capacity of the bedrock (Nanus and Clow, 2004). When chemical weathering rates are slower, the potential buffering capacity of bedrock is lower (granite) and the lake is more susceptible to acidification due to a lack of weathering products. Soil type, spatial extent, and depth of soil can affect alkalinity. Internal alkalinity is generated by cation production through chemical weathering. In thin soils, the opportunity for water to contact and react with bedrock materials is greater and water chemistry may reflect the bedrock mineralogy more closely in these basins (Moldan et al., 1994).

Based on the important controlling processes, ANC is hypothesized to be low (less than 100 microequivalents per liter) in lakes that are in moderate to high-elevation headwater basins, having little or no vegetation, with little or no soil cover, low buffering capacity bedrock, decreased chemical weathering, and lakes with locations that receive high acidic deposition. A number of previous studies have documented strong relations between certain basin characteristics and ANC concentrations in surface waters (Berg et al., 2005; Nanus et al., 2005; Rutkowski et al., 2001; Sueker et al., 2001; Clow and Sueker, 2000; Melack et al., 1985; Corbin, 2004). For example, higher lake elevation was found to be a predictor of lower surface water ANC concentrations in two Colorado Wilderness Areas (Turk and Adams, 1983; Turk and Campbell, 1987), in Yellowstone and Grand Teton National Parks (Nanus et al., 2005), and in the Swiss Alps (Drever and Zorbrist, 1992).

Few studies have explored relations between basin characteristics and NO_3, SO_4, and base cations. Clow and Sueker (2000) found that steep slopes were inversely related to ANC and positively related to NO_3 concentrations in surface waters in

Rocky Mountain National Park, and attributed this to fast hydrologic flushing rates on steep slopes because of their high rate of water transmission in the poorly developed soils and limited vegetative cover. Rutkowski et al. (2001) found that bedrock geology and elevation were significant predictors of ANC in surface waters in wilderness areas of Nevada, Idaho, Utah, and Wyoming. Bedrock geology was significantly related to ANC and NO_3 in Rocky Mountain National Park (Clow and Sueker, 2000), ANC and SO_4 in Grand Teton National Park (Corbin et al., 2004)**,** and ANC in the Sierra Nevada (Melack et al., 1985). The presence of carbonate lithology in a basin can be very effective in neutralizing acidity in high-elevation basins and generally results in elevated concentrations of ANC (Berg et al., 2005). In the Southern Alps, Marchetto et al. (1994) found that the main factors influencing water chemistry in high alpine lakes were the weathering of silicate and calcite and nitrate uptake by vegetation, which accounted for most of the alkalinity production.

Evaluation of sources and effects of DIN deposition in high-elevation lakes

There is considerable uncertainty in the emission source areas and types that contribute to deposition of DIN, which can adversely affect sensitive aquatic habitats of high-elevation lake basins (Burns, 2004). Federal and state resource managers are investigating policy options to alleviate this problem by reducing anthropogenic emissions of NOx and NH_3. However, identifying source areas and emission types is complicated. Isotopic tracers of N measured in precipitation and water samples show promise in identifying these emission sources (Elliott et al., 2007; Kendall, 1998). Previously published studies indicate $\delta^{15}N$ (NO_3) values in NO_x emissions from coal-fired power plants have higher isotopic values ranging from +6 to +13 permil (Heaton, 1990; Kiga et al., 2000), compared with negative $\delta^{15}N$ (NO_3) values from vehicle NO_x emissions in tailpipe exhaust ranging from -13 to -2 permil (Heaton, 1990). The following $\delta^{15}N$ (NO_3) values have also been reported for vehicle NO_x emissions in tailpipe exhaust (+3.7 permil), and roadside vegetation (+3.8 permil) (Ammann, 1990; Moore, 1977; Pearson et al., 2000). The use of these N isotopes has been limited in part because analytical techniques for nitrate isotopes required large sample volumes that made it logistically difficult to sample in areas with topographically complex terrain. The denitrifier method to determine the dual isotopic composition ($\delta^{15}N$ and $\delta^{18}O$) of NO_3 is well suited for studies of NO_3 contributions to streams and lakes (Ohte et al., 2004) and only requires 20-60 nmol of NO_3 (Sigman et al., 2001; Casciotti et al., 2002).

Atmospheric DIN may be deposited directly to terrestrial or aquatic systems in a watershed. In a watershed, the transformations that occur in soils, plants, and microbial activity greatly influence the form and concentration of N that can eventually reach surface waters (Stoddard, 1994). In the Rocky Mountains of Colorado, researchers have shown that elevated levels of atmospheric DIN deposition has caused considerable changes in the state and function of terrestrial and aquatic ecosystems in high-elevation basins (Baron, 2000; Burns, 2003; Campbell et al., 2000; Campbell et al., 2002; Mast et al., 2002, Williams et al., 1996). In one study, inorganic N retention of DIN in wetfall averaged 72% in high-elevation ecosystems (Sickman et al., 2002). Nitrogen (N) deposition in excess of the total combined plant and microbial demand can cause watershed N saturation and increased rates of N leaching from soils to aquatic ecosystems (Aber et al., 1989). As N saturation advances, biogeochemical responses accompany increased nitrate leaching to surface waters including elevated rates of N-mineralization and nitrification and increased fluxes of nitrous oxide gas (N_2O) from soils (Aber et al., 1989, Galloway et al., 2003). Although originally developed for forested catchments, recent research has shown that the N saturation model also describes the patterns of N leaching observed in alpine ecosystems (Williams et al., 1996; Fenn et al., 1998; Sickman et al., 2002) and this is occurring in high-elevation watersheds in the Colorado Rockies (Williams et al., 1996; Campbell et al., 2000; Baron, 2006). This excess N can result in ecological effects in surface waters, including acidification and eutrophication. Eutrophication increases primary productivity in lakes and streams and alters diatom species distributions that form the base of the food web in many high-elevation lakes (Wolfe et al., 2001). Bergstrom and Jannson (2006) show that in high DIN deposition regions across the Northern hemisphere, elevated lake DIN concentrations have resulted in eutrophication and increased biomass of phytoplankton.

Increased aquatic productivity resulting from eutrophication accelerates the accumulation of organic matter in the water column and in lake sediments. Decomposition of this organic matter promotes hypoxia in lakes when they are ice-covered during winter and may adversely affect fish populations (Vitousek et al., 1997). In Rocky Mountain National Park, mineralization of organic N in pond sediments has caused concentrations of dissolved ammonia in vernal ponds to reach levels that may be harmful to threatened amphibians (Campbell et al., 2004). Although N deposition is greatest in the Front Range of Colorado, other high-elevation sites in the Rocky Mountains also show symptoms of early-stage N saturation. The progression towards N saturation is expected to continue as N deposition continues at current or higher levels in the future (Campbell et al., 2003). Changes in the water quality of the headwater systems affect not only National Park fish, wildlife and ecosystem integrity, but also downstream ecosystems and water users.

Tools for relating basin characteristics and solute chemistry

Monitoring of surface-water quality in high-elevation watersheds is necessary to assess current conditions and evaluate the long-term effects of acidic deposition to these aquatic ecosystems. However, these environments are often remote and, thus, costly and logistically difficult to monitor long-term changes in the chemical and biological composition of these lakes across vast regions of the US Rockies. Therefore, scientifically defensible predictive models of lake chemistry are needed tools towards best management of these sensitive ecosystems. Previous studies indicate that it may be possible to develop statistical models that use basin characteristics as explanatory variables to predict solute concentrations in alpine and subalpine lakes (Clow and Sueker, 2000; Corbin, 2004). Clow and Sueker (2000) predicted solute concentrations in Rocky Mountain National Park using multiple linear regression and found that predicted concentrations versus actual concentrations yielded an $r^2 = 0.92$ for ANC, an $r^2 = 0.97$ for NO_3, an $r^2 = 0.54$ for SO_4 and an $r^2 = 0.86$ for base cations. However, when predicted values were compared with 1985 Western Lake Survey data less than 35% of the variance was explained (Clow and

Sueker, 2000). Berg et al. (2005) found that the general linear models used to predict ANC concentration in unsampled high-elevation lakes of the Sierra Nevada explained 51% of the variation in observed ANC, with the majority of predicted lake ANCs greater than measured ANC. Corbin (2004) evaluated the predictive ability of multiple linear regression models in Grand Teton and found that while they worked well for ANC and base cations, the regression model did not work well for acidic anions and overestimated NO_3.

The combined use of multivariate logistic regression and GIS modeling holds promise to improve upon previous models and more accurately predict solute chemistry in alpine and subalpine lakes of the Rocky Mountains. This approach has been used in a number of water-quality investigations at sub-regional and regional scales to predict ground-water vulnerability to contaminants (Battaglin et al., 2003; Nolan, 2001; Rupert, 2003; Teso et al., 1996). The primary advantage of logistic regression over multiple linear regression and general linear models is that the binary response can be established using a meaningful threshold, such as a targeted background concentration or laboratory detection level. The results of a logistic regression model are expressed in probability units of the predicted dependent variable in reference to the specified threshold (such as below an ANC threshold), rather than expressed as a predicted discrete value of ANC concentration, such as determined using linear regression. Because the model results from multivariate logistic regression are expressed in probability units of the predicted dependent variable in reference to the specified threshold, resource-managers can develop monitoring programs based on any particular level of acceptable resource management risk. Successful application of the GIS modeling and multivariate logistic regression approach was demonstrated for alpine and subalpine lakes in Grand Teton National Park and Yellowstone National Park (Nanus et al., 2005); however, an assessment of lake sensitivity to acidic deposition across the broader Rocky Mountain region has not been conducted.

Organization of Thesis

Chapters 2-4 present related studies conducted in five national parks in the Rocky Mountains to assess the sources and effects of acidic deposition on alpine and subalpine lake solute chemistry across a regional scale. A brief overview of each chapter is discussed here, including how they relate to each other within the broader research framework.

Chapter 2 addresses the hypothesis that modeling lake sensitivity to acidic deposition in alpine and subalpine watersheds using a scientifically defensible approach that couples the controlling hydrologic and biogeochemical processes of acidification with empirically-based modeling techniques can accurately predict lake sensitivity across a regional scale. To test this hypothesis, lake sensitivity to acidification from atmospheric deposition of acidic solutes was evaluated based on statistical relations between ANC concentrations and landscape attributes that are quantified by GIS. A Rocky Mountain regional model was developed using multivariate logistic regression to evaluate regional lake sensitivity and for inter-park comparison. Individual models for each national park were also developed. To crossvalidate the regional ANC model, 58 randomly selected lakes from all five parks were sampled during late summer through early Fall 2004.

Chapter 3 presents the results of testing the hypothesis that high-elevation lakes identified as sensitive to acidic deposition in the Rocky Mountain region are showing a response to atmospheric inputs of NO_3. The spatial distribution of $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) in the 58 lakes sampled in 2004 was compared with the isotopic composition of precipitation collected at nearby National Atmospheric Deposition Program (NADP) National Trends Network (NTN) sites. To evaluate NO_3 sources, both $\delta^{15}N$ and $\delta^{18}O$ of NO_3 were measured using the denitrifier method on samples collected from lakes and precipitation that span a range of NO_3 deposition. This is the first comprehensive evaluation of NO_3 deposition sources using NO_3 isotopes to understand the sources of NO_3 in lakes of the Rocky Mountains.

Using lakes identified as sensitive to acidic deposition (Chapter 2) and regional patterns in NO_3 sources (Chapter 3), relations between GIS-based landscape and acidic deposition attributes and lake concentrations of NO_3, SO_4, and sum of base cations are evaluated in Chapter 4. Chapter 4 presents a novel study that includes estimates of DIN deposition and SO_4 deposition in model development and evaluates relations between these solutes and basin characteristics across an entire region in a systematic approach. Chapter 5 presents recommendations for future monitoring based on findings in Chapters 2-4. These findings may be used to improve long-term monitoring designs for alpine and subalpine lakes in the Rockies and provide a framework for assessing other high-elevation environments of the Western US and the World. As a whole, these studies provide new insight and understanding into the important controls on sources and effects of acidic deposition on solute chemistry in alpine and subalpine lakes across a regional scale.

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CHAPTER 2

ASSESSMENT OF LAKE SENSITIVITY TO ACIDIC DEPOSITION

ABSTRACT

The sensitivity of high-elevation lakes to acidic deposition was evaluated in five national parks of the Rocky Mountains based on statistical relations between lake acid-neutralizing capacity concentrations and basin characteristics. Acid-neutralizing capacity (ANC) of 151 lakes sampled during synoptic surveys and basin-characteristic information derived from Geographic Information System (GIS) data sets were used to calibrate the statistical models. The explanatory basin variables that were considered included topographic parameters, bedrock type, soil type, and vegetation type. A logistic regression model was developed and modeling results were cross-validated through lake sampling during fall 2004 at 58 lakes. The model was applied to lake basins greater than 1 hectare in area in Glacier National Park (n=244), Grand Teton National Park (n=106), Great Sand Dunes National Park and Preserve (n=11), Rocky Mountain National Park (n=114), and Yellowstone National Park (n=294). Lakes that had a high probability of having an ANC concentration less than 100 μeq/L, and therefore are sensitive to acidic deposition, are located in basins with elevations greater than 3,000 meters, with less than 30% of the catchment having NE aspect, and with greater than 80% of the catchment bedrock having low buffering capacity. The modeling results indicate that the most sensitive lakes are located in Rocky Mountain National Park and Grand Teton National Park. This technique for evaluating the lake sensitivity to acidic deposition is useful for designing long-term

monitoring plans and is potentially transferable to other remote mountain areas of the United States and the world.

INTRODUCTION

Population growth, water use, and energy development in the Western U.S. are affecting natural resources and environments (Baron, 2002). Alpine and subalpine ecosystems in Glacier National Park, Grand Teton National Park, Great Sand Dunes National Park and Preserve, Rocky Mountain National Park, and Yellowstone National Park are particularly vulnerable to natural and human-induced stressors (Williams et al., 2002). Physical characteristics of high-elevation basins, such as steep topography, thin, rocky soils, and sparse vegetation, and a short growing season make lakes particularly susceptible to contaminant inputs from atmospheric deposition (Turk and Spahr, 1991). Throughout the Rocky Mountain region, energy generation, transportation, industry, and agriculture, produce emissions of SO_2, NO_x, and NH_3 that may contribute to acidification and eutrophication of alpine and subalpine lakes. Atmospheric deposition of dissolved inorganic nitrogen (DIN = nitrate (NO_3) + ammonium (NH_4)) to high-elevation lakes has the potential to change the nutrient balance of aquatic ecosystems, increasing the possibility of episodic acidification and change in nutrient status (Baron, 2006; Williams et al., 1996).

Evaluation of existing water-quality data indicates that limited data are available for high-elevation lakes in many national parks of the Rocky Mountains (Corbin et al., 2006; Woods and Corbin, 2003a; Woods and Corbin, 2003b; Clow et al., 2002). Monitoring of surface waters is needed in national parks of the Rocky Mountain region to assess current conditions of aquatic ecosystems, and evaluate the long-term effects of atmospheric deposition of contaminants on these aquatic ecosystems. These environments are often remote, however, and it can be expensive and logistically difficult to monitor long-term changes in the chemical and biological composition of these lakes.

In the Rocky Mountains, there is concern that lakes with acid-neutralizing capacity (ANC) concentrations less than 100 μeq/L are particularly sensitive to atmospheric inputs of acidity (Williams and Tonnessen, 2000). ANC is a measure of the water's capacity to buffer acidic inputs and a measure of the concentration of solutes. Previous studies have documented the relation between basin characteristics and ANC concentrations in surface waters of mountain lakes in the Western U.S. (Berg et al., 2005; Nanus et al., 2005; Rutkowski et al., 2001; Sueker et al., 2001; Clow and Sueker, 2000; Melack et al., 1985; Corbin et al., 2006). Lake elevation was found to be a predictor of surface-water ANC concentrations in two Colorado Wilderness Areas (Turk and Adams, 1983; Turk and Campbell, 1987), Yellowstone and Grand Teton National Parks (Nanus et al., 2005), and in the Swiss Alps (Drever and Zorbrist, 1992). Clow and Sueker (2000) found that slope steepness was negatively correlated with lake ANC concentrations in Rocky Mountain National Park and attributed this to fast hydrologic flushing rates on steep slopes with poorly developed soils and limited vegetative cover. Rutkowski et al. (2001) found that bedrock geology and elevation were significant predictors of ANC in surface waters in wilderness areas of Nevada, Idaho, Utah, and Wyoming. Bedrock geology was a strong predictor of ANC in the Sierra Nevada (Melack et al., 1985), Rocky Mountain National Park (Clow and Sueker, 2000), and in Grand Teton National Park (Corbin et al., 2006). The presence of carbonate lithology in a basin can be very effective in neutralizing acidity in high-elevation basins and generally results in elevated concentrations of ANC (Berg et al., 2005). Berg et al. (2005) found that the ratio of lake perimeter to lake area was significantly related to ANC concentration, such that higher ratios resulted in higher lake ANC.

Results of the previous studies indicate that it is possible to develop statistical models that use basin characteristics as explanatory variables to predict ANC concentrations in alpine and subalpine lakes. Clow and Sueker (2000) predicted ANC concentrations in Rocky Mountain National Park using multiple linear regression and found that predicted ANC concentrations versus actual yielded an r-squared = 0.92. However, when predicted values were compared with 1985 Western Lake Survey data less than 35% of the variance was explained (Clow and Sueker, 2000). Berg et al. (2005) found that the general linear models used to predict discrete ANC concentration in unsampled high-elevation lakes of the Sierra Nevada explained 51% of the variation in observed ANC, with the majority of predicted lake ANCs greater than measured ANC.

To minimize the costs and resources associated with designing and implementing regional water-quality monitoring programs in high-elevation environments, a scientific approach to identify the most sensitive lakes without sampling each one is needed. The combined use of multivariate logistic regression and GIS modeling holds promise. This approach has been used in a number of water-quality investigations at sub-regional and regional scales to predict groundwater vulnerability to contaminants (Battaglin et al., 2003; Nolan, 2001; Rupert, 2003; Teso et al., 1996). The primary advantage of logistic regression over multiple linear regression and general linear models is that the binary response can be established using a meaningful threshold, such as a targeted background concentration or laboratory detection level. The results of a logistic regression model are expressed in probability units of the predicted dependent variable in reference to the specified threshold (such as below an ANC threshold), rather than expressed as a predicted discrete value of ANC concentration, such as determined using linear regression. Because the model results are expressed in probability units, resource-managers can develop monitoring programs based on any particular level of acceptable resource management risk. Successful application of the GIS modeling and multivariate logistic regression approach was demonstrated for alpine and subalpine lakes in Grand Teton National Park and Yellowstone National Park (Nanus et al., 2005); however, an assessment of lake sensitivity to deposition across the broader Rocky Mountain region has not been conducted.

In 2004, the U.S. Geological Survey, in cooperation with the National Park Service, began a study to evaluate the sensitivity of alpine and subalpine lakes in national parks of the Rocky Mountains to atmospheric deposition of contaminants, based on statistical relations between water-quality and landscape attributes that are quantified by GIS. A Rocky Mountain regional model was developed to evaluate regional lake sensitivity and for inter-park comparison. Individual models for each national park were then developed to identify and quantify the number of lakes that are sensitive to acidic deposition. For cross-validation of the regional model, 58 randomly selected lakes from all five parks were sampled during late summer through early fall 2004. The approach developed in this study for identifying deposition-sensitive lakes in remote areas that may need to be monitored could be used as a framework for application to other high-elevation environments of the Western U.S. and the World.

Study Area

Five national parks in the Rocky Mountains were studied, including Glacier National Park (GLAC), Montana; Grand Teton National Park (GRTE), Wyoming; Great Sand Dunes National Park and Preserve (GRSA), Colorado; Rocky Mountain National Park (ROMO), Colorado; and Yellowstone National Park (YELL), Wyoming. Four of the five parks are located along the Continental Divide (Figure 2-1, Table 2-1,) and have a large number of alpine and subalpine lakes, ranging in elevation from less than 1,000 m in GLAC to more than 3,500 m in ROMO (Table 2-2). These national parks are in glaciated mountain terrain (Madole, 1976; Richmond and Fullerton, 1986). Dominant bedrock types for each park are as follows: sedimentary rocks in GLAC (U.S. Geological Survey, 1992 b), granitic rocks in GRTE U.S. Geological Survey, 1992 a) and ROMO (U.S. Geological Survey, 1990), volcanic rocks in YELL (U.S. Geological Survey, 1988), and granitic and sedimentary rocks in GRSA (National Park Service, 2004). Average annual precipitation amounts in the Rocky Mountains increase as a function of elevation and latitude, mountainous areas above 3,000 m elevation generally receive at least 800

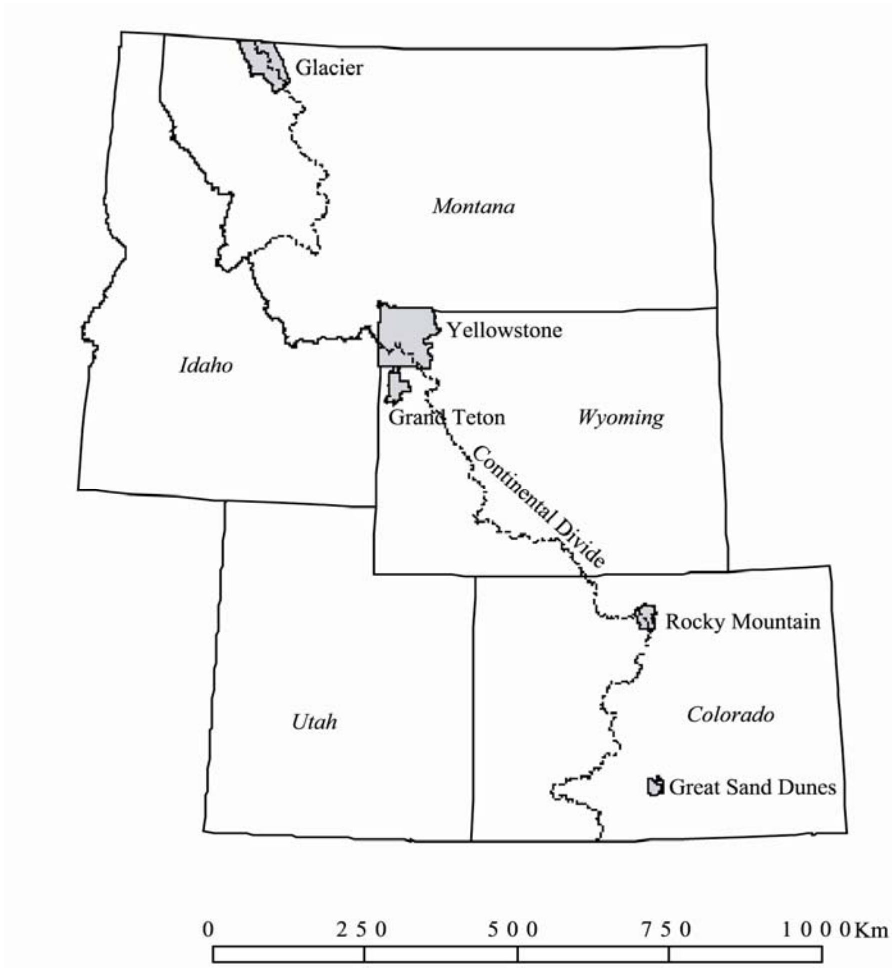

Figure 2-1. Location of National Parks included in this study.

Table 2-1. National Parks in the Rocky Mountain region and their characteristics.

	Glacier	Yellowstone	Grand Teton	Rocky Mountain	Great Sand Dunes
Area (km^2)	4,101	8,980	1,256	1,078	343
Maximum Land-Surface Elevation (m)	3,190	3,122	4,198	4,346	4,317
Number of Lakes	244	294	106	114	11
Dominant Bedrock Types	Sedimentary	Volcanic	Granite	Granite	Granite and Sedimentary

Table 2-2. Select physical and chemical characteristics of lakes used in regional model development.

		ANC (μeq/L)	Lake Elevation (m)	Bedrock Geology Low Buffering Capacity (%)	Aspect Northeast (%)
Glacier					
n = 33	Minimum	39	956	0	0
	1st quartile	326	1500	7.8	11
	Median	853	1692	35	14
	3rd quartile	1111	1826	67	18
	Maximum	1550	2605	100	35
Yellowstone					
n = 23	Minimum	54	1953	0	4
	1st quartile	170	2200	0	10
	Median	320	2322	0	17
	3rd quartile	702	2394	6	25
	Maximum	1621	2674	25	33
Grand Teton					
n=52	Minimum	18	2035	0	0
	1st quartile	69	2104	70	12
	Median	108	2805	89	18
	3rd quartile	270	2958	100	23
	Maximum	1600	3247	100	49
Rocky Mountain					
n = 40	Minimum	15	1646	50	0
	1st quartile	44	3252	88	7
	Median	55	3316	97	16
	3rd quartile	84	3478	100	21
	Maximum	214	3625	100	37
Great Sand Dunes					
n = 3	Minimum	228	3496	100	7
	1st quartile	294	3503	100	15
	Median	361	3509	100	23
	3rd quartile	392	3642	100	24
	Maximum	423	3774	100	24
Regional					
n = 151	Minimum	15	956	0	0
	1st quartile	63	2070	21	11
	Median	122	2715	81	16
	3rd quartile	583	3140	98	23
	Maximum	1621	3774	100	49

mm precipitation per year, most of which accumulates in a seasonal snowpack (Spatial Climate Analysis Service, 2000).

Many of the lakes used in model calibration and validation are located in alpine and subalpine terrain where soils are poorly developed, vegetation is sparse, and growing seasons are short. The lake selection process included a random selection component similar to the Western Lake Survey (Kanciruk et al., 1987; Landers et al., 1987; Silverstein et al., 1987), to allow extrapolation of the results to a broad population of lakes. Lakes with a surface area greater than 1 hectare (ha) were used to avoid inclusion of small tarns and ponds in calibration and validation data sets. Model results were then applied to all lakes greater than 1 ha in area, 769 lakes in the five parks included in this study.

METHODS

Description of Data Used in the Statistical Models

ANC concentrations for 151 alpine and subalpine lakes, approximately 20 % of the total number of lakes, from the five national parks were used to calibrate the logistic regression models. The focus was on baseflow conditions when ANC concentrations are relatively consistent over time; therefore, only available data from lakes that were sampled during late summer through early fall were included. Few data were available for YELL (Woods and Corbin, 2003b), GLAC (Landers et al., 1987), and GRSA (Bunch, GRSA, written commun., 2004). More extensive water-quality information exists for alpine and subalpine lakes in ROMO (Clow et al., 2002) and GRTE (Gulley and Parker, 1985; Landers et al., 1987; Woods and Corbin, 2003a; Corbin et al., 2006; Tonnessen and Williams, NPS and INSTAAR, written commun., 1997), than for the other three parks in this study. Chemical concentrations in high-elevation lakes in these five parks were sampled in 1985, and then again in 1999, and it was determined that the only national park with significant changes in baseflow ANC concentrations was ROMO (Clow et al., 2003). For ROMO, ANC concentrations were not variable across the concentration thresholds used for this study. These results indicate that data collected at the five parks may be combined for sites with multiple years of ANC concentration data from one data set or multiple data sets.

Lakes were sampled during late summer through early fall 2004 in all five parks for cross-validation of the statistical models. Samples were collected at 58 randomly selected high-elevation lakes that are spatially distributed within each of the national parks (Nanus et al., 2005). Lake samples were filtered (0.45 micron) and collected at the lake outlets and chemical concentrations were analyzed following standard procedures at the Kiowa Environmental Chemistry Laboratory, Boulder, Colorado (Seibold, 2001), which specializes in analysis of extremely dilute waters such as those found in the study area.

Physical characteristics that were tested in the statistical model for correlation with ANC concentrations included elevation, slope, aspect, bedrock geology, and soils. Boundaries for the watershed of each lake were delineated using a 10-m digital elevation model (DEM) (U.S. Geological Survey, 2000 a, b). To evaluate the effect of error associated with basin delineation on the model, basin boundaries using multiple methods, including manual and automated in GIS, were compared and the difference in model outputs were calculated at less than 5%.

For each lake basin, the 10-m DEM was used to calculate mean elevation, slope, basin area, lake/basin area, percentage of steep slopes (slopes >30 degrees), and percentage of aspect (by 45-degree increments). Additional basin characteristics that were derived from National Park Service GIS data include watershed area, lake area, ratio of lake area to watershed area, ratio of lake perimeter to lake area, percentage watershed covered by rock type (National Park Service 2004; U.S. Geological Survey, 1988, 1990, 1992 a, b), percentage watershed covered by soil type (National Park Service, 1994a, 1997, 1999; Rocky Mountain National Park GIS Program, 1995 a), and percentage watershed covered by vegetation type (National Park Service, 1981, 1990, 1994 b; Rocky Mountain National Park GIS Program, 1995 b). Errors associated with a given resolution for each basin characteristic were evaluated to determine whether data with different levels of resolution could be combined.

Bedrock lithologies were grouped into six different geochemical classes that were ranked from low to high on the basis of buffering capacity of the bedrock (Nanus and Clow, 2004; Nanus et al., 2005). The geochemical rankings (GC) are as follows: GC 1: Low buffering capacity: gneiss, quartzite, schist, granite; GC 2: Moderate buffering capacity: andesite, dacite, diorite, phyllite; GC 3: High buffering capacity: basalt, gabbro, greywacke, argillite, undifferentiated volcanics; GC 4: Very high buffering capacity: amphibolite, hornfels, paragneiss, undifferentiated metamorphic rocks; GC 5: Class 5, Extremely high buffering capacity:

metacarbonate, marine sedimentary rocks, calc silicate and basic intrusive rocks; and GC 6: Class 6, Unknown buffering capacity. Vegetation type was classified into low, medium, and high classes based on the basins sensitivity to acidic deposition (Nanus et al., 2005). Basins with high sensitivity include a large percentage of snow, ice, rock, and water. Basins with medium sensitivity include forest and tundra, and those with low sensitivity include subalpine meadow. Soil type was not included in the regional model, because not all parks had complete digital soil data, and the data that were available were not comparable across all five parks. However, soil type was included for individual park models at GLAC, GRTE, ROMO, and YELL.

Atmospheric factors, including precipitation amount and atmospheric deposition rates of acidic solutes were evaluated for inclusion in the statistical models, following protocols presented in Nanus et al. (2003). The mean-annual precipitation variable for each basin was derived from a 30-year (1961-1990) average annual precipitation grid (PRISM) (Spatial Climate Analysis Service, 2000). Average annual deposition loadings of hydrogen ion, inorganic N, and sulfate were determined by multiplying the precipitation grid with kriged chemical concentrations (Nanus et al., 2003), these deposition estimates were then calculated for each basin for inclusion in the statistical analysis. The spatial variability in solute deposition was largely controlled by precipitation amount (Nanus et al., 2003).

Model Development and Validation

Multivariate logistic regression was used to predict lakes with a high probability of sensitivity to atmospheric deposition for the Rocky Mountain region and for each of the five parks. Logistic regression differs from linear regression in that the result is the probability of being above or below a threshold, rather than a predicted value (Helsel and Hirsch, 1992). The general linear model relates continuous dependent variables to independent variables that are either classification variables or continuous variables (SAS Institute, 1990). This works well if the assumptions of linear regression are satisfied. However, with a dichotomous (binary) dependent variable, assumptions of linearity, normality of the error term, and homoscedasticity (constant variance of the error term) are violated (Menard, 2002).

The logistic regression approach uses the maximum likelihood method to fit linear logistic regression models for binary response data (SAS Institute, 1999). The probability (p) of being in a response category is defined by the odds ratio, the log of which transforms a variable between 0 and 1 into a continuous variable that is a linear function of the explanatory variables (Helsel and Hirsch, 1992) as follows:

$$\ln\left(\frac{p}{1-p}\right) = b_o + b_x \qquad \text{(eqn. 1)}$$

where b_o is the intercept, x is a vector of k independent variables, and b_x includes the slope coefficients for each explanatory variable. The logistic transformation is used to return the predicted values of the response variable to probability units, with the logistic regression model as follows:

$$Logit(P) = \frac{e^{(b_o + b_x)}}{1 + e^{(b_o + b_x)}} \qquad \text{(eqn. 2)}$$

where *Logit (P)* is the probability that the ANC concentration is less than a specified ANC concentration threshold (binary response) (Helsel and Hirsch, 1992).

Multivariate logistic regression models for all five national parks in the Rocky Mountains were developed for sensitivity thresholds with ANC concentrations of 50 μeq/L, 100 μeq/L, and 200 μeq/L. Surface waters with ANC concentrations less than 50 μeq/L have been defined as sensitive to the effects of atmospheric deposition (Herlihy et al., 1993). ANC concentrations greater than 200 μeq/L have been defined as insensitive to acidification (Schindler, 1988; Camarero et al., 1995; Sullivan et al., 2004). ANC concentrations of about 100 μeq/L provide an intermediate threshold that represents moderate sensitivity (Williams and Tonnessen, 2000). For each of the three ANC thresholds, probabilities for sensitive lakes were calculated from 0-100%. For ease of presentation, the results were binned into three groups representing: 0-33% (low probability), 33-66% (medium probability), and 66-100% (high probability).

Basin-characteristic information derived from GIS was used as explanatory variables, and existing ANC concentration data (n=151) were used as the dependent variables to calibrate the regression models for the identification of sensitive lakes. First, all explanatory variables were tested independently using univariate logistic regression, and the explanatory variables that have significant influence at p-value $\leq$ 0.1 were tested in the multivariate logistic regression models. A p-value of $\leq$ 0.1, was chosen over a p-value $\leq$ 0.05 so that more variables could be included in the multivariate analysis (Hosmer and Lemeshow, 1989).

To evaluate the calibration of the logistic regression models for each national park, model-based predicted probabilities were compared to measured concentrations by using the Hosmer-Lemeshow (HL) goodness-of-fit test (Hosmer and Lemeshow, 1989). A subset of lakes were not included in the calibrations so as to provide an independent data set to evaluate the calibration results. To evaluate model agreement, measured ANC concentrations were compared to predicted ANC concentrations by randomly grouping the verification lakes with measured ANC into 10 groups with an equal number of lakes. These random groupings of 10 % were used to evaluate model agreement between measured and predicted ANC concentration. A higher HL value indicates a well-calibrated model (Hosmer and Lemeshow, 1989). The *c*-statistic is a measure of rank correlation of ordinal variables (SAS Institute, 1990). The *c*-statistic is normalized so that it ranges from 0 (no association) to 1 (perfect association). It is a variant of Somers' D index (SAS Institute, 1990). The multivariate logistic regression model with the best statistical outcome with respect to predicting probability measured by r-squared, the *c*-statistic, and the HL goodness-of-fit test is presented for the Rocky Mountain region and for individual parks in the results section.

The resulting multivariate logistic regression models were applied to all lakes greater than or equal to 1 ha in the five parks. Modeling results were evaluated through samples collected from lakes in all five parks during the fall of 2004 for cross-validation of the results. Only the results of the regional model were cross-validated. The individual park models were not cross-validated due to the lack of validation data within each park. For the regional model, the actual percentage of lakes with ANC less than 100 μeq/L was calculated for the calibration (n=151) and validation (n= 58) set of data separately and is equal to the number of lakes with measured ANC less than 100 μeq/L divided by the total number of analyses for each 10 % of decile data.

RESULTS

ANC of the 151 lakes used for the model calibration ranged widely, from 15 to 1,621 μeq/L (Table 2-2). The ANC concentrations also differed among parks (Figure 2-2). The maximum ANC concentration of 214 μeq/L for ROMO (n=40) was below the median concentration of 853 μeq/L for GLAC (n=33) and 320 μeq/L for YELL (n=23) (Table 2-2). The amount of bedrock composed of low buffering capacity also varied widely among the parks. The maximum amount of low-buffering capacity bedrock in lake basins of YELL was 25%, in contrast to ROMO where the minimum amount of low-buffering capacity bedrock was 50% and the median amount was 97%. In general, lakes in parks with low ANC (ROMO, GRTE) were associated with large amounts of low-buffering capacity bedrock.

Concentrations of ANC were not normally distributed, with a median concentration of 122 μeq/L (Figure 2-3, Table 2-2). Therefore, the assumptions underlying multiple linear regression, such as normal distribution, were not satisfied. Because logistic regression is used to explore the relations between binary response and a set of explanatory variables and does not require a normal distribution, it is well-suited for the ANC data.

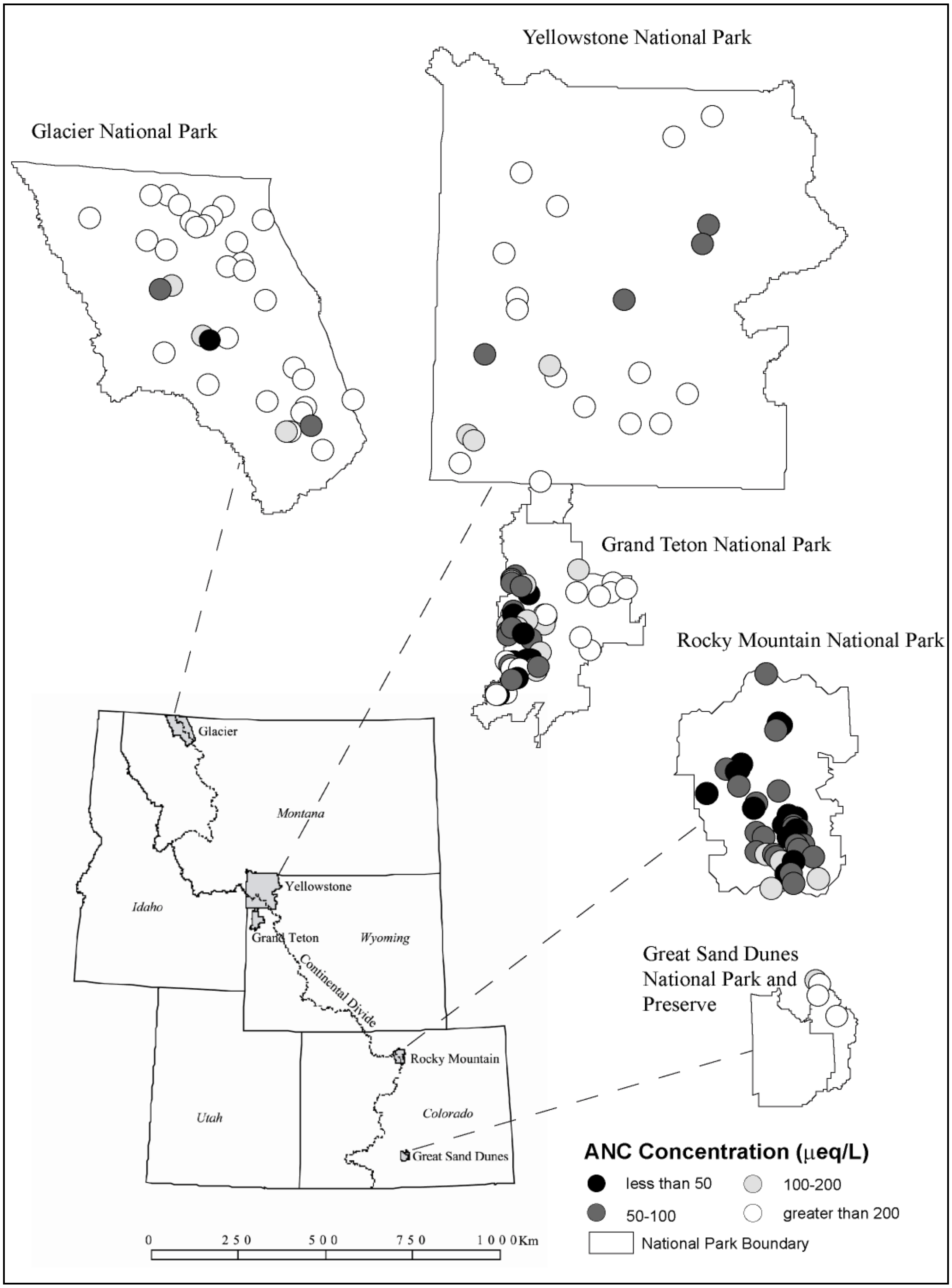

Figure 2-2. ANC concentrations from lakes used for the regional model calibration (n=151) (Gulley and Parker, 1985; Landers et al., 1987; Clow et al., 2002; Woods and Corbin, 2003a,b; Corbin et al., 2006; Tonnessen and Williams, NPS and INSTAAR, written commun., 1997). The data are binned into four groups based on the thresholds for our ANC modeling: ANC < 50 μeq/L, ANC ranging from 50-100 μeq/L, ANC ranging from 100-200 μeq/L, and ANC > 200 μeq/L.

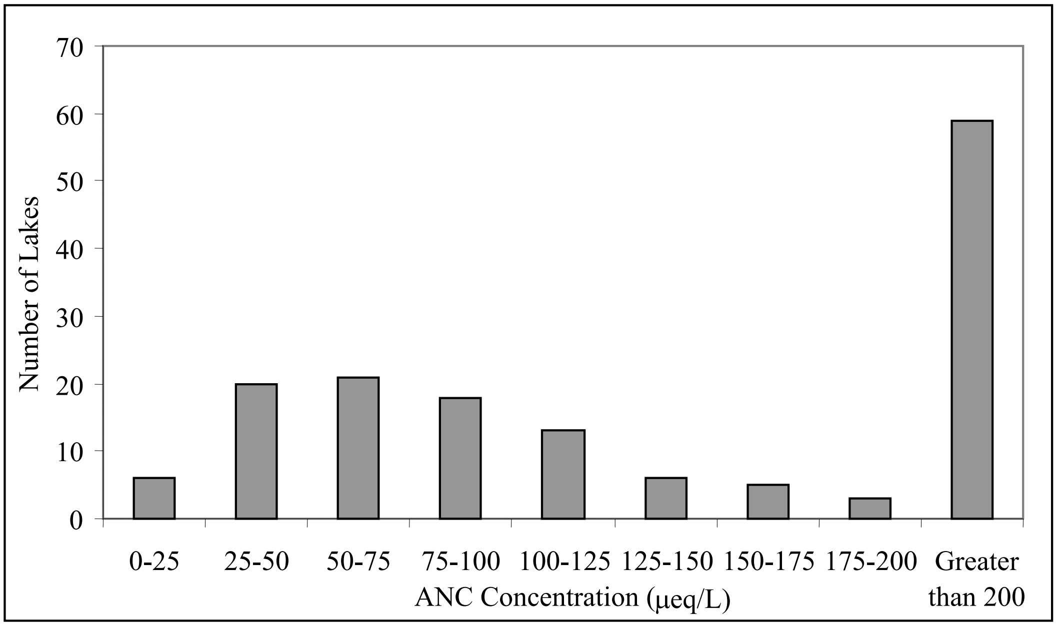

Figure 2-3. Frequency of ANC concentrations for Rocky Mountain lakes in 25 μeq/L increments, used in the regional model calibration.

Rocky Mountain Regional Lake Sensitivity Model

Results of the regional model calibration indicate that ANC concentrations for a threshold of less than 50 μeq/L are significantly related to the percentage of bedrock with low buffering capacity (*p*-value = 0.02) and elevation (*p*-value = 0.06). ANC concentrations less than 200 μeq/L are significantly related to the percentage of basin with low buffering capacity bedrock (*p*-value < 0.001), elevation (*p*-value = 0.004), and the ratio of lake-perimeter to lake-area (*p*-value = 0.04). About 43% of the lakes in the calibration data set have ANC concentrations less than 100 μeq/L (Figure 2-3). Variables that were significantly related to ANC concentrations less than 100 μeq/L in the regional model are as follows: elevation (*p*-value < 0.001), percentage of basin with low buffering capacity bedrock (p-value = 0.04), and percentage of basin with northeast aspects (*p*-value = 0.08) (Table 2-3).

Table 2-3. Results of multivariate logistic regression analyses, logistic regression coefficients, and *p*-values.

Multivariate Model (Number of lakes)	ANC (μeq/L) Threshold	Explanatory Variable: Name	Range	Median	Explanatory variable coefficient (p-value)	HL Goodness of fit (r^2)	c-statistic	Percent Correct (%)
Rocky Mountain (n=40)	50	Elevation at lake outlet (m)	2896-3625	3321	0.007 (0.07)	0.80	0.92	90
		Bedrock geology with low buffering capacity (%)	0-100	97	8.09 (0.06)			
		Soils with very slow infiltration and high runoff potential (%)	0-100	87	12.3 (0.03)			
Grand Teton (n=52)	50	Slopes greater than 30 degrees (steep slopes) (%)	0-100	37	13.8 (0.03)	0.70	0.90	85
		Soils Leighcan-Moran-Walcott (%)	0-100	1	9.70 (0.004)			
Glacier (n=33)	200	Elevation at lake outlet (m)	956-2298	1646	0.008 (0.1)	0.86	0.95	92
		Bedrock geology with high buffering capacity (%)	0-100	42	-12.0 (0.09)			
Yellowstone (n=23)	200	Elevation at lake outlet (m)	1953-2674	2359	0.002 (0.09)	0.61	0.78	69
		Bedrock geology with medium buffering capacity (%)	0-100	66	3.08 (0.05)			
Regional (n=151)	100	Elevation at lake outlet (m)	956-3774	2577	0.003 (<0.001)	0.98	0.90	81
		Bedrock geology with low buffering capacity (%)	0-100	62	1.65 (0.04)			
		Aspect northeast (0-45 degrees) (%)	0-100	17	-4.15 (0.08)			

The 100 μeq/L model had the best statistical outcome (Table 2-3) and the resultant probability equations were applied to the 769 lakes greater than 1 ha in the five national parks.

$$Logit(P) = \frac{e^{\left(-7.4 + (0.0025 \times elevation) - (4.2 \times \%ba\sin\ aspect\ 0{-}45\ \deg.) + (1.6 \times \%ba\sin\ with\ low\ buffering\ capacity\ bedrock)\right)}}{1 + e^{\left(-7.4 + (0.0025 \times elevation) - (4.2 \times \%ba\sin\ aspect\ 0{-}45\ \deg.) + (1.6 \times \%ba\sin\ with\ low\ buffering\ capacity\ bedrock)\right)}}$$

(eqn. 3)

where *Logit (P)* and *e* are defined in equation 2. The HL goodness-of-fit yielded an r-squared = 0.98, and the *c*-statistic = 0.90 indicating a good model fit to the calibration data (Table 2-3).

Results of the regional model indicate that 53% of lakes in ROMO, GRSA, and GRTE had a high probability (66-100%) for lake ANC concentrations less than 100 μeq/L (Figure 2-4). Few lakes in GLAC and YELL had a high probability for ANC concentrations less than 100 μeq/L (Figure 2-4). The lakes that had a high probability of having an ANC concentration less than 100 μeq/L in the regional model, are located at elevations above 3,000 m, with less than 30% of the basin with northeast aspect, and greater than 80% of the basin with bedrock of low buffering capacity (ie. quartzite, granite, gneiss, and schist).

Modeling results were cross-validated from 58 additional lakes sampled during 2004. The spatial distribution of ANC concentrations (Figures 2-5 and 2-6) indicate that ANC concentrations are lowest (more sensitive to deposition of atmospheric contaminants) in ROMO and GRTE and highest (less sensitive to deposition of atmospheric contaminants) in GLAC and YELL, similar to the calibration data set.

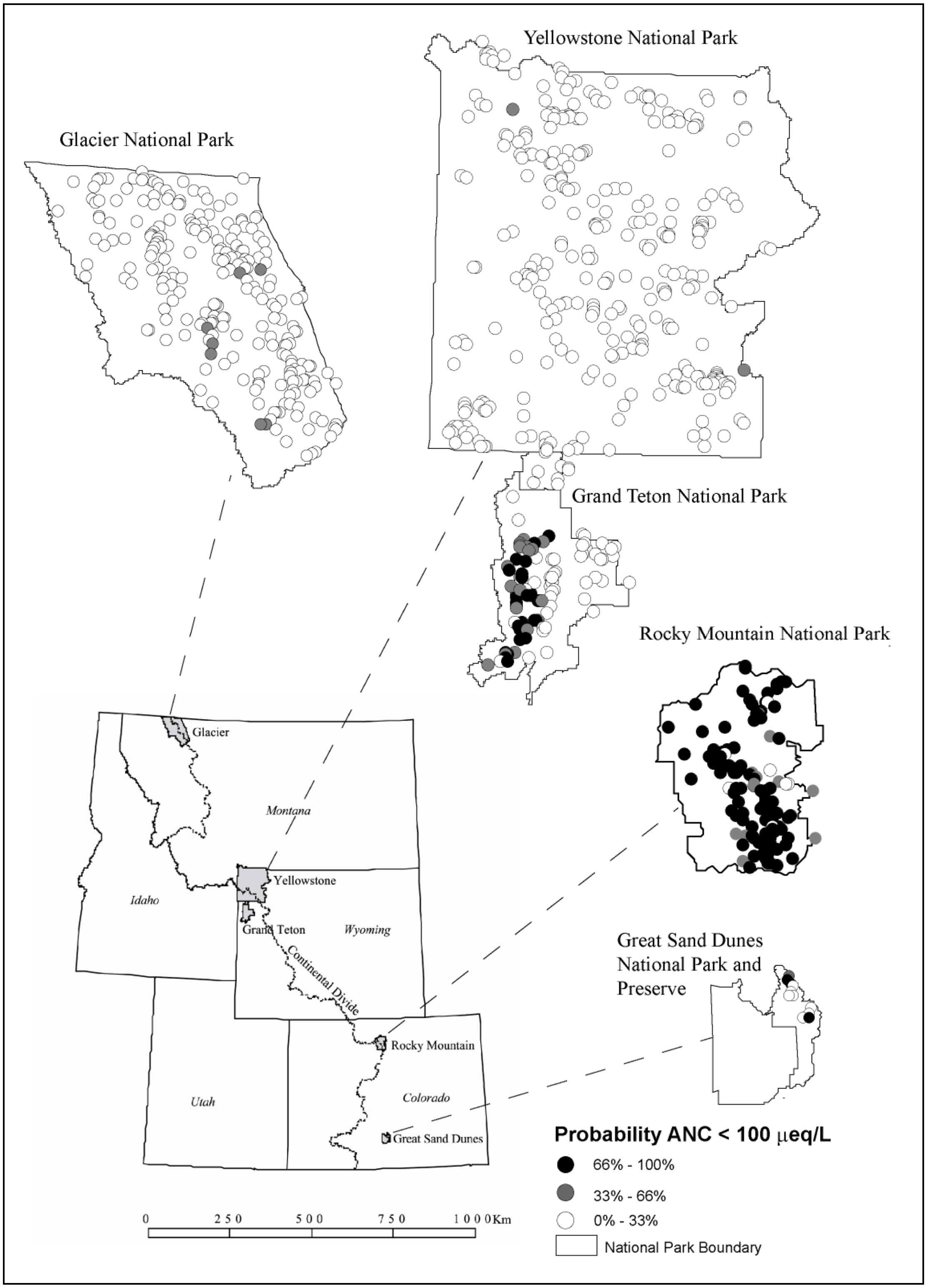

Figure 2-4. Probability of lake ANC concentrations less than 100 μeq/L in five Rocky Mountain National Parks. All probabilities ranging from 0-100% were evaluated; the data are binned into three groups for ease of presentation.

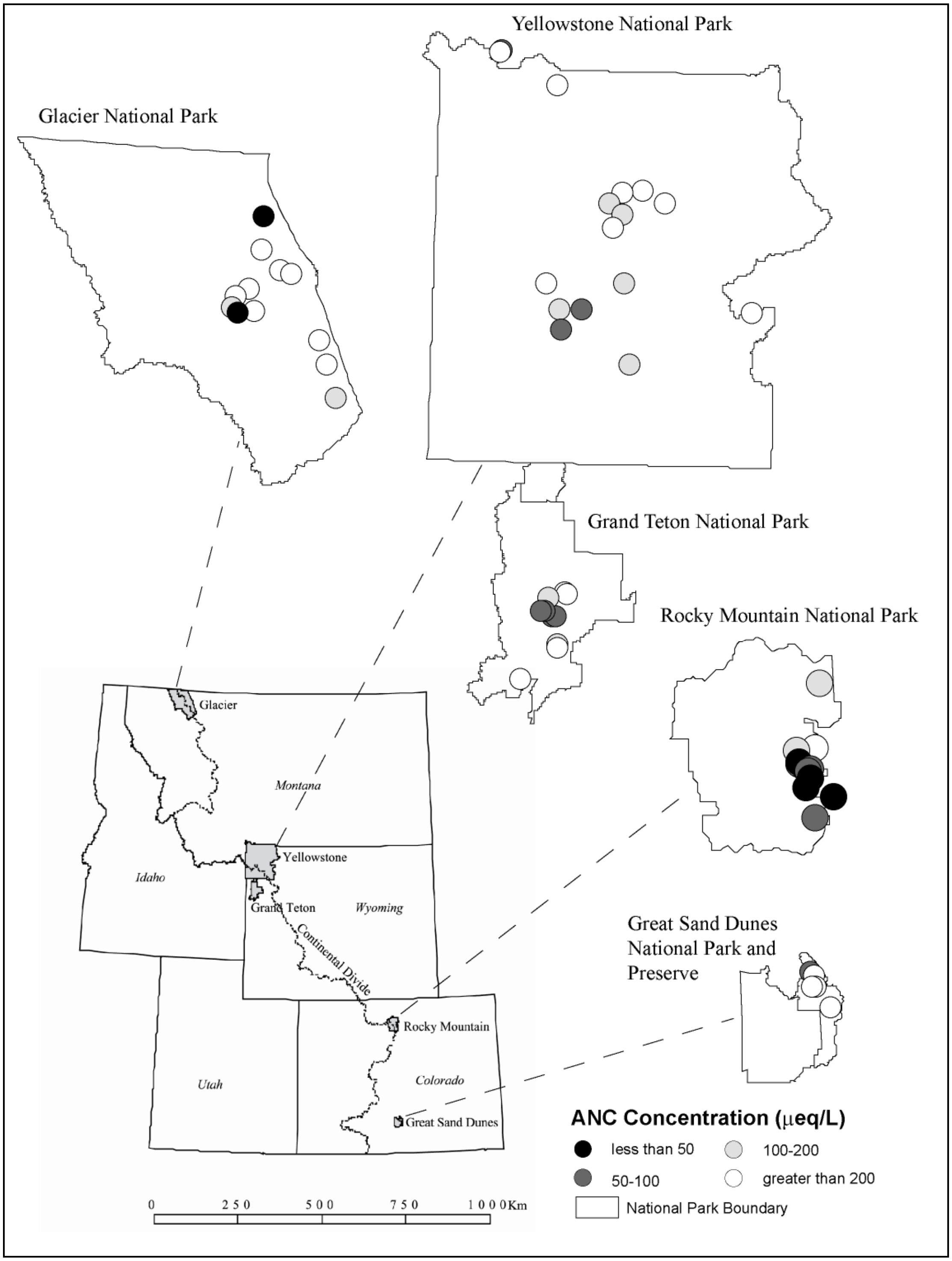

Figure 2-5. ANC concentrations from lakes used for validation of the regional model (n=58).

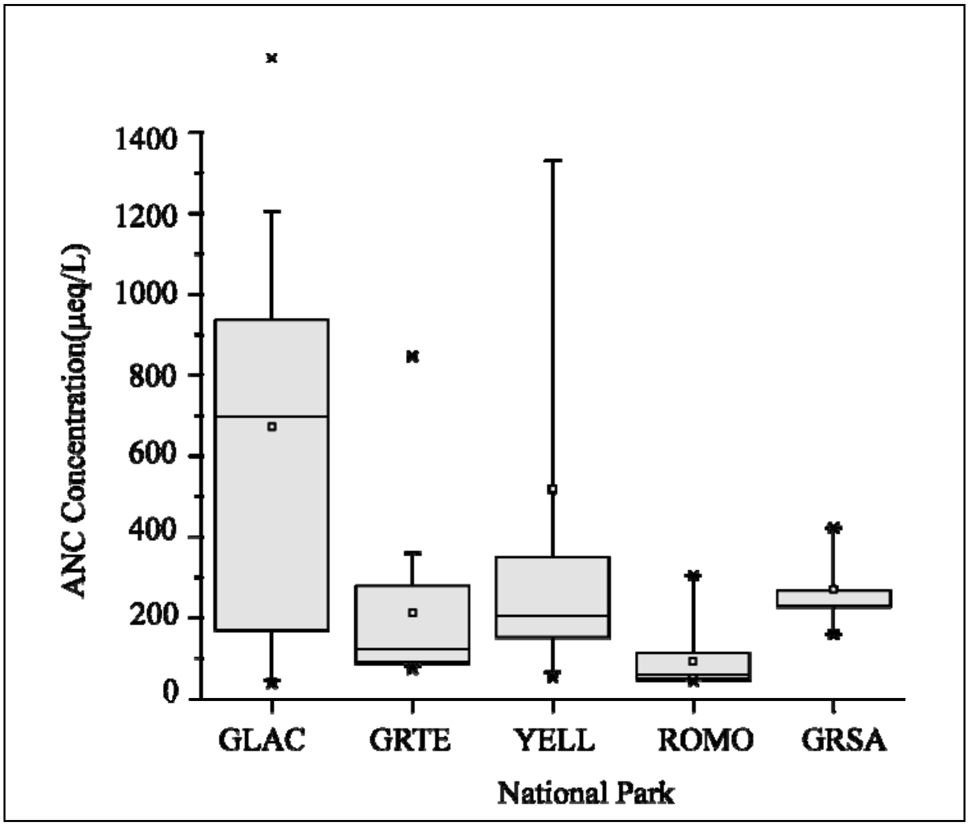

Figure 2-6. Box plot showing distribution of concentrations measured at 58 lakes during fall 2004.

Predicted probabilities for ANC concentrations less than 100 μeq/L were compared with measured ANC concentrations to evaluate predictive ability for the Rocky Mountain regional model. ANC concentrations from the measured lakes were converted to binary classification of "one" for ANC concentrations < 100 μeq/L and "zero" for ANC concentrations > 100 μeq/L. The conversion to binary classification enabled a direct comparison between the percentage of measured ANC concentrations and the average predicted probability within each 10 % decile of the data. For the calibration data set (n=151), the r-squared value is 0.98 for lakes with a predicted probability less than 100 μeq/L compared with actual percentage of lakes with ANC concentrations less than 100 μeq/L (Figure 2-7). For the validation data set (n=58), the r-squared value is 0.93 is very similar to the calibration value of 0.98 (Figure 2-8).

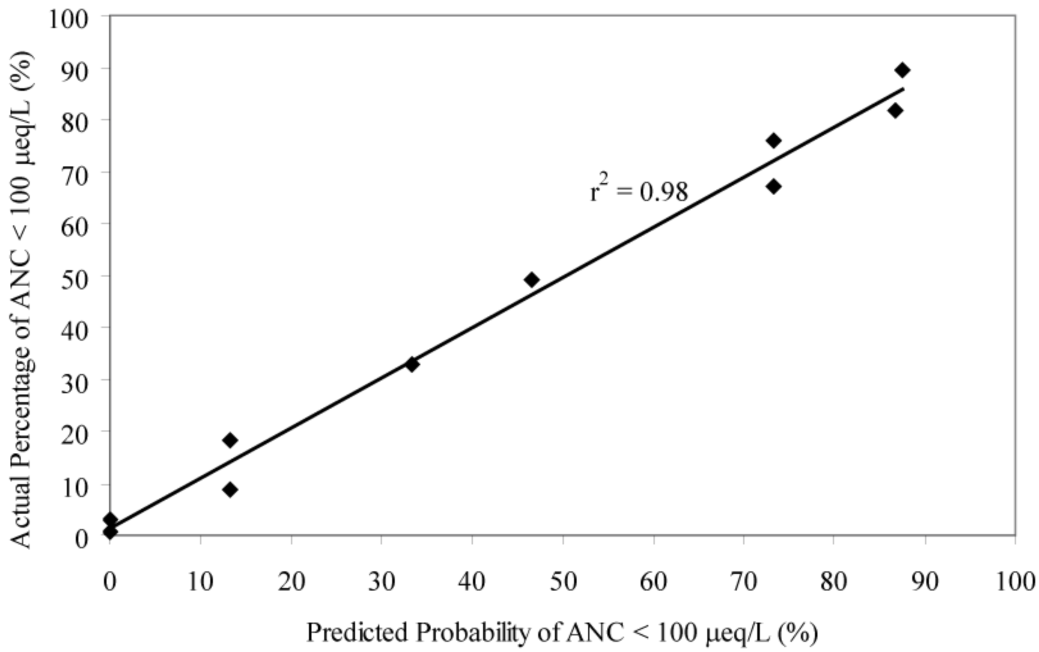

Figure 2-7. Percentage of actual ANC concentrations less than 100 μeq/L and the predicted probability of ANC concentrations less than 100 μeq/L (calibration data, n=151).

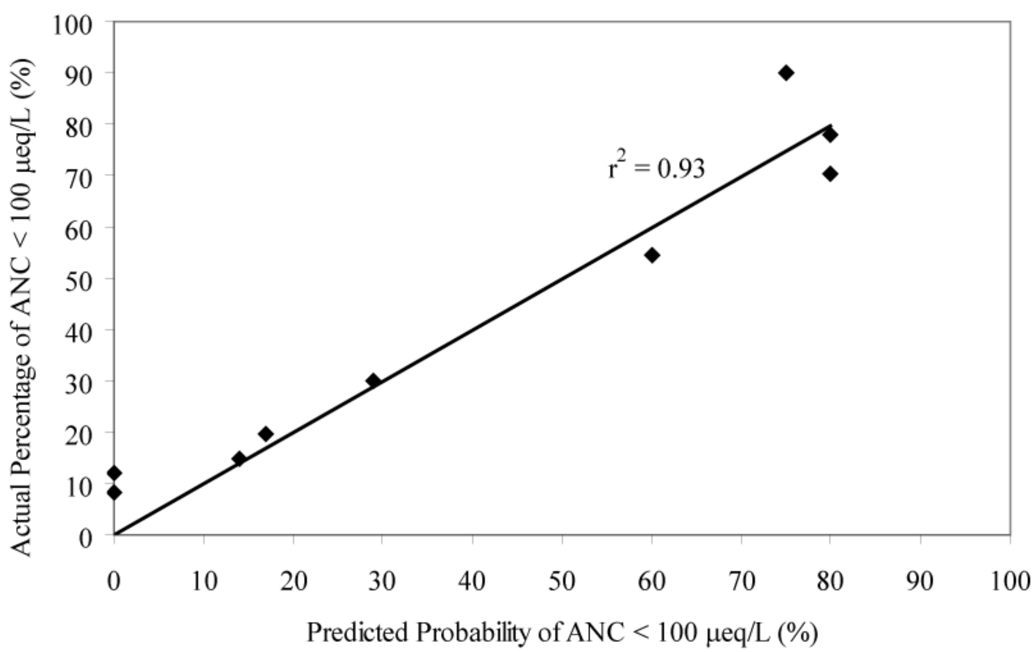

Figure 2-8. Percentage of actual ANC concentrations less than 100 μeq/L and the predicted probability of ANC concentrations less than 100 μeq/L (validation data, n=58).

Lake Sensitivity Classification for Four Individual Park Models

Multivariate logistic regression models also were developed for each park because some of the explanatory variables were not available for all the parks and could not be included in the regional model. For example, digital soils data were not available for GRSA. Moreover, for those parks with more calibration lakes, the individual models may have greater statistical strength than the regional model. GRSA did not have adequate calibration data (n=4) to conduct an individual park level assessment. For the individual parks, the same three ANC concentration thresholds were tested: 50 μeq/L, 100 μeq/L, and 200 μeq/L.

For ROMO, 40 lakes were used for model calibration (Table 2-2). The model with the best statistical outcome with respect to predicted probability was the model that used a 50 μeq/L threshold. This is likely due to the fact that none of the lakes included in the calibration data set had ANC concentrations greater than 200 μeq/L and only four lakes had ANC concentrations greater than 100 μeq/L. The total percentage of lakes in the calibration data set that had lakes with ANC concentrations less than 50 μeq/L is 43%. ANC concentration less than 50 μeq/L was significantly related to elevation at lake outlet (*p-value* = 0.07), bedrock geology with low buffering capacity (*p-value* = 0.06) and soils with very slow infiltration and high runoff potential, such as clays (*p-value* = 0.03) (Table 2-3).

The resulting probability equation was applied to lakes in ROMO.

$$Logit(P) = \frac{e^{\left(-46.3 + (0.007 \times elevation) + (12.3 \times \%\, basin\ soil\ type) + (8.09 \times \%\, basin\ with\ low\ buffering\ capacity\ bedrock)\right)}}{1 + e^{\left(-46.3 + (0.007 \times elevation) + (12.3 \times \%\, basin\ soil\ type) + (8.09 \times \%\, basin\ with\ low\ buffering\ capacity\ bedrock)\right)}}$$

(eqn. 4)

where *Logit(P)* and *e* are defined in equation 2. The HL goodness-of-fit r-squared = 0.80, and the *c*-statistic = 0.92, indicating a good model fit to the calibration data (Table 2-3). The probabilities associated with each basin provide an indication of the lake sensitivity to atmospheric deposition. Results indicate that there were 31 lakes in ROMO (27% of lakes > 1 ha) with a high probability for lake ANC concentrations less than 50 μeq/L during baseflow.

For GRTE, 52 lakes were used for model calibration. ANC concentrations less than 100 μeq/L are significantly related (*p*-value < 0.1) to elevation and aspect. ANC concentrations less than 200 μeq/L are significantly related (*p*-value < 0.1) to basins with a high percentage of low buffering capacity bedrock. For GRTE, the model with the best statistical outcome was the model that used a 50 μeq/L threshold, and approximately 17% of lakes used in the calibration dataset had ANC concentrations less than 50 μeq/L. Results of the 50 μeq/L model indicate that the percentage of the basin with steep slopes (slopes greater than 30 degrees) (*p*-value = 0.03) and the percentage of the basin composed of the Leighcan Moran-Walcott association soils (*p*-value = 0.004) were the only basin characteristics that were statistically significant (*p*-value < 0.1) (Table 2-3).

The resultant probability equation was applied to the lakes > 1ha.

$$Logit(P) = \frac{e^{\left(-8.62 + (13.8 \times steep\ slopes) + (9.70 \times \%\ basin\ leighcan-moran-walcott\ soil)\right)}}{1 + e^{\left(-8.62 + (13.8 \times steep\ slopes) + (9.70 \times \%\ basin\ leighcan-moran-walcott\ soil)\right)}} \quad \text{(eqn. 5)}$$

where *Logit (P)* and *e* are defined in equation 1. In GRTE, results indicate that five lakes (5% of GRTE lakes greater than 1ha) had a high probability for lake ANC concentrations less than 50 μeq/L during baseflow. In GRTE, measured ANC concentrations less than 50 μeq/L compared to predicted in random groupings of 10 %, showed good agreement with an HL r-squared value equal to 0.70. The *c*-statistic is equal to 0.90 indicating good association (Table 2-3).

In GLAC, using an ANC threshold of 50 μeq/L and 100 μeq/L, no variables were statistically significant (p-values > 0.1). For GLAC, the model with the best statistical outcome used an ANC concentration threshold of 200 μeq/L. Unlike ROMO and GRTE, which had many lakes with ANC concentrations less than 50 μeq/L, GLAC only had one lake with ANC less than 50 μeq/L and two between 50 μeq/L to 100 μeq/L in the calibration data set. However, using an ANC concentration of 200 μeq/L, 18% of the 33 lakes that were used to calibrate the model had ANC concentrations less than the threshold of 200 μeq/L. Results of the statistical model indicate that elevation (p-value = 0.10) and the percentage of the basin with high buffering capacity bedrock (*p-value* = 0.09) were significant (Table 2-3).

The resulting probability equations were applied to the delineated lake basins.

$$Logit(P) = \frac{e^{\left(-13.6 + \left(0.008 \times elevation\right) - \left(12.0 \times \%\ basin\ with\ high\ buffering\ capacity\ bedrock\right)\right)}}{1 + e^{\left(-13.6 + \left(0.008 \times elevation\right) - \left(-12.0 \times \%\ basin\ with\ high\ buffering\ capacity\ bedrock\right)\right)}} \qquad \text{(eqn. 6)}$$

where *Logit (P)* and *e* are defined in equation 1. Results indicate that 52 lakes in GLAC had a high probability for lake ANC concentrations less than 200 μeq/L during baseflow. This represents 21% of lakes in GLAC that are greater than 1 ha. The HL goodness-of-fit (r-squared = 0.86), and the *c*-statistic (*c*-statistic = 0.95), indicate good association (Table 2-3).

For YELL, 23 lakes were used to calibrate the statistical model. For ANC less than 50 μeq/L, none of the variables tested were statistically significant (p-value >

0.1), and there were no lakes with ANC concentrations less than 50 μeq/L. Lakes with ANC concentrations less than 100 μeq/L were significantly related (p-value < 0.1) to elevation. The model with the best statistical outcome with respect to predicted probability was the model that used a 200 μeq/L threshold. Of the 23 lakes, 35% had ANC concentrations less than 200 μeq/L. Results indicate that increasing elevation (p-value = 0.09) and percentage of the basin with medium buffering capacity bedrock (p-value = 0.05) were significantly related to ANC concentration less than 200 μeq/L.

Resulting probability equations were applied to lakes in YELL.

$$Logit(P) = \frac{e^{\left(-14.9 + \left(0.002 \times elevation\right) + \left(3.08 \times \%\ basin\ with\ medium\ buffering\ capacity\ bedrock\right)\right)}}{1 + e^{\left(-14.9 + \left(0.002 \times elevation\right) + \left(3.08 \times \%\ basin\ with\ medium\ buffering\ capacity\ bedrock\right)\right)}} \qquad \text{(eqn. 7)}$$

where $Logit\ (P)$ and e are defined in equation 1. In YELL, 18 lakes (6% of lakes greater than 1 ha) had a high probability for ANC concentrations less than 200 μeq/L during baseflow. The HL goodness-of-fit (r-squared = 0.61) and c-statistic = 0.78 (Table 2-3), indicated a reasonable model fit.

DISCUSSION

The combined use of GIS modeling and multivariate logistic regression using available GIS and water-quality data made regional-scale predictions that passed rigorous field validation. This approach was validated with independent water-quality data collected at 58 lakes during baseflow in 2004. The high r-squared (r-squared=0.93) in the regional model using the 2004 validation data, indicates that the model can be used to successfully identify a subset of lakes in national parks of the

Rocky Mountains that are most likely to be sensitive to acidic deposition. The approach presented in this paper may be transferable to other remote high-elevation protected areas that are sensitive to atmospheric deposition of contaminants, such as wilderness areas in national forests in the Western US.

Relations between ANC and Basin Characteristics

Lakes across the region were identified as sensitive based on a high predicted probability of having an ANC concentration of less than a specified threshold (50 μeq/L, 100 μeq/L, and 200 μeq/L) (Figure 2-5). These results support the initial hypothesis that in the Rocky Mountains, ANC is low (less than 100 μeq/L) in lakes that are in moderate to high-elevation headwater basins, with little soil cover, and low-buffering capacity bedrock. This hypothesis is in agreement with previous work conducted in the Rocky Mountains (Clow and Sueker, 2000; Rutkowski et al., 2001). Of the basin characteristics that were considered in either the regional model or the park models, elevation, bedrock geology, and soils were found to have the greatest effect on predicted ANC concentration probabilities (Table 2-3).

Elevation was found to be a predictor of lake ANC, such that ANC was inversely correlated with elevation, consistent with Rutkowski et al. (2001). However, Berg et al. (2005) did not find elevation to be a significant predictor of ANC concentration in the Sierra Nevada. In the Rocky Mountains, physical characteristics of high-elevation basins, combined with the storage and release of contaminants in snowmelt runoff from deep snowpacks, make them susceptible to atmospheric contamination (Williams et al., 1996, Baron and Campbell, 1997, Clow and Sueker, 2000). Results of this study indicate that for the Rocky Mountain regional model, increased elevation at the lake outlet was significantly related (p-value < 0.10) to ANC concentrations less than 100 μeq/L, for ROMO to ANC less than 50 μeq/L, and for YELL and GLAC to ANC less than 200 μeq/L.

Results of bedrock geology for the regional and park models are consistent with findings of Clow and Sueker (2000) in ROMO and Corbin et al. (2006) in GRTE. For the Rocky Mountain regional model and ROMO, significant relations were found between low ANC concentrations and bedrock types with low buffering capacity, including quartzite and granite. For GLAC, significant relations were found between high ANC concentrations and high-buffering capacity bedrock types that included carbonates and calc-silicates, due to the potential for the production of base cations that balance acidic anions.

The type of soil and the spatial extent and depth of soils has the potential to affect ANC concentrations. In the Rocky Mountains, soils that were classified as having a high runoff potential were significantly related (p-value < 0.10) to low ANC concentrations (< 50 μeq/L). For example in ROMO, soils with a very slow infiltration rate (high runoff potential) consist primarily of clays and have a very slow rate of water transmission. These soils have a layer that impedes the downward movement of water. Thus, there is limited water-soil contact time and limited interaction with geologic weathering products that could buffer the acidity. Basins with a high percentage of this soil type are significantly related (p-value < 0.10) to low ANC concentrations, such that as the percentage of the basin with this soil type increases, probability for ANC less than 50 μeq/L increases. Given that soils were significantly related to ANC concentrations in ROMO and GRTE, omitting soils in the regional model may have affected the results.

Atmospheric deposition of hydrogen ion, inorganic N, and sulfate has the potential to alter the chemistry of aquatic ecosystems through N saturation and episodic acidification, thus increasing the sensitivity of lakes to future changes related to atmospheric deposition. Deposition estimates were calculated for each basin and were included in the analysis for the region and for each park. Results, however, indicated that deposition was not statistically significantly related to ANC concentration and, therefore, it was not included in the final equations. This result may be because deposition is fairly similar among sites within each park (Nanus et al., 2003), due to the resolution of PRISM precipitation maps used to develop deposition estimates. It may also be due to the fact that acidic deposition in the Western U.S. is not sufficient to cause chronic acidification.

Management Applications

Deposition estimates and lake sensitivity maps can be used together by resource managers to identify lakes that have both a high probability for sensitivity to atmospheric deposition and a relatively high deposition of acidic solutes. Average annual atmospheric deposition estimates were overlaid with the results of the sensitivity analysis to locate these lakes. In GRTE and ROMO, 53% of lakes are sensitive to the deposition of acidic solutes and areas within these parks have high rates of inorganic N, sulfate, and acid deposition (Nanus et al., 2003). In the Colorado Rockies, researchers have shown that elevated levels of N deposition have caused changes in aquatic ecosystems at high-elevations (Baron, 2006; Williams et. al., 1996). Therefore, there is considerable interest in identifying areas in GRTE and ROMO that have both sensitive lakes and high inorganic N deposition as shown in Figure 2-9. In GRTE and ROMO, lakes with a high probability (66-100%) for ANC less than 100 μeq/L are primarily located in areas that receive 2-5 kg/ha/yr inorganic N. In GRTE, lakes with a low probability (0-33%) are located in areas that receive less than 2 kg/ha/yr inorganic N. This is an example of an application for resource managers using these maps to find lakes most at risk to change due to acidic atmospheric deposition.

The GIS and logistic regression modeling approach can be used as a cost-effective tool to help resource managers. In the future, probability estimates can be used to select lakes to be included in a monitoring program. For GRTE and ROMO, where lakes appear to be most sensitive to acidic deposition, long-term monitoring programs are needed in order to capture seasonal variability and potential episodic acidification. This analysis predicts the probability of ANC concentrations below a set threshold, but does not include other chemical constituents. Nitrate and sulfate concentrations may be evaluated using a similar approach to identify significant relations between basin characteristics and lake-chemical concentrations in select national parks of the Western U.S. Future sampling sites also could include climate stations for precipitation sample collection that would allow resource managers to observe short-term and long-term variability in inorganic N and sulfate wet deposition.

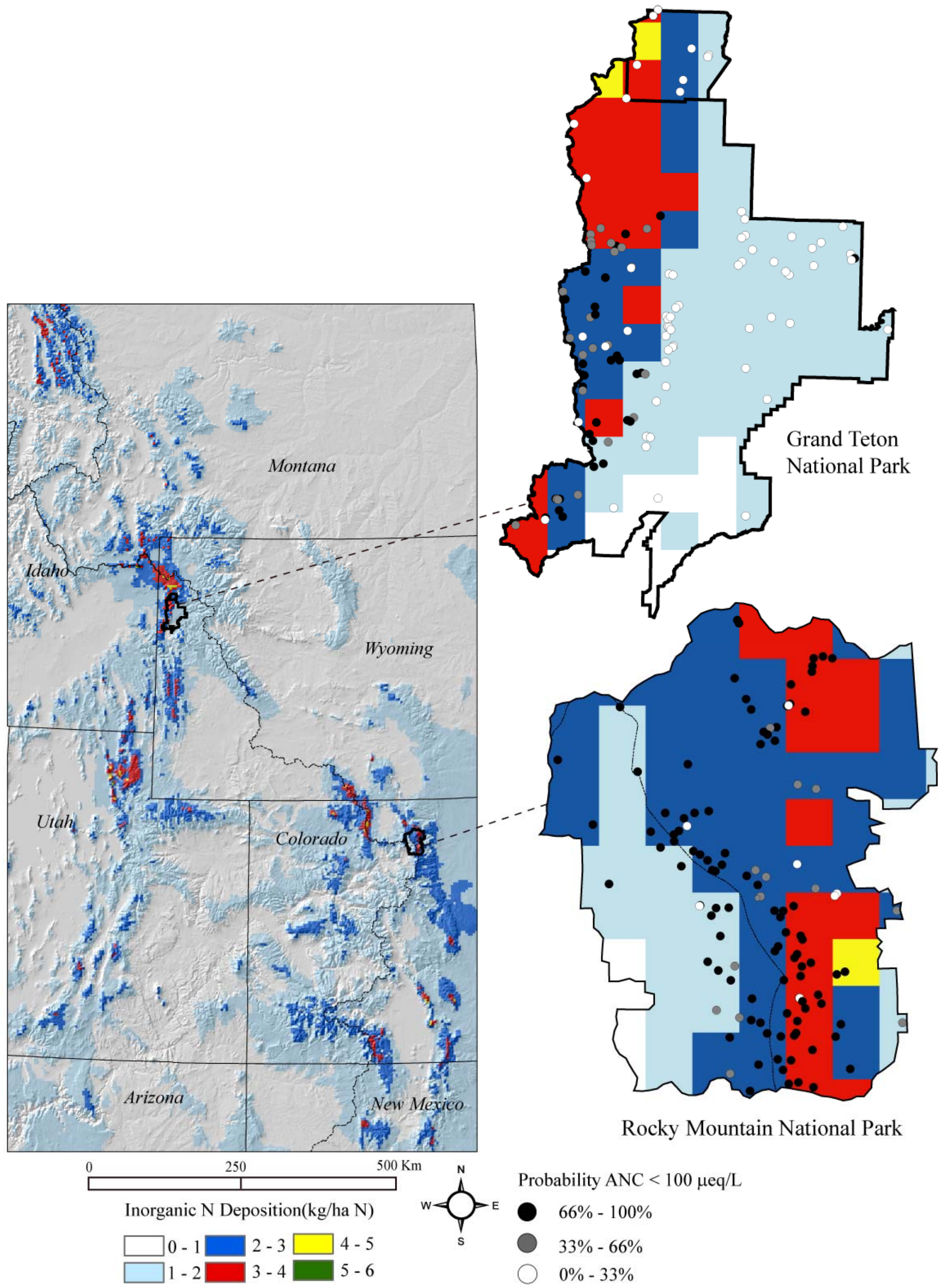

Figure 2-9. Average annual inorganic N deposition in the Rocky Mountains (Nanus et al., 2003) overlaid with the probability for lake sensitivity in Grand Teton and Rocky Mountain National Parks.

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CHAPTER 3

EVALUATING REGIONAL PATTERNS IN NITRATE SOURCES TO WATERSHEDS USING NITRATE ISOTOPES[1]

ABSTRACT

In the Rocky Mountains, there is uncertainty in the source areas and emission types that contribute to nitrate (NO_3) deposition, which can adversely affect sensitive aquatic habitats of high-elevation watersheds. Regional patterns in NO_3 deposition sources were evaluated using NO_3 isotopes in five National Parks, including 37 lakes and 7 precipitation sites. Results indicate that lake NO_3 ranged from detection limit to 38 μeq/L, $\delta^{18}O$ (NO_3) ranged from -5.7 to +21.3 permil, and $\delta^{15}N$ (NO_3) ranged from -6.6 to +4.6 permil. $\delta^{18}O$ (NO_3) in precipitation ranged from +71 to +78 permil. $\delta^{15}N$ (NO_3) in precipitation and lakes overlap; however, $\delta^{15}N$ (NO_3) in precipitation is more depleted than $\delta^{15}N$ (NO_3) in lakes, ranging from -5.5 to -2.0 permil. $\delta^{15}N$ (NO_3) values are significantly related ($p < 0.05$) to wet deposition of inorganic N, sulfate, and acidity, suggesting that spatial variability of $\delta^{15}N$ (NO_3) over the Rocky Mountains may be related to source areas of these solutes. Regional patterns show that NO_3 and $\delta^{15}N$ (NO_3) are more enriched in lakes and precipitation from the southern Rockies and at higher elevations compared to the northern Rockies. The correspondence of high NO_3 and enriched $\delta^{15}N$ (NO_3) in precipitation with high NO_3 and enriched $\delta^{15}N$ (NO_3) in lakes, suggests that deposition of inorganic N in wetfall may affect the amount of NO_3 in lakes through a combination of direct and indirect processes such as enhanced nitrification.

INTRODUCTION

In the western United States, anthropogenic emissions of NO_x (nitrogen oxides) and NH_3 (ammonia) from energy generation activities, transportation, industry, and agricultural activities contribute to deposition of dissolved inorganic nitrogen (DIN $=NO_3$ + NH_4) in high-elevation watersheds (Baron et al., 2000; Williams and Tonnessen, 2000). There is considerable uncertainty in the source areas and emission types that contribute to deposition of DIN, which can adversely affect sensitive aquatic habitats of high-elevation lake basins (Burns, 2004). Deposition data indicate that DIN in wetfall has increased steadily over much of the Rocky Mountains in recent years for a variety of reasons (Burns, 2004; Fenn et al., 2003; Nilles and Conley, 2001), including increases in motor vehicle emissions which have offset reductions in NO_x emissions from fossil fuel burning industries (USEPA, 2000) and the regional increases in ammonia emissions (NADP, 2007). The percent of DIN in wet deposition contributed by NH_4 has increased from 1992-1996 to 2002-2006, and is now approximately 50% of measured DIN in over half the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) sites in the Rocky Mountains (NADP, 2007), including sites located near national parks. A previous study that evaluated all major emission sources (including both stationary and mobile sources) across the Rocky Mountains found that the Colorado Front Range, which is located near large urban centers, has the highest N emissions (Williams and Tonnessen, 2000). Spatial trends in deposition of DIN in wetfall over the Rocky Mountains, show that deposition is greatest near Rocky Mountain National Park and Great Sand Dunes National Park and Preserve near the Colorado Front

Range in the southern Rocky Mountains, compared to National Parks in the northern Rocky Mountains that are also located further west (NADP, 2008; Nanus et al., 2003) (Figure 3-1, Figure 3-2).

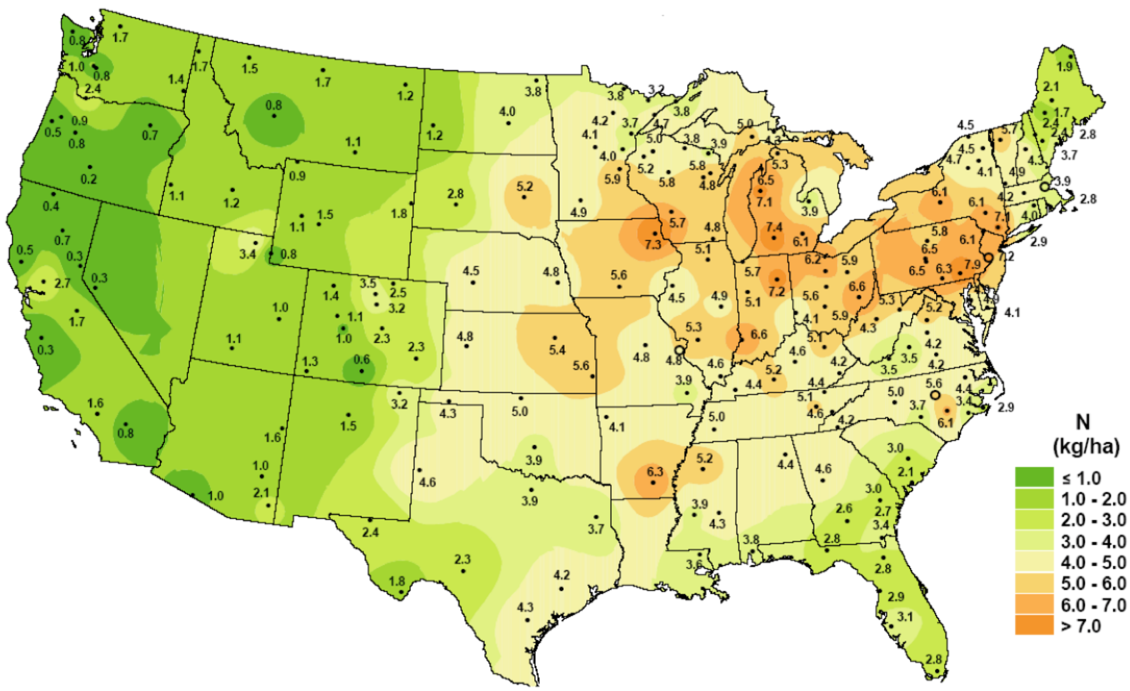

Figure 3-1. Inorganic nitrogen wet deposition from nitrate and ammonium, 2004 (from NADP, 2008).

Figure 3-2. Location of National Parks included in study area and NADP/NTN (NADP) sites.

Federal and state resource managers are investigating policy options to alleviate this problem by reducing anthropogenic emissions of NO_x and NH_3. However, identifying source areas and emission types is complicated (Elliott et al., 2007; Kendall, 1998). Isotopic tracers of N measured in precipitation and water samples show promise in helping to identify these emission sources (Elliott et al., 2007; Kendall, 1998). Previously published studies (Heaton, 1990; Kiga et al., 2000) indicate $\delta^{15}N$ (NO_3) values in NO_x emissions from coal-fired power plants have isotopic values ranging from +6 to +13 permil (Heaton, 1990; Kiga et al., 2000). $\delta^{15}N$ (NO_3) values from motor vehicle NO_x emissions in tailpipe exhaust range from -13 to -2 permil (Heaton, 1990). The following $\delta^{15}N$ (NO_3) values have also been reported for vehicle NO_x emissions in tailpipe exhaust (+3.7 permil) and roadside vegetation (+3.8 permil) (Ammann et al., 1999; Pearson et al., 2000). The use of these NO_3 isotopes has been limited in part because analytical techniques for NO_3 isotopes required large sample volumes that made it logistically difficult to sample in areas with topographically complex terrain.

Researchers have shown that elevated levels of atmospheric N deposition in the Front Range of Colorado have caused substantial changes in the state and function of terrestrial and aquatic ecosystems at high-elevations (Baron et al., 2000; Burns, 2003; Campbell et al., 2000; Campbell et al., 2002; Mast et al., 2003; Williams et al., 1996). In one study, inorganic N retention of DIN in wetfall averaged 72% in high-elevation ecosystems (Sickman et al., 2002). N deposition in excess of the total combined plant and microbial demand can cause watershed N saturation and increased rates of N leaching from soils to aquatic ecosystems (Aber et al., 1989), which is occurring in the Colorado Rockies (Williams et al., 1996; Baron, 2006). This excess N can result in a cascade of ecological effects in surface waters, that includes acidification, eutrophication, and increased emissions of N_2O, a greenhouse gas. Eutrophication increases primary productivity in lakes and streams and alters diatom species distributions that form the base of the food web in many high-elevation lakes (Wolfe et al., 2001). The combined effects of increasing N deposition and drought have sharply increased stream water concentrations of NO_3 in Rocky Mountain National Park (Rocky) in recent years (Williams et al., 1996). Changes in the water quality of the headwater systems affects not only fish, wildlife and ecosystem integrity, but also downstream ecosystems and water users.

Increased aquatic productivity resulting from eutrophication accelerates the accumulation of organic matter in the water column and in lake sediments. Decomposition of this organic matter promotes hypoxia in lakes when they are ice-covered during winter, and may adversely affect fish populations (Vitousek et al., 1997). In Rocky, mineralization of organic N in pond sediments has caused concentrations of dissolved ammonia in vernal ponds to reach levels that may be harmful to threatened amphibians that breed there (Campbell et al., 2004). Although N deposition is greatest in the Front Range of Colorado, other high-elevation sites in the Rocky Mountains also show symptoms of early-stage N saturation. The progression towards N saturation is expected to continue if N deposition continues at current or higher levels in the future (Campbell, 2003).

The denitrifier method to determine the dual isotopic composition ($\delta^{15}N$ and $\delta^{18}O$) of NO_3 is well suited for studies of NO_3 contributions to streams and lakes (Ohte et al., 2004). This method requires only 20-60 nmol of NO_3 and enables high throughput of samples (Sigman et al., 2001; Casciotti et al., 2002). The development of a new analytical technique for analyzing $\delta^{15}N$ (NH_4) holds promise in terms of tracing other sources of N deposition. However, it requires large sample volumes and is beyond the scope of this study.

To evaluate NO_3 sources, we analyzed both $\delta^{15}N$ and $\delta^{18}O$ of NO_3 using the denitrifier method in samples collected from lakes in the Rocky Mountains that span a range of NO_3 deposition (Nanus et al., 2003). The objectives of this study were to: 1) evaluate the spatial distribution of $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) in lake samples from five National Parks collected during baseflow conditions; and 2) compare the isotopic composition of the lake water from these watersheds with that of precipitation collected at nearby NADP/NTN sites. This study is the first comprehensive evaluation using NO_3 isotopes to investigate the possible relationship between atmospheric deposition of NO_3 in wetfall and the NO_3 in lakes of the Rocky Mountains.

EXPERIMENTAL SECTION

Study Area and Field Methods

The five National Parks in the Rocky Mountains included in this study are Glacier National Park (Glacier), Yellowstone National Park (Yellowstone), Grand Teton National Park (Grand Teton), Rocky Mountain National Park (Rocky), and Great Sand Dunes National Park and Preserve (Great Sand Dunes) (Figure 3-2). The areas under investigation are in the headwaters of most of the major rivers in the western United States, and their airsheds extend across state and national political boundaries. Precipitation chemistry was measured at 7 NADP/NTN sites located near National Parks (Figure 3-2). The precipitation samples collected weekly at NADP/NTN sites from 2000 were pooled into bimonthly, volume-weighted-mean composites, and analyzed for NO_3 concentrations and for $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) (Elliott et al., 2007). For this study, bimonthly $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) values were aggregated into average annual wet deposition values. Sample duplicates had an average standard deviation of 0.6‰ for $\delta^{18}O$ (NO_3) and 0.4‰ for $\delta^{15}N$ (NO_3). Dry deposition was not included in this evaluation.

Lakes were randomly selected for sampling during late summer 2004 and are spatially distributed within each of the National Parks. The 56 sampled lakes range in elevation from 2,000 to 3,800 m and from 1 to 46 hectares (ha) in area, with 65% of the lakes less than 5 ha in area. Surface waters were collected from the outflow of each lake as grab samples during the low-flow period from August to September. Samples were collected at baseflow when NO_3 concentrations in surface waters are generally near or at their annual minima as a result of biological assimilation (Williams and Tonnessen, 2000). Surface waters that have elevated NO_3 concentrations (ie. greater than about 5 μeq/L) during baseflow conditions may be approaching the initial stage of N saturation (Williams and Tonnessen, 2000) and may be particular susceptible to inputs of DIN in wetfall. Polyethylene bottles (250-ml) were soaked with deionized (DI) water overnight and then rinsed with DI water 5 times; bottles were further rinsed 3 times with sample water at the time of collection. Samples were frozen after collection and transported to the Kiowa Environmental Chemistry Laboratory (Seibold, 2001) run by the Niwot Ridge Long-Term Ecological Research Program (University of Colorado, INSTAAR), which specializes in analysis of dilute waters such as those found in the study area (Williams et al., 2001).

Laboratory Analyses

All lake samples were analyzed for pH, acid neutralizing capacity (ANC), conductance, and major ions. ANC and pH were measured immediately after melting or after return to the laboratory using the Gran titration technique. Subsamples were immediately filtered through pre-rinsed (300 ml), 47-mm Gelman A/E glass fiber filters with ca. 1-μM pore size. Filtered samples were stored in the dark at $4^{o}C$ for subsequent analyses within 1 to 4 weeks. Anions were measured using ion chromatography (Dionex DX 500) employing chemical ion suppression and conductivity detection. Base cations were analyzed with a Varian AA6 atomic absorption spectrophotometer using an air-acetylene flame. Quality assurance for this study was addressed with field duplicate samples separated by 10-15 samples in each run. Analytical precision for all solutes was less than 2% and detection limits were less than 1 μeq/L.

Frozen aliquots were analyzed for $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) using the denitrifier method at the USGS Stable Isotope Laboratory in Menlo Park. In this method, denitrifying bacteria (*Pseudomonas aureofaciens*) quantitatively convert the N and O from NO_3 into gaseous nitrous oxide (N_2O) for isotopic analysis (Sigman et al., 2001; Casciotti et al., 2002). A minimum of 20 nmol NO_3 was required to analyze samples on a Micromass IsoPrime Isotope Ratio Mass Spectrometer (IRMS). Sample duplicates had an average standard deviation of 0.7‰ for $\delta^{18}O$ (NO_3) and 0.2‰ for $\delta^{15}N$ (NO_3).

To evaluate regional differences in spatial patterns, $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) values from lakes were compared to $\delta^{15}N$ (NO_3) values in precipitation from co-located NADP/NTN sites. Also, isotopic values of NO_3 in lakes were compared with emissions within a specified buffer distance calculated using emissions inventories (Williams and Tonnessen, 2000; USEPA, 2000; Elliott et al., 2007). For this study, stationary source NO_x emission inventory data that is readily available (USEPA, 2000) was used as a surrogate for various anthropogenic N emission sources that are not as readily available including motor vehicles, agriculture, feedlots, power plants and other industrial emission sources (Williams and Tonnessen, 2000).

Basin characteristics were evaluated to determine their potential influence on spatial patterns in NO_3 concentrations, $\delta^{18}O$ (NO_3), and $\delta^{15}N$ (NO_3) values. Forty-eight basin characteristics were derived using Geographic Information Systems (GIS) software, and included bedrock type, slope, aspect, elevation, lake area, soil type, and vegetation type, following the protocols presented in (Nanus et al., 2005).

RESULTS AND DISCUSSION

NO_3 concentrations in the 56 lakes sampled ranged from below the detection limit (~1 μeq/L) to 38 μeq/L (Figure 3-3). Mean values were highest in Rocky (20 μeq/L) and lowest in Yellowstone (0.2 μeq/L). An analysis of variance test (ANOVA) shows that mean concentrations of NO_3 varied significantly among National Parks ($n=56$, $p < 0.001$). A follow-up Tukey-Kramer HSD test shows that the mean value of 20 μeq/L for NO_3 at Rocky was significantly higher than the other 4 parks.

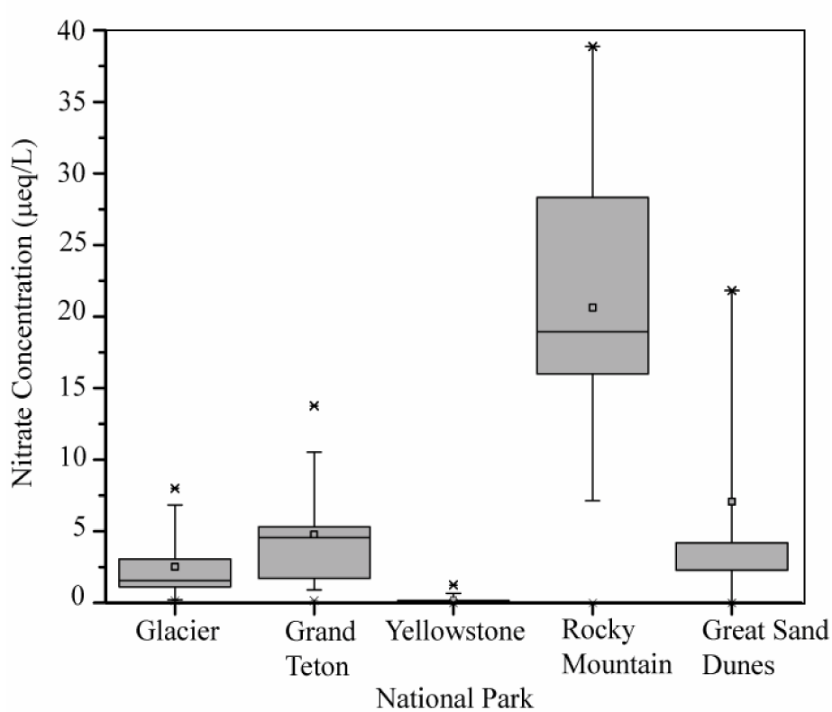

Figure 3-3. Nitrate Concentrations at 56 lakes sampled during late summer 2004, aggregated by National Park.

Of the 56 lakes sampled, 37 lakes had sufficient mass of NO_3 to analyze for $\delta^{15}N$ (NO_3) and $\delta^{18}O$ (NO_3) (Table 3-1). Lake samples with NO_3 mass greater than 20 nmoles were distributed among the parks as follows: Glacier (n=11), Grand Teton (n=11), Great Sand Dunes (n=3), Rocky (n=11), and Yellowstone (n=1).

Table 3-1. NO_3 concentration, $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) values for National Park lakes.

State	National Park	Site Name	Elevation (meters)	Nitrate Concentration (μeq/L)	$\delta^{15}N(NO_3)$ (permil)	$\delta^{18}O(NO_3)$ (permil)
MT	Glacier	Aikayan	2344	1.9	-2.4	21.3
MT	Glacier	Avalanche Lake	1190	3.1	0.7	9.7
MT	Glacier	Feather Woman	2298	1.6	-1.7	17.0
MT	Glacier	Grinnel Lake	1505	6.8	0.4	7.9
MT	Glacier	Gunsight Lake	1620	4.1	-0.4	12.2
MT	Glacier	Hidden Lake	1943	1.5	-0.6	10.4
MT	Glacier	Lake No Name-Siyeh Pass	2216	8.0	-1.1	12.9
MT	Glacier	Medicine Grizzly Lake	1696	1.1	-1.3	11.1
MT	Glacier	Oldman Lake	2026	1.1	-0.8	15.0
MT	Glacier	Ptarmigan Lake	2019	1.5	0.1	14.4
MT	Glacier	Snyder Lake Glac	1594	1.8	-1.0	-5.7
WY	Grand Teton	Arrowhead Lake	2790	5.3	-3.2	2.6
WY	Grand Teton	Bear Paw Lake	2088	1.7	-2.5	6.6
WY	Grand Teton	Bradley Lake	2141	10.5	-0.6	12.2
WY	Grand Teton	Lake No Name 44	2376	5.5	-1.8	0.9
WY	Grand Teton	Lake No Name 44A	2376	4.9	-0.4	2.0
WY	Grand Teton	Lake No Name 49	2967	13.8	-1.5	10.4
WY	Grand Teton	Lake No Name 89	2380	1.5	-2.4	11.1
WY	Grand Teton	Lake of the Crags	2916	4.6	-2.8	10.1
WY	Grand Teton	Ramshead Lake	2894	4.5	-3.4	7.1
WY	Grand Teton	Taggart Lake	2104	5.0	-1.1	9.1
WY	Grand Teton	Trapper Lake	2107	3.6	-1.9	1.2
CO	Great Sand Dunes	Lower Sand Creek Lake	3496	4.2	1.3	6.2
CO	Great Sand Dunes	Medano Lake	3509	21.8	2.4	3.3
CO	Great Sand Dunes	Upper Sand Creek Lake	3580	2.3	-2.4	3.6
CO	Rocky Mountain	Chasm Lake	3592	38.9	2.4	9.8
CO	Rocky Mountain	Dream Lake	3035	26.4	2.0	12.5
CO	Rocky Mountain	Emerald Lake	3083	27.8	1.7	13.4
CO	Rocky Mountain	Lake Haiyaha	3111	30.0	1.8	12.5
CO	Rocky Mountain	Lake Helene	3226	17.6	1.7	8.8
CO	Rocky Mountain	Lawn Lake	3349	16.0	4.6	8.1
CO	Rocky Mountain	Loch Vale	3103	18.6	1.5	13.0
CO	Rocky Mountain	Sky Pond	3316	18.6	2.3	13.6
CO	Rocky Mountain	Spruce Lake	2942	7.1	0.6	10.3
CO	Rocky Mountain	Thunder Lake	3217	16.0	2.6	10.3
CO	Rocky Mountain	Ypsilon Lake	3234	19.0	3.2	7.6
WY	Yellowstone	Sylvan Lake	2564	1.2	-6.6	5.9

$\delta^{18}O$ (NO_3) values in lake samples ranged from –5.7 to +21.3 permil, with a median value of +10.1 (Figure 3-4a). Both of the lakes with elevated $\delta^{18}O$ (NO_3) values are located in Glacier and receive direct input from glacier outflow, which may explain the enriched $\delta^{18}O$ (NO_3) values. $\delta^{15}N$ (NO_3) values for the 37 lakes ranged from –6.6 to +4.6 permil (Figure 3-4a), with a median value of –0.6 permil. Somewhat surprisingly, this range in $\delta^{15}N$ (NO_3) values is similar to $\delta^{15}N$ (NO_3) values in wet NO_3 deposition from NADP/NTN sites across the Northeastern and Mid-Atlantic US (Elliott et al., 2007). A plot of $\delta^{15}N$ (NO_3) values compared to NO_3 concentration, color coded by park, is shown in Figure 3-4b. In general, there was a trend for the lake $\delta^{15}N$ (NO_3) values to increase with increasing NO_3 concentrations (r^2 = 0.5; $p < 0.05$). Lake $\delta^{18}O$ (NO_3) values however, did not increase with increasing NO_3 concentrations ($r^2 = 0.02$; $p > 0.1$).

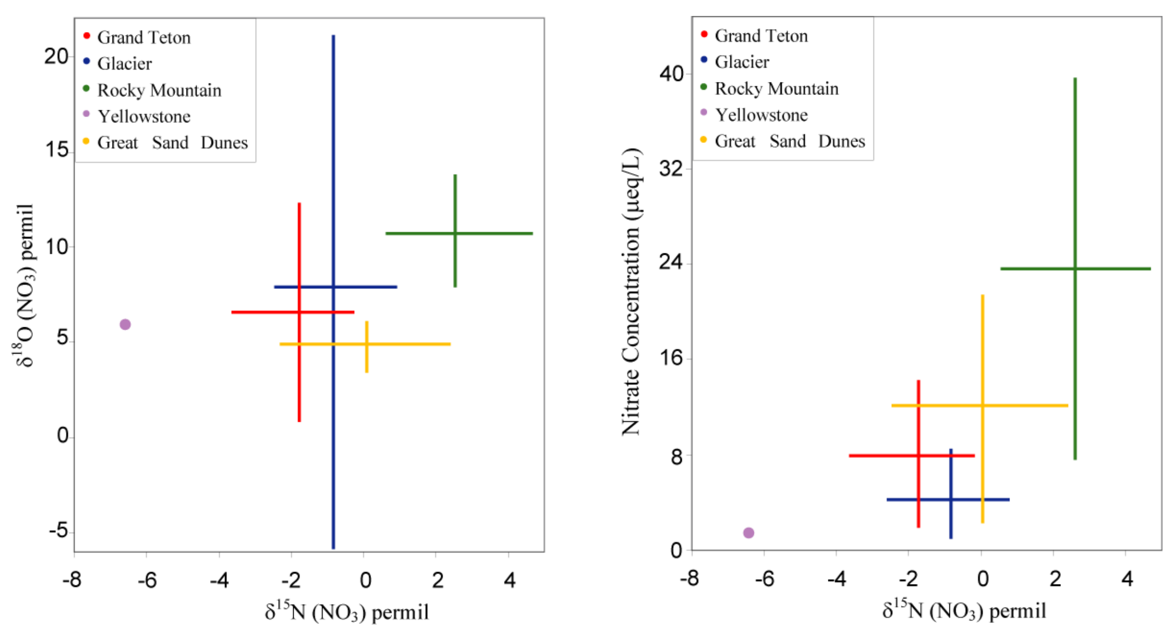

Figure 3-4a Figure 3-4b

Figure 3-4. Range in lake $\delta^{15}N$ (NO_3) values compared with range in lake $\delta^{18}O$ (NO_3) values (3-4a) and range in lake NO_3 concentrations (3-4b).

We compared the isotopic composition of NO_3 from these catchments to the isotopic composition of NO_3 in precipitation collected at the 7 nearby NADP/NTN sites in Figure 3-5 (Table 3-2). The $\delta^{18}O$ (NO_3) values in precipitation ranged from +71 to +78 permil, significantly more enriched than the –5.7 to +21.3 permil of samples in lake water ($p < 0.001$). The $\delta^{18}O$ (NO_3) values in lake water are not indicative of a direct atmospheric source (Figure 3-5). The $\delta^{15}N$ (NO_3) values in precipitation tended to be significantly more depleted than values in lake waters ($p < 0.001$), ranging from –5.5 to –2.0 permil (Figure 3-5). $\delta^{15}N$ (NO_3) values in precipitation generally increased from north to south. For example, $\delta^{15}N$ (NO_3) values were –5.5 permil for Glacier, and increased to –2.0 permil continuing south towards Wolf Creek Pass, Colorado. There was a significant positive trend for the atmospheric $\delta^{15}N$ (NO_3) values to increase with increasing NO_3 concentrations ($r^2 = 0.7$; $p < 0.01$).

Table 3-2. NO_3 concentration, $\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3) values for NADP/NTN sites.

State	Site ID	Site Name	Elevation (meters)	Nitrate Concentration (μeq/L)	$\delta^{15}N(NO_3)$ (permil)	$\delta^{18}O(NO_3)$ (permil)
CO	CO00	Alamosa	2298	15.8	-4.1	75.1
CO	CO18	Ripple Creek Pass	2929	12.5	-3.0	71.1
CO	CO19	Rocky Mountain -Beaver Meadows	2490	17.7	-2.1	74.8
CO	CO91	Wolf Creek Pass	3292	16.5	-2.2	74.0
CO	CO98	Rocky Mountain -Loch Vale	3159	14.0	-3.6	71.2
MT	MT05	Glacier -Fire Weather Station	980	8.6	-5.1	74.5
WY	WY08	Yellowstone -Tower Falls	1912	9.3	-5.5	78.3

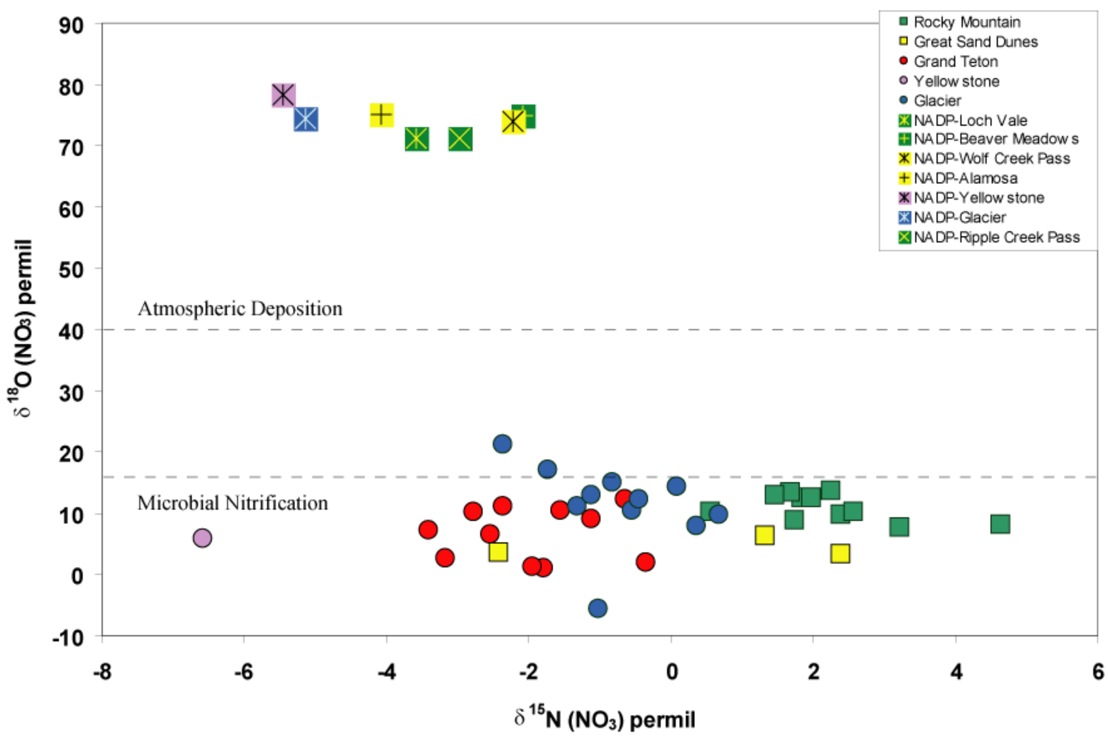

Figure 3-5. $\delta^{15}N$ (NO_3) values compared with $\delta^{18}O$ (NO_3) values at NADP/NTN sites and at lakes. Dashed lines represent estimated $\delta^{18}O$ (NO_3) end-members.

The occurrence of higher NO_3 concentrations and enriched $\delta^{15}N$ (NO_3) values in precipitation in National Parks characterized by higher NO_3 concentrations and enriched $\delta^{15}N$ (NO_3) values in lake waters, suggests that atmospheric deposition of DIN in wetfall affects the amount of NO_3 in lakes. $\delta^{15}N$ (NO_3) values in both precipitation and lakes are more enriched in Colorado parks than in the northern parks in Wyoming and Montana. The correspondence of enriched $\delta^{15}N$ with higher concentrations of NO_3 in precipitation and lakes suggests that areas with higher deposition are affected by a source of anthropogenic N emissions that is enriched in $\delta^{15}N$. These results are similar to an earlier study in which significant correlations between $\delta^{15}N$ (NO_3) values in precipitation and stationary source NO_x emissions within source areas of 500-600 km in the eastern US were attributed to regional transport of NO_x (Elliott et al., 2007).

To evaluate this idea, isotopic values of $\delta^{15}N$ (NO_3) from lake water were compared with total stationary source NO_x emissions in USEPA Region 8 during 1990-1999, the longest recent period of record (USEPA, 2000), as a proxy for total anthropogenic N emissions (Williams and Tonnessen, 2000). Total stationary source NO_x emissions within a variety of buffer distances ranging from 50 km to 600 km were tested, and 300 km was the most highly correlated with the data for all parks. Results for Colorado and Wyoming indicate that there is a significant positive correlation ($r^2 = 0.8$, $p < 0.05$) between $\delta^{15}N$ (NO_3) from lake waters and NO_x emissions within a 300 km buffer for Rocky, Grand Teton, Great Sand Dunes, and Yellowstone (Figure 3-6). For Glacier, there are no reported stationary sources of NO_x emissions within a 300 km buffer during 1990-1999 (USEPA, 2000). As NO_x emissions increase, $\delta^{15}N$ (NO_3) values in NO_3 from lakes increase, suggesting a relationship between spatial variations in $\delta^{15}N$ across the Rocky Mountains and N emissions. This correlation suggests a contribution of regional anthropogenic N emissions sources. This is particularly apparent for National Parks in Colorado (Rocky and Great Sand Dunes) which are located near larger anthropogenic N emission sources compared to parks in northern Wyoming (Grand Teton and Yellowstone).

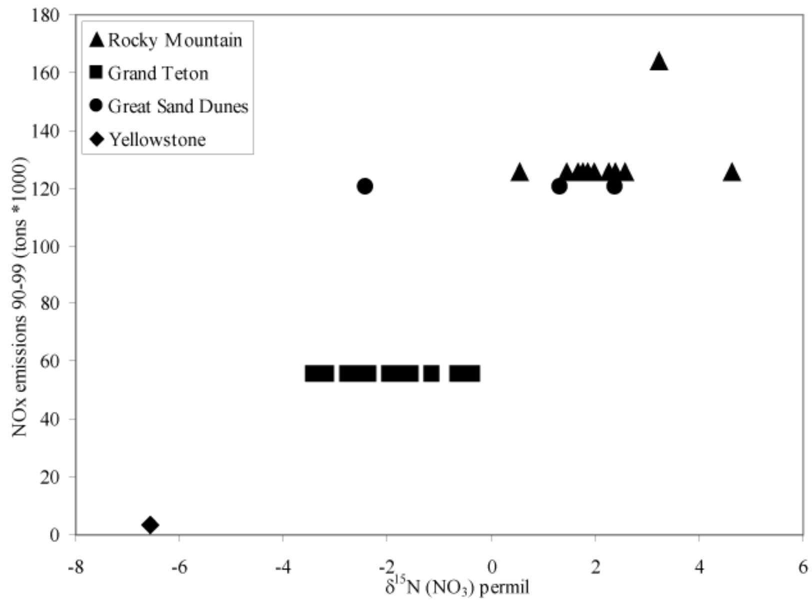

Figure 3-6. $\delta^{15}N$ (NO_3) values of lake waters compared with stationary source NO_x emissions within a 300-km buffer.

To further evaluate a potential connection between atmospheric deposition of pollutants and NO_3 in lake waters, lake $\delta^{15}N$ (NO_3) values were compared with average annual deposition estimates of DIN, SO_4, and H^+ in wetfall (Nanus et al., 2003) at Rocky and Grand Teton (Figure 3-7). Results indicate that lake water $\delta^{15}N$ (NO_3) values are significantly correlated ($p < 0.001$) with average annual deposition estimates of DIN ($r^2 = 0.61$), SO_4 ($r^2 = 0.63$), and H^+($r^2 = 0.60$). This correlation is consistent with the spatial variability of $\delta^{15}N$ (NO_3) in lake waters being related to the atmospheric deposition of pollutants in wetfall (Figure 3-7).

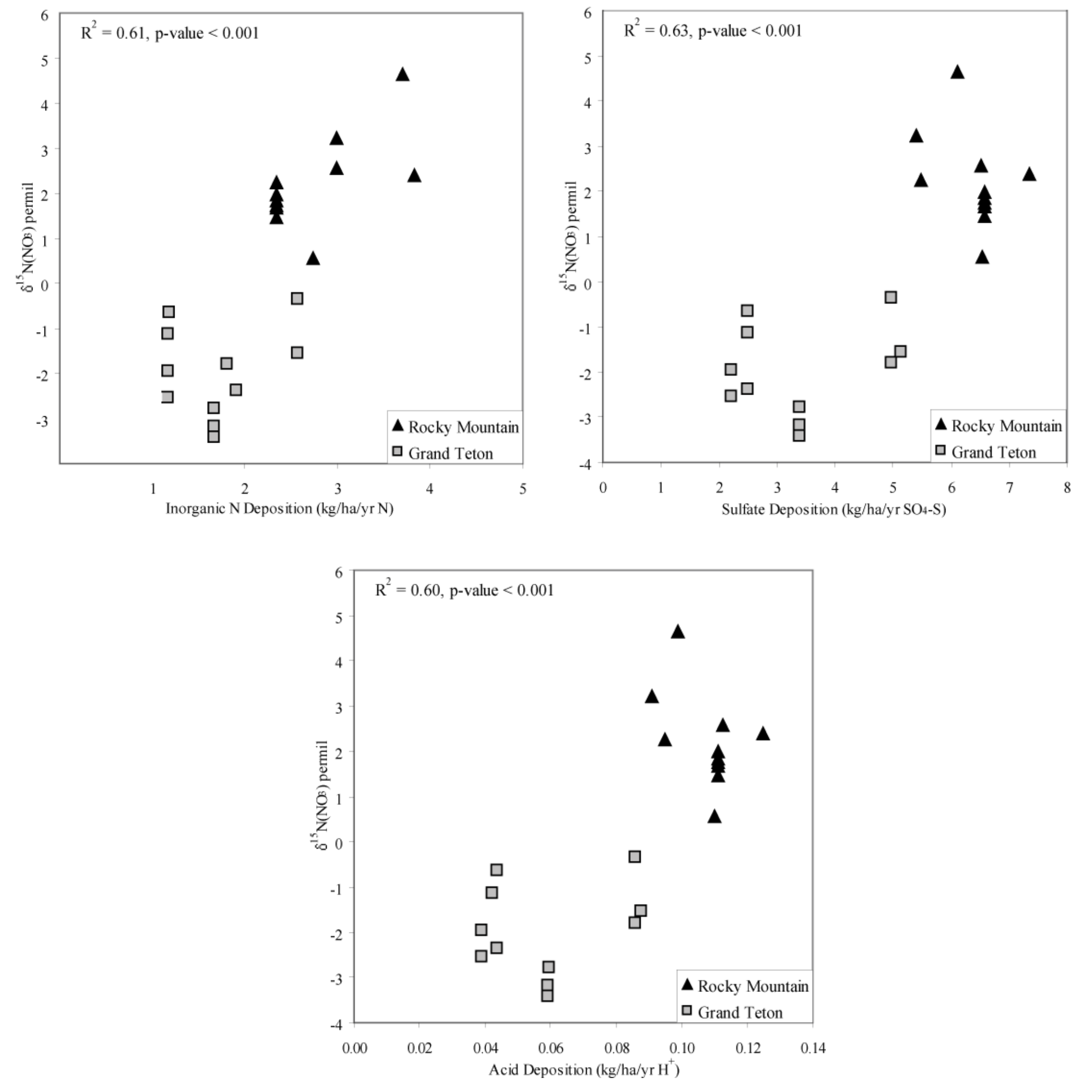

Figure 3-7. $\delta^{15}N$ (NO_3) values of lake water compared to average annual deposition of inorganic N (DIN), SO_4, and H^+ in wetfall for two parks (estimates of wetfall (Nanus et al., 2003)).

Modeling results using 48 GIS attributes for each watershed show that there is a significant positive relation ($p < 0.05$) between elevation for lakes located at elevations greater than 2500 meters and $\delta^{15}N$ (NO_3) values in lake water (Figure 8a). There was no significant relation between $\delta^{18}O$ (NO_3) values in lake waters and any GIS attribute. Geographic patterns of NO_3 concentrations of high-elevation lakes in the Rockies have previously been reported and results show particularly high NO_3

concentrations in Rocky (Clow et al., 2002). Here we show an increase in $\delta^{15}N$ (NO_3) values of lake waters with increasing elevation of the lakes (Figure 3-8a). There was also a trend for $\delta^{15}N$ (NO_3) values in precipitation to increase with increasing elevation ($p < 0.05$) (Figure 3-8b). It is possible that the enriched $\delta^{15}N$ (NO_3) values in precipitation with increasing elevation may be influencing the NO_3 isotopic values collected from lake water.

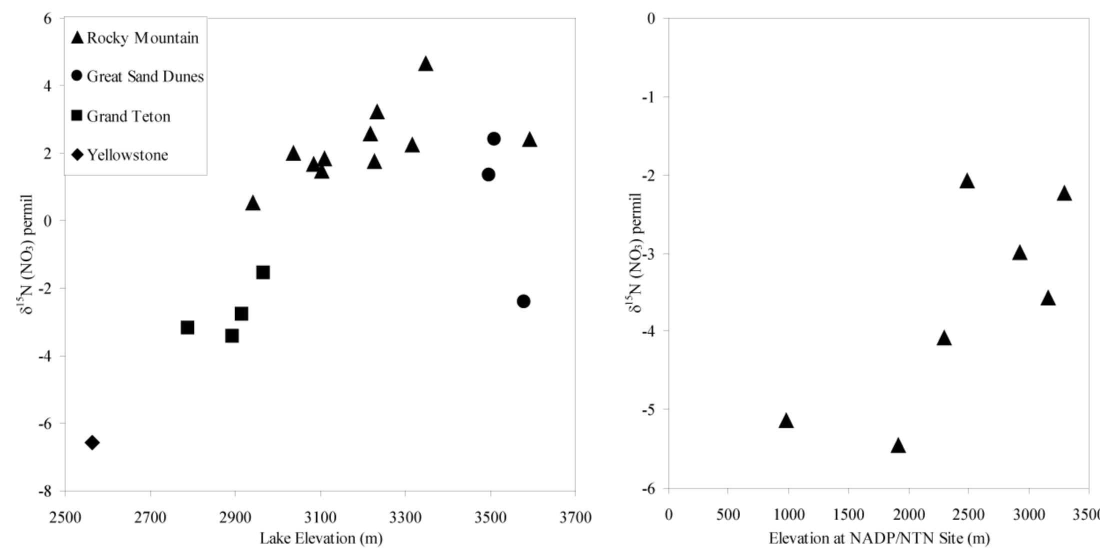

Figure 3-8a   Figure 3-8b

Figure 3-8. Lakes with elevation greater than 2500 meters compared with $\delta^{15}N(NO_3)$ values in lake water (3-8a) and $\delta^{15}N(NO_3)$ values in NADP/NTN precipitation compared with elevation (3-8b).

The source of NO_3 in lake and stream waters from mountain catchments in the Sierra Nevada has previously been examined with a simple end member mixing analysis using $\delta^{18}O$ (NO_3) values (Sickman et al., 2003). Here we have direct measurements of the atmospheric end-member, where $\delta^{18}O$ (NO_3) values range from +71 to +78 permil. The range in $\delta^{18}O$ (NO_3) values of lake water is from -5.7 to + 21.3 (Figure 5), with 95% of samples having $\delta^{18}O$ (NO_3) values less than +15 permil

(Figure 5). However, the terrestrial end-member is difficult to parameterize without direct measurements of subsurface $\delta^{18}O$ (NO_3) values. The generally accepted upper limit for $\delta^{18}O$ of microbial NO_3 is +15 permil for the terrestrial end-member (Kendall, 1998).

Several previous studies have evaluated N sources in stream water draining undisturbed catchments using a dual NO_3 isotope approach ($\delta^{18}O$ (NO_3) and $\delta^{15}N$ (NO_3)) to differentiate watershed NO_3 sources (Campbell et al., 2002; Burns and Kendall, 2002; Pardo et al., 2004). NO_3 in stream water was found to be mainly derived from nitrification in the Catskill mountains of New York (Burns and Kendall, 2002). Most of the NO_3 in streamflow was nitrified within two forested catchments in New Hampshire (Pardo et al., 2004). Similarly, most of the stream water NO_3 in Rocky had an isotope signature indicative of substantial biological cycling of atmospherically derived N prior to release from the ecosystem (Campbell et al., 2002).

However, assuming a $\delta^{18}O$ (NO_3) value of less than +15 permil for a terrestrial source could underestimate the actual contribution of atmospheric NO_3 to the NO_3 in lake waters. Laboratory incubation experiments and field studies have shown that the $\delta^{18}O$ (NO_3) formed by microbial nitrification range between +2 and +14 permil, assuming that soil water $\delta^{18}O$ (NO_3) values vary between -15 and -5 permil (Mayer et al., 2001). In the Catskill Mountains of New York, $\delta^{18}O$ (NO_3) values from +13.2 to +16.0 permil were measured for NO_3 derived by nitrification in incubated soil samples (Burns and Kendall, 2002). In Sleepers River, a snowmelt-dominated catchment in Vermont, $\delta^{18}O$ values of stream NO_3 ranged from -7.7 to +18.3 permil and generally were correlated with NO_3 concentrations (Ohte et al., 2004). It was concluded that a significant amount of NO_3 during snowmelt was directly from atmospheric deposition of NO_3 (Ohte et al., 2004). A $\Delta^{17}O$ of NO_3 technique was used to quantify these contributions in a semiarid ecosystem, and it was found that a large portion of atmospheric NO_3 in surface water did not undergo biologic processing before being exported from the system (Michalski et al., 2004). Thus, it is possible that the direct atmospheric contribution to NO_3 may be underestimated in earlier reports. A quantitative source apportionment of atmospheric NO_3 to the NO_3 in lake waters is difficult because of the wide range of $\delta^{18}O$ values from microbial nitrification.

The trend towards increasing $\delta^{15}N(NO_3)$ values with increasing concentrations of NO_3 in lake waters, which in turn are associated with increasing elevation and increasing inputs of DIN in wetfall, is intriguing. One potential explanation that deserves additional study is that the more enriched values of $\delta^{15}N$ (NO_3) may result from increasing rates of net nitrification in the watersheds. Well-drained soils typically show an increase in total soil-$\delta^{15}N$ with increasing soil depth and age (Kendall, 1998). This increase in $\delta^{15}N$ is attributed to fractionation during net mineralization and generally results from the metabolism of microbial heterotrophs that produce ^{15}N-enriched biomass as a result of excreting ^{15}N-depleted waste (Nadelhoffer and Fry, 1988; Nadelhoffer and Fry, 1994). DIN deposition in wetfall in the Rocky Mountains increases with increasing elevation (>2500m) compared to lower elevations (<2500), due in part to orographically enhanced precipitation amounts at high elevations (Williams and Tonnessen, 2002; Nanus et al., 2003). This increased DIN deposition in wetfall may lead to enhanced N cycling in high-elevation watersheds. The percent of DIN in wet deposition contributed by NH_4 is now approximately 50% of measured DIN in the Rocky Mountains (NADP, 2007). However, export of NH_4 in these watersheds is small, making up less than 6% of DIN compared to approximately 94% of DIN that is NO_3 in the baseflow lake outlet samples. This indicates that at least some of the NH_4 in atmospheric deposition of DIN in wetfall is mineralized and nitrified to NO_3 that is exported to lake waters (Campbell et al., 2000). Research in high-elevation areas of the Colorado Rockies show high rates of N-mineralization (Brooks and Williams, 1999) and less NH_4 assimilation due to a lack of vegetation, particularly in talus areas (Williams et al., 1997). These areas tend to be carbon limited (Ley et al., 2004), driving systems towards net nitrification (Brooks and Williams, 1999; Williams et al., 2007). It is possible that high rates of DIN deposition in wetfall at high elevations in the Colorado Rockies (Williams and Tonnessen, 2000; Burns, 2004), characterized by enriched $\delta^{15}N$ (NO_3) values (Elliott et al., 2007), may lead to enhanced nitrification and more enriched $\delta^{15}N$ values in the NO_3 exported to lake waters.

The role of denitrification in these systems was also considered, however, denitrification does not appear to substantially affect $\delta^{15}N$ (NO_3). If denitrification were important in these systems, a progression towards decreased NO_3 concentration would be expected. This trend does not exist in the data. Instead, the data show a significant correlation with increasing NO_3 concentrations and increasing $\delta^{15}N$ (NO_3) values in lake waters suggesting that enhanced nitrification may be important in these systems. Our results are consistent with previous work evaluating pathways for NO_3

release from an alpine watershed using $\delta^{18}O(NO_3)$ and $\delta^{15}N(NO_3)$ which found that denitrification does not affect fluxes of NO_3 from surface water or talus springs (Campbell et al., 2002).

The results presented in this study suggest that relatively high anthropogenic emissions of NO_x may be contributing to high NO_3 concentrations in high-elevation lakes in the southern Rocky Mountains through a combination of direct and indirect processes such as enhanced nitrification. This study provides valuable information on spatial patterns in $\delta^{15}N$ (NO_3) in lakes and precipitation across the Rocky Mountains and has important implications as N emissions (stationary and mobile sources) and inorganic N deposition continues to increase into the future. Results of this study may be helpful to resource managers who are considering the best way to reduce N emissions to control inorganic N deposition in sensitive, protected areas.

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CHAPTER 4

REGIONAL ASSESSMENT OF THE RELATION BETWEEN BASIN CHARACTERISTICS AND LAKE CHEMISTRY

ABSTRACT

Atmospheric deposition of acidic solutes can adversely affect sensitive aquatic habitats of high-elevation watersheds. The relation between basin characteristics, acidic deposition, and lake chemistry was evaluated in five national parks of the Rocky Mountains. Solute concentrations of 144 lakes and basin-characteristic information derived from GIS data sets were used to develop multivariate logistic regression models for nitrate (NO_3), sulfate (SO_4), and base cations. The model for NO_3 had the best statistical outcome, with the status of NO_3 in 95% of lakes in the validation data correctly predicted. Lakes with a high probability (> 67%) of having NO_3 concentrations greater than 5 microequivalents per liter (μeq/L) are located in basins with high elevations (> 2,700 meters), steep slopes (maximum slope > 43 degrees), low buffering capacity bedrock (> 66% of the basin area), and high NO_3 deposition (>1 kg/ha/yr NO_3-N deposition). SO_4 concentrations (>35 μeq/L) are significantly ($p< 0.1$) related to bedrock type (< 14% of composed granite, gneiss, quartzite, or schist) and SO_4 deposition (> 5 kg/ha/yr SO_4-S). Lakes with low base cation concentrations (< 124 μeq/L) are located in high-elevation basins (>3400 meters) with low buffering capacity bedrock (> 90%). At the regional scale, elevation had the greatest influence on lake chemistry followed by bedrock type, basin slope, and NO_3 deposition. The significant correlation ($p < 0.01$) between lake NO_3 concentrations and atmospheric NO_3 deposition at high elevations suggests that these

lakes may be showing a response to NO_3 deposition, through either direct (wet deposition) or indirect (enhanced nitrification) processes. Over 33% of lakes in Colorado national parks have a high probability for elevated NO_3 concentrations (> 5 μeq/L) and low base cation concentrations (< 124 μeq/L), and are coincident with areas that have increasing rates of inorganic nitrogen deposition. These findings may be used to improve long-term monitoring designs for high-elevation watersheds in the Rockies and the modeling approach may be transferable to other remote mountain areas of the United States and the world.

INTRODUCTION

In the western United States (US), energy generation, transportation, industry, and agriculture produce anthropogenic emissions of NO_x (nitrogen oxides), NH_3(ammonia), and SO_2 (sulfur dioxide) that contribute to deposition of dissolved inorganic nitrogen (DIN = nitrate (NO_3) + ammonium (NH_4)) and sulfate (SO_4) to high-elevation watersheds (Baron et al., 2000; Williams and Tonnessen, 2000). Over the last decade, DIN in wetfall has increased steadily in the Rocky Mountains (Baumgardner et al., 2002; Fenn et al, 2003; Nilles and Conley, 2001) while SO_4 deposition has decreased (Lynch et al. 1996). Previous work in US national parks of the Rocky Mountains suggests that high-elevation lakes are showing a response to atmospheric inputs of NO_3 and SO4 (Clow et al., 2002).

In the Rocky Mountains of Colorado, high atmospheric DIN deposition has caused considerable changes in the state and function of terrestrial and aquatic ecosystems in high-elevation basins (Baron, 2000; Burns, 2003; Campbell et al., 2000; Campbell et al., 2002; Mast et al., 2002, Williams et al., 1996). Nitrogen (N) deposition in excess of the total combined plant and microbial demand can cause watershed N saturation and increased rates of N leaching from soils to aquatic ecosystems (Aber et al., 1989), which is currently occurring in the Colorado Rockies (Williams et al., 1996; Baron 2006). This excess N can result in ecological effects in surface waters, including acidification and eutrophication. Eutrophication increases primary productivity in lakes and streams and alters diatom species distributions that form the base of the food web in many high-elevation lakes (Wolfe et al., 2001).

High-elevation watersheds have physical characteristics that make them particularly vulnerable to acidic deposition, including steep topography, thin and rocky soils, sparse vegetation, and a short growing season (Turk and Spahr, 1991). Lake sensitivity to acidification increases during periods of high influx of water that have shorter hydraulic residence times and rapidly transport atmospherically derived chemicals through or over the catchment into lakes without interacting with geologic weathering products that could buffer the acidity (Stoddard, 1987, Landers et al., 1987, Clow et al., 2002). In pristine mountain ecosystems, mineral weathering, cation exchange, and biologic processes in soils can affect water chemistry (Clow and Sueker, 2000). The acidification of lakes can be described as a loss of alkalinity over time, which can be related to the increase of acidic anions (NO_3 and SO_4), and a decline in bicarbonate (Brakke et al., 1989). These lakes also have typically low sum of base cations (Brakke et al., 1989) that increases their vulnerability to acidic deposition.

In the western US, high-elevation lakes with low acid-neutralizing capacity (ANC) concentrations (less than 100 μeq/L) are particularly vulnerable to DIN deposition (Williams and Tonnessen, 2000). A number of previous studies have evaluated the relation between basin characteristics and lake ANC concentrations (Berg et al., 2005; Nanus et al., 2005; Rutkowski et al., 2001; Sueker et al., 2001; Clow and Sueker, 2000; Melack et al., 1985; Corbin, 2004; Nanus et al., in review). However, few studies have explored relations between basin characteristics and additional solutes. Clow and Sueker (2000) found that slope steepness was negatively correlated with ANC and positively related to NO_3 concentrations in surface water in Rocky Mountain National Park and attributed this to fast hydrologic flushing rates on steep slopes with poorly developed soils and limited vegetative cover. Bedrock geology was significantly related to ANC and NO_3 in Rocky Mountain National Park (Clow and Sueker, 2000), ANC and SO_4 in Grand Teton National Park (Corbin et al., 2004)**,** and ANC in the Sierra Nevada (Melack et al., 1985). The presence of carbonate lithology in a basin can be very effective in neutralizing acidity in high-elevation basins and generally results in elevated concentrations of ANC (Berg et al., 2005). In the Southern Alps, Marchetto et al. (1994) found that the main factors influencing water chemistry in high alpine lakes were the weathering of silicate and calcite and nitrate uptake by vegetation, which accounted for most of the alkalinity production.

Results of the previous studies indicate that it may be possible to develop statistical models that use basin characteristics as explanatory variables to predict not only ANC concentrations in alpine and subalpine lakes, but also for NO_3, SO_4, and the sum of base cation concentrations (Ca, Mg, K, Na). Clow and Sueker (2000) predicted solute concentrations in Rocky Mountain National Park using multiple linear regression and found that predicted concentrations versus actual yielded the following for ANC (r^2 = 0.92), for NO_3 (r^2 = 0.97), and for SO_4 and base cations (r^2 = 0.54 to 0.86). However, when predicted values were compared with 1985 Western Lake Survey data less than 35% of the variance was explained (Clow and Sueker, 2000). Corbin (2004) evaluated the predictive ability of multiple linear regression models in Grand Teton and found that while they worked well for ANC and base cations, the regression model did not work well for acidic anions and overestimated NO_3.

Monitoring of surface-water quality in high-elevation watersheds provides the basis to assess current conditions and the long-term effects of acidic deposition to these aquatic ecosystems. However, monitoring across vast regions of the US Rockies is costly and impractical. Therefore, predictive models of lake chemistry are needed tools toward best management of these sensitive ecosystems. In a companion study, Nanus et al. (in review) demonstrate the value of using GIS-based basin characteristics and multivariate logistic regression models to predict ANC in lakes across the Rocky Mountain region. Because the model results from multivariate logistic regression are expressed in probability units of the predicted dependent variable in reference to the specified threshold, resource managers can develop monitoring programs based on any particular level of acceptable resource management risk. Predictive models for additional solutes in high-elevation lakes are also needed.

Therefore, the objectives of this study were to evaluate relations between GIS-based landscape and acidic deposition attributes and concentrations of NO_3, SO_4, and sum of base cations in alpine and subalpine lakes located in national parks of the Rocky Mountains. A model was developed for each constituent to evaluate regional relations and for inter-park comparison. This is the first study to include atmospheric deposition estimates in model development and to evaluate relations between these solutes and basin characteristics across an entire region in a systematic approach. This approach helps identify remote lakes that may need to be monitored and provides a framework for assessing other high-elevation environments of the western U.S. and the world.

METHODS

Study Area

Five national parks in the Rocky Mountains were studied, including Glacier National Park (GLAC), Montana; Grand Teton National Park (GRTE), Wyoming; Great Sand Dunes National Park and Preserve (GRSA), Colorado; Rocky Mountain National Park (ROMO), Colorado; and Yellowstone National Park (YELL), Wyoming. Four of the five parks are located along the Continental Divide (Figure 4-1, Table 4-1) and have a large number of alpine and subalpine lakes, ranging in elevation from less than 1,000 m in GLAC to more than 3,500 m in ROMO (Table 4-1).

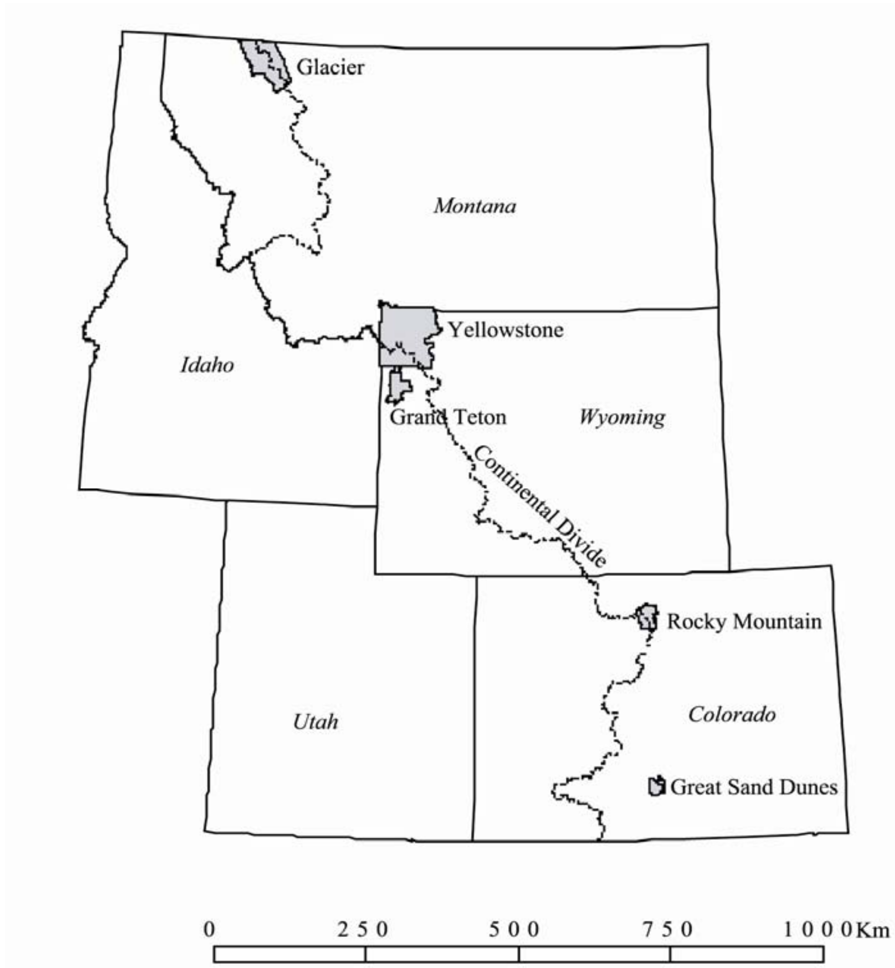

Figure 4-1. Location of national parks included in this study.

Table 4-1. National Parks in the Rocky Mountain region and their characteristics.

	Glacier	Yellowstone	Grand Teton	Rocky Mountain	Great Sand Dunes
Area (km^2)	4,101	8,980	1,256	1,078	343
Maximum Land-Surface Elevation (m)	3,190	3,122	4,198	4,346	4,317
Number of Lakes	244	294	106	114	11
Dominant Bedrock Types	Sedimentary	Volcanic	Granite	Granite	Granite and Sedimentary

These national parks are in glaciated mountain terrain (Madole, 1976; Richmond and Fullerton, 1986). Dominant bedrock types for each park are as follows: sedimentary rocks in GLAC (U.S. Geological Survey, 1992 b), granitic rocks in GRTE U.S. Geological Survey, 1992 a) and ROMO (U.S. Geological Survey, 1990), volcanic rocks in YELL (U.S. Geological Survey, 1988), and granitic and sedimentary rocks in GRSA (National Park Service, 2004). Average annual precipitation amounts in the Rocky Mountains increase as a function of elevation and latitude, mountainous areas above 3,000 m elevation generally receive at least 800 mm precipitation per year, most of which accumulates in a seasonal snowpack (Spatial Climate Analysis Service, 2000).

Many of the lakes used in model calibration and validation are located in alpine and subalpine terrain where soils are poorly developed, vegetation is sparse, and growing seasons are short. The lake selection process included a random selection component similar to the Western Lake Survey (Kanciruk et al., 1987; Landers et al., 1987; Silverstein et al., 1987), to allow extrapolation of the results to a broad population of lakes. Lakes with a surface area greater than 1 hectare (ha) were used to avoid inclusion of small tarns and ponds in calibration and validation data sets. Model results were then applied to all lakes greater than 1 ha in area, 769 lakes in the five parks included in this study.

Description of Data Used in the Statistical Models

Chemical concentrations for 144 alpine and subalpine lakes greater than 1 hectare, which represents approximately 20% of the total number of lakes (n=769), from the five national parks were used in the analysis. The study was focused on baseflow conditions when chemical concentrations are relatively consistent over time; therefore, only data from lakes that were sampled during late summer through early fall were included. Historical data were available for YELL (Woods and Corbin, 2003b; Gibson, 1980), GLAC (Landers et al., 1987; Ellis et al., 2002), GRSA (Bunch, GRSA, written commun., 2004), ROMO (Clow et al., 2002), and GRTE (Landers et al., 1987; Woods and Corbin, 2003a; Corbin et al., 2006; Tonnessen and Williams, NPS and INSTAAR, written commun., 1997).

Chemical concentrations in high-elevation lakes in these five parks were sampled in 1985 and in 1999. Over this period, ROMO was the only national park with significant changes in baseflow concentrations (Clow et al., 2003). Chemical concentrations in ROMO were not variable across the concentration thresholds used for this study. A number of different data sources were combined to evaluate long term trends in the historical data for GLAC, ROMO, and GRSA (Mast, 2007) and for GRTE (Woods and Corbin, 2003a) and YELL (Woods and Corbin, 2003b), and indicate that the different data sources can be combined for these parks. Temporal trends in surface water quality in GLAC were not significant, except for a slight downward trend in SO_4 ($p<0.01$) which may be attributed to different analytical techniques (Mast, 2007). Results of this previous work indicate that data collected at the five parks may be combined for sites with multiple years of chemical concentration data from one data set or multiple data sets using samples collected from lake outlets during baseflow conditions, for the multivariate logistic regression analysis.

To expand upon the available water quality data, lakes were sampled during late summer through early Fall 2004 in all five parks. Samples were collected at 58 randomly selected high-elevation lakes that are spatially distributed within each of the national parks (Nanus et al., 2005). Lake samples were filtered (0.45 micron) and collected at the lake outlets and chemical concentrations were analyzed following standard procedures at the Kiowa Environmental Chemistry Laboratory, Boulder, Colorado (Seibold, 2001), which specializes in analysis of extremely dilute waters such as those found in the study area.

Physical characteristics that were tested in the statistical model for correlation with NO_3 concentration, SO_4 concentration, DIN (NO_3 + NH_4) concentration, and sum of base cations concentration included elevation, slope, aspect, bedrock geology, and soils. Boundaries for the watershed of each lake were delineated using a 10-m digital elevation model (DEM) (U.S. Geological Survey, 2000 a, b). To evaluate the effect of error associated with basin delineation, basin boundaries delineated manually and automated in GIS, were compared and the difference in model outputs were calculated at less than 5%.

For each lake basin, the 10-m DEM was used to calculate mean elevation, slope, basin area, lake/basin area, percentage of steep slopes (slopes >30 degrees), and percentage of aspect (by 45-degree increments). Additional basin characteristics that were derived from National Park Service GIS data include watershed area, lake area, ratio of lake area to watershed area, ratio of lake perimeter to lake area, percentage watershed covered by rock type (National Park Service 2004; U.S. Geological Survey, 1988, 1990, 1992 a, b), percentage watershed covered by soil type

(National Park Service, 1994a, 1997, 1999; Rocky Mountain National Park GIS Program, 1995 a), and percentage watershed covered by vegetation type (National Park Service, 1981, 1990, 1994 b; Rocky Mountain National Park GIS Program, 1995 b). Errors associated with a given resolution for each basin characteristic were evaluated to determine whether data with different levels of resolution could be combined.

Bedrock lithologies were grouped into six different geochemical classes that were ranked from low to high on the basis of buffering capacity of the bedrock (Nanus and Clow, 2004; Nanus et al., 2005). The geochemical rankings (GC) are as follows: GC 1: Low buffering capacity: gneiss, quartzite, schist, granite; GC 2: Moderate buffering capacity: andesite, dacite, diorite, phyllite; GC 3: High buffering capacity: basalt, gabbro, greywacke, argillite, undifferentiated volcanics; GC 4: Very high buffering capacity: amphibolite, hornfels, paragneiss, undifferentiated metamorphic rocks; GC 5: Class 5, Extremely high buffering capacity: metacarbonate, marine sedimentary rocks, calc silicate and basic intrusive rocks; and GC 6: Class 6, Unknown buffering capacity. Vegetation type was classified into unvegetated, forest, and subalpine meadow (Nanus et al., 2005). Unvegetated areas were composed of snow, ice, rock, and water. Soil type was not included in the regional model, because not all parks had complete digital soil data, and the data that were available were not comparable across all five parks. However, soil type was evaluated for individual parks.

Atmospheric factors, including precipitation amount and atmospheric deposition rates of acidic solutes were evaluated for inclusion in the statistical models, following protocols described by Nanus et al. (2003). The mean-annual precipitation variable for each basin was derived from a 30-year (1961-1990) average annual precipitation grid (PRISM) (Spatial Climate Analysis Service, 2000). Average annual deposition loadings of hydrogen ion, inorganic N, and sulfate were determined by multiplying the precipitation grid with kriged chemical concentrations (Nanus et al., 2003), these deposition estimates were then calculated for each basin for inclusion in the statistical analysis. The spatial variability in solute deposition was largely controlled by precipitation amount (Nanus et al., 2003).

For the individual parks, spearman correlation matrices were used to identify relationships between NO_3, DIN, SO_4, sum of base cations and the basin characteristics described above and to determine whether there were differences between parks.

Model Development and Validation

Multivariate logistic regression was used to predict the probability for lakes to have elevated NO_3 concentration, SO_4 concentration, DIN (NO_3 + NH_4) concentration, and low sum of base cation concentrations for the Rocky Mountain region. Logistic regression differs from linear regression in that the dependent value from the logistic regression model is expressed as the probability of exceeding a threshold value, rather than a discrete value (Helsel and Hirsch, 1992). The general linear model relates continuous dependent variables to independent variables that are either classification variables or continuous variables (SAS Institute, 1990). This works well if the assumptions of linear regression are satisfied. However, with a dichotomous (binary) dependent variable, assumptions of linearity, normality of the error term, and homoscedasticity (constant variance of the error term) are violated (Menard, 2002).

The logistic regression approach uses the maximum likelihood method to fit linear logistic regression models for binary response data (SAS Institute, 1999). The probability (p) of being in a response category is defined by the odds ratio, the log of which transforms a variable between 0 and 1 into a continuous variable that is a linear function of the explanatory variables (Helsel and Hirsch, 1992) as follows:

$$\ln\left(\frac{p}{1-p}\right) = b_o + b_x \qquad \text{(eqn. 1)}$$

where b_o is the intercept, x is a vector of k independent variables, and b_x includes the slope coefficients for each explanatory variable. The logistic transformation is used to return the predicted values of the response variable to probability units, with the logistic regression model as follows:

$$Logit(P) = \frac{e^{(b_o + b_x)}}{1 + e^{(b_o + b_x)}} \qquad \text{(eqn. 2)}$$

where *Logit (P)* is the probability that the chemical concentration is greater than a specified concentration threshold (binary response) (Helsel and Hirsch, 1992).

Multivariate logistic regression models were developed for NO_3, DIN, SO_4, and base cations using meaningful thresholds based on the data distribution. Previous work has indicated that surface waters with NO_3 concentrations less than 5 μeq/L are considered generally pristine (Inyan and Williams, 2001; Williams and Tonnessen, 2000, Van Miegroet, 1994; Lepori et al., 2003). The 67th percentile of the data distribution across the 5 parks for NO_3 = 5 μeq/L, and for DIN = 5 μeq/L. Thus, having used the 67^{th} percentile of the data distribution for both NO_3 and DIN, it will also be used to determine thresholds for model development for SO_4 and base cations. For sulfate in pristine systems, published background sulfate concentrations range from 10-60 μeq/L in dilute lakes (Almer, 1980, Brakke et al., 1989, Gibson et al., 1983). The 67^{th} percentile for SO_4 is 35 μeq/L which fits well into this range of values. For base cations, the probability below 124 μeq/L will be predicted, which is the 33^{rd} percentile of data distribution and well below the published values that indicate low base cations as less than 400 μeq/L for sensitive lakes (Lepori et al., 2003). Thus, the probability that NO_3, DIN, and SO_4 concentrations will be higher, and Bc concentrations lower, than the 67^{th} percentile of the distribution of observed lake concentrations based on basin characteristics will be predicted. For each of the thresholds, probabilities for sensitive lakes were calculated from 0-100%. For the purpose of presentation, the results were binned into three groups representing: 0-33% (low probability), 33-67% (medium probability), and 67-100% (high probability).

Basin characteristics derived from GIS were used as explanatory variables and NO_3, DIN, SO_4, and Bc (sum of base cations = Ca + Mg + Na + K) data (n=144) were used as the dependent variables to calibrate the logistic regression models. A total of 108 lakes were used to calibrate the models and a randomly selected subset of 25% of lakes in each park (n=36) were reserved to evaluate the models. First, all explanatory variables were tested independently using univariate logistic regression and the explanatory variables that have significant influence at p-value ≤ 0.1 were tested in the multivariate logistic regression models. A p-value of ≤ 0.1 was chosen over a p-

value $\leq$ 0.05 so that more meaningful variables could be included in the multivariate analysis (Hosmer and Lemeshow, 1989).

To evaluate the calibration of the logistic regression models for each National Park, model-based predicted probabilities were compared to measured concentrations by using the Hosmer-Lemeshow (HL) goodness-of-fit test (Hosmer and Lemeshow, 1989). A subset of lakes was excluded from calibrations to provide an independent data set to evaluate the calibration results. To evaluate model agreement, measured concentrations were compared to predicted concentrations by randomly grouping the verification lakes with measured concentrations into 10 groups with an equal number of lakes. These random groupings of 10 % were used to evaluate model agreement between measured and predicted concentration. A higher HL value indicates a well-calibrated model (Hosmer and Lemeshow, 1989). The *c*-statistic is a measure of rank correlation of ordinal variables (SAS Institute, 1990). The *c*-statistic is normalized so that it ranges from 0 (no association) to 1 (perfect association). It is a variant of Somers' D index (SAS Institute, 1990). The multivariate logistic regression model with the best statistical outcome with respect to predicting probability measured by r^2, the *c*-statistic, and the HL goodness-of-fit test for NO_3, DIN, SO_4, and Bc is presented for the Rocky Mountain region.

Because the explanatory variables are reported in different units, the model coefficients were standardized in the final multivariate logistic regression models to compare their relative effect on model prediction. The standardized coefficients were calculated using a technique that follows the protocol of Menard (2002). The basin characteristic with the greatest influence on the model has the largest standardized logistic regression coefficient (b*).

The resulting multivariate logistic regression models were applied to all lakes greater than or equal to 1 ha in the five parks. Modeling results were validated using a randomly selected subset of lakes, 25% of total lakes (n=36), that were not included in model development. For the regional model, the actual percentage of lakes with concentrations above a threshold for NO_3, DIN, and SO_4 (and below for Bc) were calculated for the calibration (n=108) and validation (n= 36) set of data separately and is equal to the number of lakes with measured concentrations above the threshold for NO_3, DIN, and SO_4 (and below for Bc) divided by the total number of analyses for each 20 % of decile data. For example, NO_3 concentrations from the measured lakes were converted to binary classification of "one" for NO_3 concentrations > 5 μeq/L and "zero" for NO_3 concentrations < 5 μeq/L. The conversion to binary classification enabled a direct comparison between the percentage of measured NO_3 concentrations and the average predicted probability within each 20 % decile of the data.

RESULTS

Chemical concentrations of the 144 lakes used for the model calibration and validation ranged widely. For NO_3 and DIN (NH_4 + NO_3) concentrations ranged from below the detection limit (0.2 μeq/L) to 38.9 μeq/L, for SO_4 from 2 μeq/L to 2937 μeq/L, and for Bc (Ca + Mg + K + Na) from 33.9 μeq/L to 6556 μeq/L (Table 4-2). The concentrations also differed among parks (Table 4-2, Figure 4-2).

Table 4-2. Chemical concentration data used in model development (calibration and validation data combined).

		ANC(μeq/L)	Ca(μeq/L)	Mg(μeq/L)	Na(μeq/L)	K(μeq/L)	NH4(μeq/L)	Cl(μeq/L)	NO_3(μeq/L)	SO_4(ueq/L)	Sum Base Cations(μeq/L)	DIN(μeq/L)
Regional	Minimum	14.7	16.1	6.1	5.4	0.4	≤0.2	0.9	≤0.2	2.1	33.9	≤0.2
(n=144)	Quartile 1	65.2	63.1	16.5	15.0	3.3	≤0.2	2.0	0.3	17.1	113.2	1.0
	Median	127.5	93.6	35.8	23.7	4.9	0.0	3.1	2.1	30.1	178.9	2.6
	Quartile 3	409.7	295.5	115.0	39.9	13.3	0.9	6.5	7.1	41.7	605.9	7.6
	Maximum	1621.1	1196.6	836.2	4948.0	891.5	4.0	1938.6	38.9	2937.3	6556.0	38.9
	67th percentile	297.8	187.8	71.4	30.9	10.1	0.7	5.0	5.1	36.4	352.2	5.3
GLAC	Minimum	37.1	22.7	17.1	5.4	0.8	≤0.2	0.9	≤0.2	3.6	46.7	≤0.2
(n=30)	1st Quartile	542.9	326.9	202.7	12.8	2.8	0.3	1.5	1.2	25.3	597.7	1.0
	Median	821.9	572.9	290.9	15.6	3.7	0.5	2.5	1.9	34.9	833.0	1.6
	3rd Quartile	1121.8	745.9	361.6	19.6	4.6	1.0	3.1	3.8	52.4	1157.4	3.2
	Maximum	1584.2	1002.0	836.2	50.8	5.9	2.9	11.4	11.6	133.5	1704.6	11.6
	67th percentile	1008.6	651.5	337.5	18.6	4.2	0.9	2.8	2.4	39.8	1007.6	2.7
GRSA	Minimum	110.1	108.1	12.3	13.9	2.0	≤0.2	2.2	≤0.2	22.2	136.3	1.3
(n=6)	1st Quartile	163.7	134.5	22.5	22.3	4.0	≤0.2	3.0	2.5	29.6	185.2	3.0
	Median	211.2	201.4	37.2	29.3	8.3	0.9	4.8	3.7	41.8	281.3	4.2
	3rd Quartile	255.6	285.1	42.0	35.1	12.7	1.3	6.7	8.9	57.8	368.1	9.2
	Maximum	418.4	379.5	72.9	45.4	13.7	1.8	7.1	21.8	73.8	511.1	23.6
	67th percentile	240.9	271.0	40.9	32.1	11.8	1.1	6.2	6.4	50.8	353.9	7.1
GRTE	Minimum	36.9	32.4	10.5	6.8	0.4	≤0.2	1.6	≤0.2	5.9	62.3	≤0.2
(n=23)	1st Quartile	83.6	66.2	17.2	13.9	6.4	≤0.2	2.4	1.5	13.1	107.1	2.7
	Median	116.8	86.3	28.1	17.7	10.3	0.8	3.1	4.7	18.6	147.3	4.7
	3rd Quartile	234.5	189.0	62.5	25.2	15.6	1.0	4.3	5.7	31.3	322.0	5.9
	Maximum	846.7	663.7	214.8	50.1	29.8	1.7	8.1	20.1	214.5	896.5	20.1
	67th percentile	171.8	161.8	40.3	23.8	14.2	0.9	3.7	5.3	29.1	238.1	5.7
ROMO	Minimum	14.7	16.1	6.1	8.0	2.0	≤0.2	1.2	≤0.2	10.5	33.9	≤0.2
(n=50)	1st Quartile	44.3	53.1	12.5	17.7	2.7	≤0.2	1.7	1.4	17.0	88.4	2.5
	Median	55.7	68.8	15.8	22.5	3.5	0.0	2.1	7.6	27.9	116.7	7.8
	3rd Quartile	84.4	80.8	20.3	28.4	4.5	0.9	3.5	14.7	35.2	133.8	15.1
	Maximum	304.3	188.5	91.8	132.7	11.4	4.0	7.0	38.9	66.0	424.4	38.9
	67th percentile	80.4	76.2	18.8	27.3	3.9	0.7	2.8	11.5	33.9	125.2	11.9
Yell	Minimum	52.4	32.6	16.5	11.7	4.1	≤0.2	3.3	≤0.2	2.1	96.0	≤0.2
(n=35)	1st Quartile	155.1	86.9	41.1	39.9	13.7	0.7	9.2	≤0.2	20.8	199.9	0.7
	Median	242.6	139.6	60.2	82.6	30.5	1.0	14.6	0.7	35.2	373.8	1.3
	3rd Quartile	614.6	349.8	107.8	280.6	49.5	1.3	28.2	1.6	67.8	924.8	1.9
	Maximum	1621.1	1196.6	517.6	4948.0	891.5	3.5	1938.6	5.2	2937.3	6556.0	7.6
	67th percentile	410.6	232.8	82.3	147.0	41.4	1.2	28.2	1.4	61.4	805.5	1.4

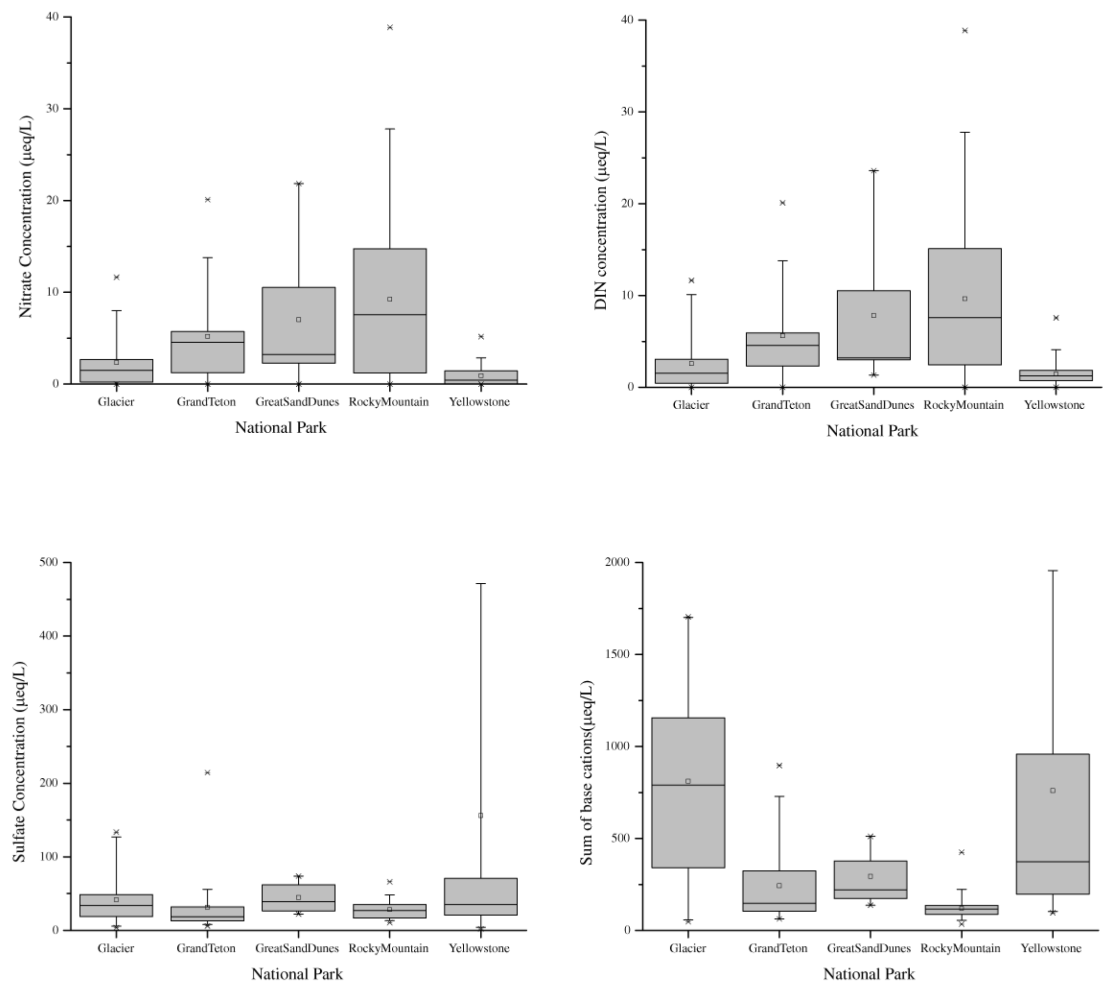

Figure 4-2. Solute concentrations at 144 lakes aggregated by National Park. Data sources include: Bunch, GRSA, written commun., 2004; Clow et al., 2002; Corbin et al., 2004; Ellis et al., 2002; Gibson, 1980; Landers et al., 1987; Nanus et al., in review; Tonnessen and Williams, NPS and INSTAAR, written commun., 1997; Woods and Corbin, 2003a; Woods and Corbin, 2003b.

The median NO_3 concentration of 7.6 μeq/L and DIN concentration of 7.8 μeq/L for ROMO (n=50), were above the 67th percentile of the data in all other parks and above the maximum NO_3 concentration of 5.2 μeq/L and DIN concentration of 7.6 μeq/L in YELL (n=35) for the historical data combined with the data collected in 2004 (Table 4-2). The median SO_4 concentration of 35 μeq/L for YELL (n=35) was above the 67th percentile of the data in ROMO and GRTE, and YELL had a maximum SO_4 concentration of 2937 μeq/L (Table 4-2). The median Bc concentration of 833 μeq/L for GLAC (n=30) was above the 67th percentile of the data in all other parks, and YELL had a maximum Bc concentration of 6556 μeq/L (Table 4-2).

The amount of bedrock composed of low buffering capacity also varied widely among the parks (Table 4-3). The median amount of low-buffering capacity bedrock in lake basins of ROMO was 97%, while lake basins in YELL have a median of 0%. In general, lakes in parks with high NO_3, DIN, and low Bc concentrations (ROMO, GRSA) were associated with high elevations and large amounts of low-buffering capacity bedrock.

For the individual parks, spearman correlation matrices between NO_3, DIN, SO_4, Bc and physical basin characteristics are shown in Table 4-4, along with the regional results for comparison. Differences in significant relations exist between parks. For example in ROMO and for the region, NO_3 was significantly ($p < 0.05$) and positively related to mean elevation, steep slopes, unvegetated area, NO_3 deposition, and was inversely related to forested area. However, in YELL and GRSA, NO_3 was only significantly related to mean elevation ($p < 0.01$), and NO_3 was not significantly related to any basin characteristics in GLAC. NO_3 in ROMO and for the region was also inversely related to ANC concentration ($p < 0.001$). SO_4 was not significantly related to any basin characteristics in ROMO or GRSA, but was significantly and positively related to watershed area in GRTE ($p < 0.01$) and was significantly inversely related to bedrock GC1 in GLAC and for the region ($p < 0.01$). There was a significant ($p < 0.05$) inverse relation between low base cation concentrations and mean elevation for ROMO, YELL, and for the region, and it was inversely related to low buffering capacity bedrock in GRTE, GLAC, and for the region.

Table 4-3. Summary statistics for data used in regional model development for nitrate, dissolved inorganic nitrogen, sulfate, and base cations.

		Minimum Elevation (m)	Max Slope (deg)	Low Buffering Bedrock (%)	NO_3 Deposition (kg/ha/yr N)	SO_4 Deposition (kg/ha/yr S)
GLAC	min	1126	57.0	0.0	0.8	3.4
n=30	median	1773	79.0	0.4	1.3	4.9
	max	2344	87.0	1.0	1.8	6.7
GRSA	min	3496	43.0	0.5	1.4	4.8
n=6	median	3566	59.5	0.9	1.9	7.0
	max	3774	65.0	1.0	2.1	7.4
GRTE	min	2088	46.1	0.7	0.6	2.5
n=23	median	2799	69.3	1.0	1.4	5.2
	max	3066	79.1	1.0	1.6	6.0
ROMO	min	2640	22.0	0.5	0.9	3.9
n=50	median	3306	58.5	1.0	1.8	6.2
	max	3625	71.0	1.0	3.0	7.4
YELL	min	1953	40.2	0.0	0.5	1.6
n=35	median	2411	69.1	0.0	1.1	3.4
	max	2699	83.2	0.9	2.2	5.7
Regional	min	1126	22.0	0.0	0.5	1.6
n=144	median	2686	65.9	0.8	1.4	5.0
	max	3774	87.0	1.0	3.0	7.4

Table 4-4. Spearman correlation coefficients for relationships between select basin physical characteristics and lake water concentrations in each individual park and the region for NO_3, DIN, SO_4, and sum of base cations (Ca, Mg, K, Na).

	Topography			Vegetation Classes			Geology Classes			Deposition		
	Watershed Area	Mean Elevation	Slope > 30 deg	Unvegetated	Forest	Subalpine Meadow	Bedrock GC1	Bedrock GC4	Bedrock GC5	NO_3 Deposition	DIN Deposition	SO_4 Deposition
ROMO(n=50)												
NO_3	-0.04	†0.31	*0.54	*0.38	†-0.34	-0.15	0.17	-0.11	na	†0.28	†0.31	*0.56
DIN	-0.07	†0.33	*0.52	*0.37	†-0.34	-0.12	0.20	-0.13	na	0.23	0.27	*0.53
SO_4	-0.11	-0.21	0.05	-0.12	0.06	-0.01	-0.11	0.07	na	0.18	0.21	0.24
Sum Base Cations	-0.17	*-0.46	-0.11	*-0.34	†0.30	-0.04	-0.12	0.07	na	0.07	0.12	0.20
GRTE (n=23)												
NO_3	0.34	0.20	0.23	0.15	†0.48	†-0.47	0.01	-0.01	0.08	0.14	0.11	-0.10
DIN	0.29	0.24	0.24	0.08	†0.46	†-0.45	-0.01	0.02	0.09	0.07	0.05	-0.11
SO_4	*0.53	-0.20	-0.33	0.37	-0.33	0.32	-0.33	0.40	0.19	*0.65	*0.62	0.39
Sum Base Cations	0.39	-0.09	-0.13	0.30	†-0.41	0.37	*-0.62	*0.48	*0.59	*0.72	*0.68	*0.57
GLAC (n=30)												
NO_3	0.24	-0.27	-0.08	-0.24	0.30	na	-0.25	0.28	0.24	-0.13	-0.12	-0.23
DIN	0.25	-0.31	-0.11	-0.21	0.28	na	-0.22	0.25	0.22	-0.11	-0.10	-0.17
SO_4	0.35	-0.14	0.07	-0.34	*0.48	na	*-0.57	†0.43	*0.51	0.04	0.04	-0.21
Sum Base Cations	0.32	-0.10	0.24	*-0.45	*0.52	na	*-0.67	0.33	*0.58	0.08	0.07	-0.03
YELL (n=35)												
NO_3	0.04	*-0.42	0.15	0.07	-0.07	0.19	0.14	0.01	0.08	0.06	0.08	0.24
DIN	0.08	-0.19	0.07	0.00	-0.22	†0.34	-0.10	0.12	0.10	-0.06	-0.05	-0.03
SO_4	0.31	†-0.38	0.19	-0.17	0.04	0.27	-0.08	0.19	-0.11	-0.01	0.02	0.14
Sum Base Cations	†0.35	†-0.39	0.22	0.07	-0.32	*0.45	0.00	*0.50	0.18	-0.18	-0.13	0.06
GRSA (n=6)												
NO_3	0.26	*0.94	0.31	-0.20	0.20	-0.26	0.14	na	-0.13	-0.36	-0.36	-0.41
DIN	0.26	*0.94	0.31	-0.20	0.20	-0.26	0.14	na	-0.13	-0.36	-0.36	-0.41
SO_4	0.54	-0.43	0.09	0.26	-0.26	0.60	-0.03	na	-0.65	0.24	0.24	0.26
Sum Base Cations	0.71	-0.03	0.26	-0.60	0.60	0.66	-0.46	na	-0.13	-0.72	-0.72	-0.68
Regional (n=144)												
NO_3	-0.02	*0.44	*0.35	*0.26	*-0.21	0.04	*0.40	-0.13	-0.06	*0.37	*0.23	*0.53
DIN	-0.03	*0.47	*0.31	*0.21	†-0.18	0.11	*0.41	-0.11	-0.12	*0.33	*0.18	*0.48
SO_4	*0.25	†-0.17	-0.07	-0.08	0.07	0.08	*-0.32	*0.21	†0.17	0.06	0.13	-0.02
Sum Base Cations	*0.29	*-0.66	0.01	-0.09	0.08	-0.05	*-0.59	*0.34	*0.51	*-0.29	0.04	*-0.32

*correlation is signifiant at the 0.01 level

†correlation is significant at the 0.05 level

na = not available

Solute concentrations of NO_3, DIN, SO_4, and Bc were not normally distributed across all five parks in the region, with median concentrations of 2.1 μeq/L, 2.6 μeq/L, 30 μeq/L, and 179 μeq/L respectively (Figure 4-2, Table 4-2). Therefore, the assumptions underlying multiple linear regression, such as normal distribution, were not satisfied. Because logistic regression is used to explore the relations between binary response and a set of explanatory variables and does not require a normal distribution, it is well-suited for the NO_3, DIN, SO_4, and Bc data.

The results of the multivariate logistic regression regional model analysis for NO_3, DIN, SO_4, and Bc concentrations are summarized in Table 4-5. Results of the regional model calibration indicate that both NO_3 and DIN concentrations, for a threshold of greater than 5 μeq/L, are significantly related to elevation of lake outlet ($p \leq 0.001$), the percentage of bedrock with low buffering capacity bedrock ($p = 0.01$), maximum basin slope ($p \leq 0.01$), and NO_3 deposition ($p = 0.1$) (Table 4-5). The resultant NO_3 probability equation was applied to the 769 lakes greater than 1 ha in the five parks. The HL goodness-of-fit yielded an $r^2 = 0.93$, and the c-statistic = 0.90 indicating a good model fit to the calibration data (Table 4-5).

$$Logit(P) = \frac{e^{(-19.2 + (0.0026 \times elevation) + (0.10 \times \max basin slope) + (3.4 \times \% basin with low buffering capacity bedrock) + (1.2 x NO3 deposition))}}{1 + e^{(-19.2 + (0.0026 \times elevation) + (0.10 \times \max basin slope) + (3.4 \times \% basin with low buffering capacity bedrock) + (1.2 x NO3 deposition))}}$$

(eqn3)

where *Logit (P)* and *e* are defined in equation 1.

Table 4-5. Results of multivariate logistic regression analyses, logistic regression coefficients, and *p*-values.

Regional Multivariate Model Constituent (number of lakes (n) =108)	Threshold (μeq/L)	Explanatory Variable	Explanatory variable coefficient (p-value)	Standardized coefficients (b*)	HL Goodness of fit (r^2)	c-statistic	Model Validation (r^2) (n=36)
Nitrate	5	Elevation at Lake Outlet	0.0026(0.002)	0.416	0.94	0.90	0.95
		Maximum Basin Slope	0.10(0.01)	0.321			
		Low Buffering Capacity Bedrock	3.4(0.01)	0.356			
		Nitrate Deposition	1.2(0.1)	0.137			
Dissolved Inorganic Nitrogen	5	Elevation at Lake Outlet	0.0033(<0.001)	0.490	0.94	0.92	0.75
		Maximum Basin Slope	0.14(0.004)	0.400			
		Low Buffering Capacity Bedrock	3.5(0.01)	0.337			
		Nitrate Deposition	1.3(0.1)	0.139			
Sulfate	35	Low Buffering Capacity Bedrock	-2.1(0.003)	-2.700	0.78	0.68	0.73
		Sulfate Deposition	0.39(0.07)	4.414			
Sum of Base Cations	124	Elevation at Lake Outlet	0.002(0.002)	0.364	0.89	0.8	0.56
		Low Buffering Capacity Bedrock	1.5(0.08)	0.213			

The equation for DIN uses the same variables with different coefficients as shown in Table 4-5. NO_3 and DIN are highly correlated (spearman correlation coefficient r = 0.95; p<0.0001). Approximately 94% of DIN is NO_3 in these lakes at the 67th percentile of the data distribution. Because the resultant models of DIN and NO_3 are so similar, since the NH_4 contribution to DIN is relatively low (Table 4-2), only the NO_3 values are reported for the model results.

SO_4 concentrations are positively related to SO_4 deposition (p-value=0.07) and are negatively related to bedrock GC1 (*p*-value = 0.003) (Table 4-5). About 33% of the lakes in the calibration data set have SO_4 concentrations greater than 35 μeq/L, the 67th percentile of the data distribution (Figure 4-2). The resultant probability equation was applied to the lakes.

$$Logit\ (P) = \frac{e^{(-1.4 - (2.1 \times \%basin\ with\ low\ buffering\ capacity\ bedrock) + (0.39\ xSO4\,deposition))}}{1 + e^{(-1.4 - (2.1 \times \%basin\ with\ low\ buffering\ capacity\ bedrock) + (0.39\ xSO4\,deposition))}} \text{(eqn 4)}$$

The HL goodness-of-fit yielded an r^2= 0.78, and the *c*-statistic = 0.68 (Table 4-5), which were not as high as for NO_3.

Bc concentrations less than 124 μeq/L, are significantly related to the percentage of bedrock with low buffering capacity (*p*-value = 0.08), and elevation of lake outlet (p-value=0.002) (Table 5). The probability equation follows:

$$Logit\ (P) = \frac{e^{(-6.2 + (0.002 \times elevation) + (1.5 \times \%basin\ with\ low\ buffering\ capacity\ bedrock))}}{1 + e^{(-5.8 + (0.002 \times elevation) + (1.5 \times \%basin\ with\ low\ buffering\ capacity\ bedrock))}} \text{(eqn 5)}$$

The HL goodness-of-fit yielded an r^2 = 0.89, and the *c*-statistic = 0.81 (Table 4-5).

The presence of multicollinearity was evaluated for the explanatory variables as defined by Allison (1991), whereby a tolerance < 0.4 or variance inflation factor > 2.5

might indicate the presence of multicollinearity. Results indicate that multicollinearity is not present in the final multivariate logistic regression models (Table 4-6).

Table 4-6. Multicollinearity diagnostics for regional nitrate, sulfate, and sum of base cation models.

	Tolerance	Variance Inflation Factor
Regional Nitrate Model		
Elevation at Lake Outlet	0.41	2.44
Maximum Basin Slope	0.68	1.46
Low Buffering Capacity Bedrock	0.59	1.68
Nitrate Deposition	0.70	1.42
Regional Sulfate Model		
Low Buffering Capacity Bedrock	0.68	1.48
Sulfate Deposition	0.68	1.48
Regional Sum of Base Cation Model		
Low Buffering Capacity Bedrock	0.71	1.4
Sulfate Deposition	0.71	1.4

Resultant probability maps for NO_3, SO_4, and Bc are shown in Figures 4-3 through 4-5. Regional model results indicate that ROMO and GRSA have a high probability (defined as greater than 67% probability) for elevated NO_3 concentrations and low Bc concentrations. Thirty-nine percent of lakes in ROMO and GRSA had a high probability for lake NO_3 concentrations greater than 5 μeq/L (Figure 4-3). Few lakes in GRTE and YELL had a high probability for NO_3 concentrations greater than 5 μeq/L (Figure 4-3) and no lakes in GLAC. The lakes that had a high probability of having NO_3 concentrations greater than 5 μeq/L in the regional model are located at elevations greater than 2,700 m, with a maximum basin slope greater than 43 degrees, with greater than 66 % of the basin with bedrock of low buffering capacity (ie. quartzite, granite, gneiss, and schist), and with NO_3 deposition greater than 1 kg/ha/year NO_3-N deposition.

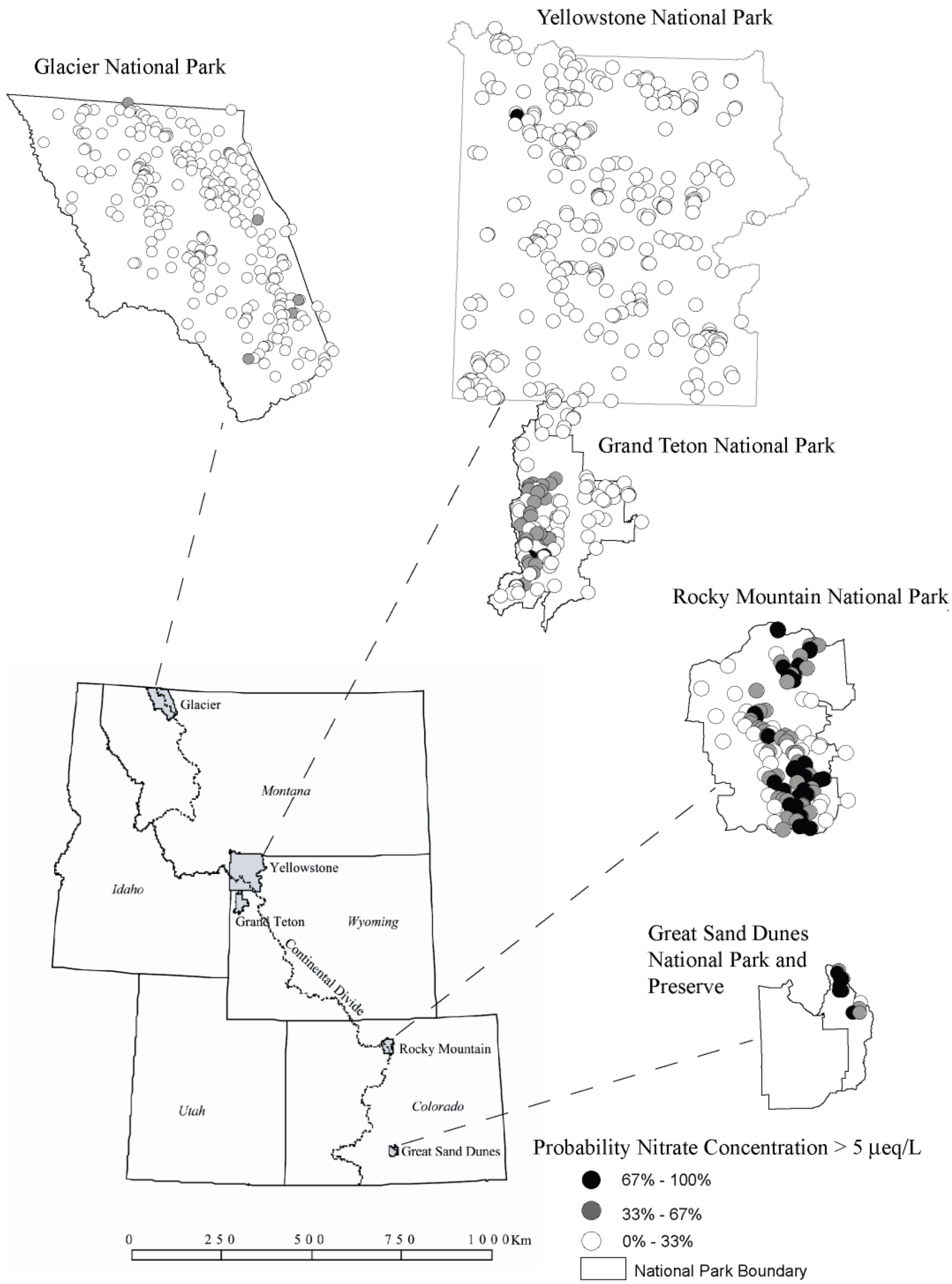

Figure 4-3. Probability of lake nitrate concentrations greater than 5 µeq/L in five Rocky Mountain National Parks. All probabilities ranging from 0-100% were evaluated; the data are binned into three groups for ease of presentation.

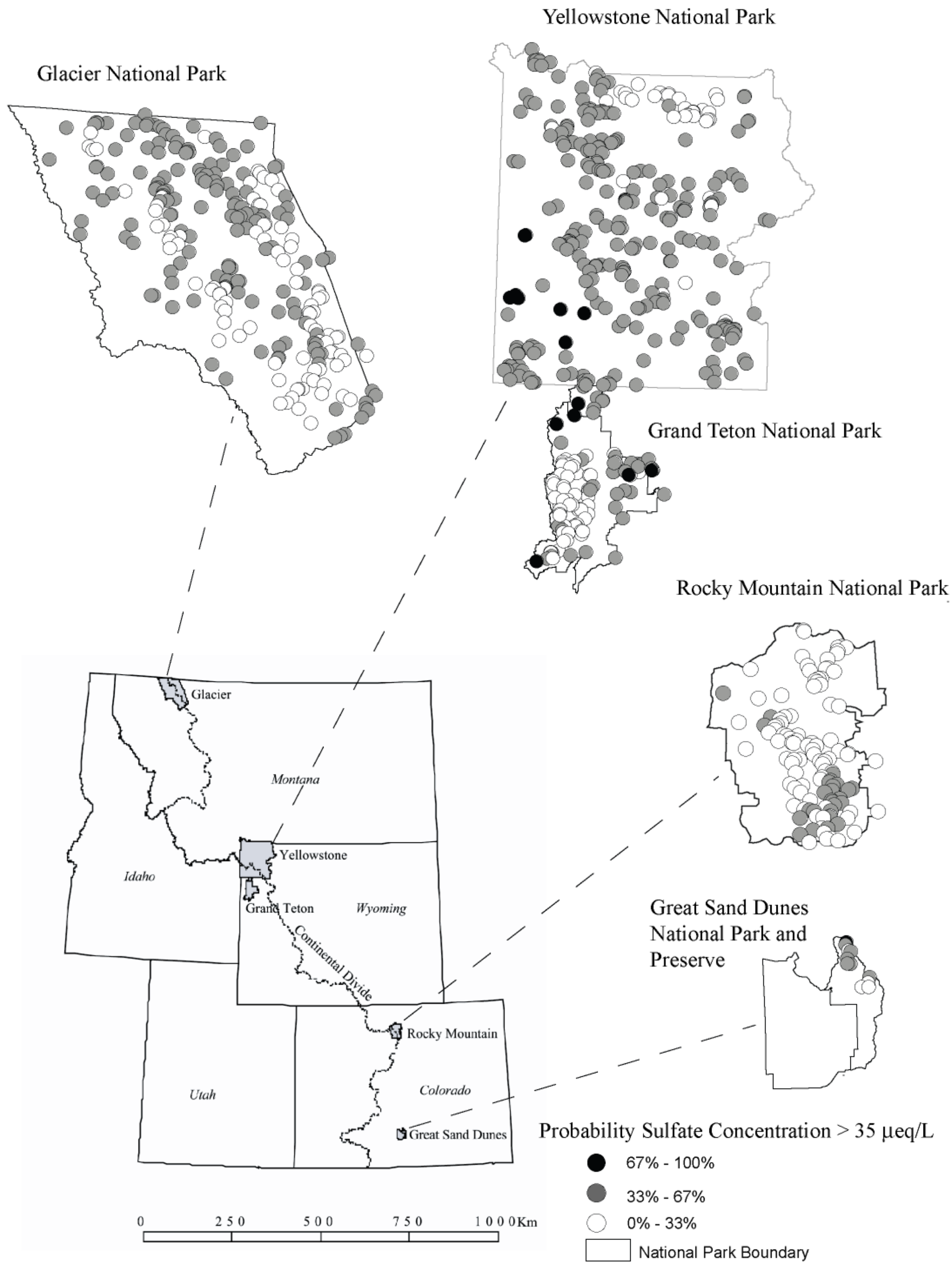

Figure 4-4. Probability of lake sulfate concentrations greater than 35 µeq/L in five Rocky Mountain National Parks. All probabilities ranging from 0-100% were evaluated; the data are binned into three groups for ease of presentation.

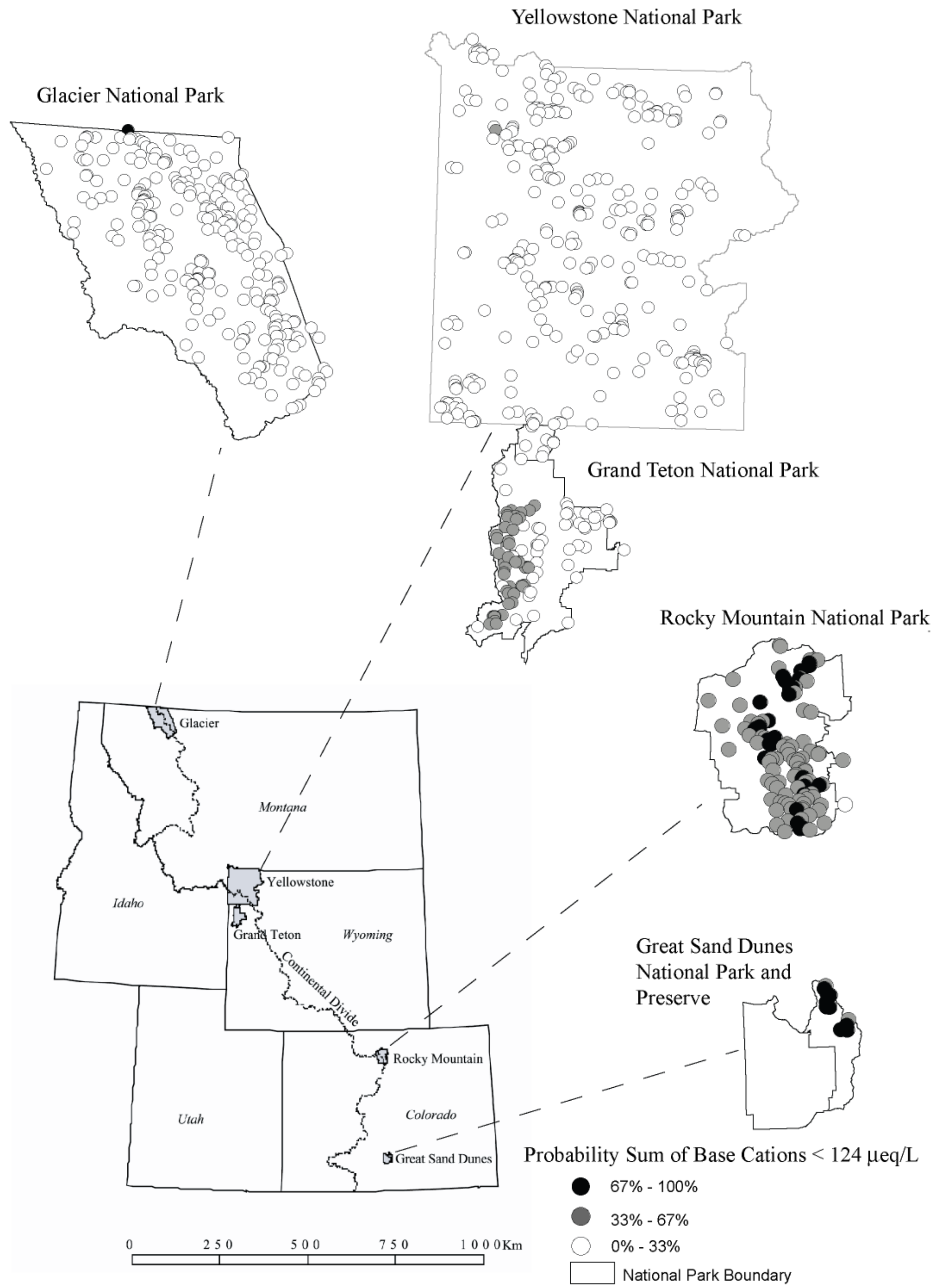

Figure 4-5. Probability of lake base cation concentrations less than 124 µeq/L in five Rocky Mountain National Parks. All probabilities ranging from 0-100% were evaluated; the data are binned into three groups for ease of presentation.

Lakes with a high probability for Bc concentrations less than 124 μeq/L (Figure 4-5) comprise approximately 33% of lakes in ROMO and GRSA (Figure 4-5) and are located at elevations greater than 3400 m with greater than 90% of the basin with low buffering capacity bedrock. Results indicate that less than 10% of lakes in YELL, GRSA, and GRTE had a high probability for lake SO_4 concentrations greater than 35 μeq/L (Figure 4-4), and few lakes in ROMO and GLAC. The lakes that had a high probability of having a SO_4 concentration greater than 35 μeq/L have less than 14% of the basin with bedrock composed of granite, gneiss, quartzite or schist and have SO_4 deposition greater than 5 kg/ha/year SO_4-S deposition.

Modeling results were validated with 25% of the data (n=36) for each constituent. The spatial distribution of NO_3, Bc, and SO_4 concentrations (Figure 4-6) are similar to the calibration data set, for example NO_3 concentrations are highest in ROMO and GRSA and lowest in GLAC, GRTE, YELL. Predicted probabilities for solute concentrations were compared with measured concentrations to evaluate predictive ability, and results of the model validation for each chemical constituent are presented in Table (4-5) and Figure (4-6). For the calibration data set (n=108), the r^2 value is 0.97 for lakes with a predicted probability greater than 5 μeq/L compared with actual percentage of lakes with NO_3 concentrations less than 5 μeq/L. For the validation data set (n=36), the r^2 value is 0.95 is very similar to the calibration value of 0.97 for NO_3. The model validation results for SO_4 (r^2 = 0.73) and Bc (r^2 = 0.56) are lower than for NO_3.

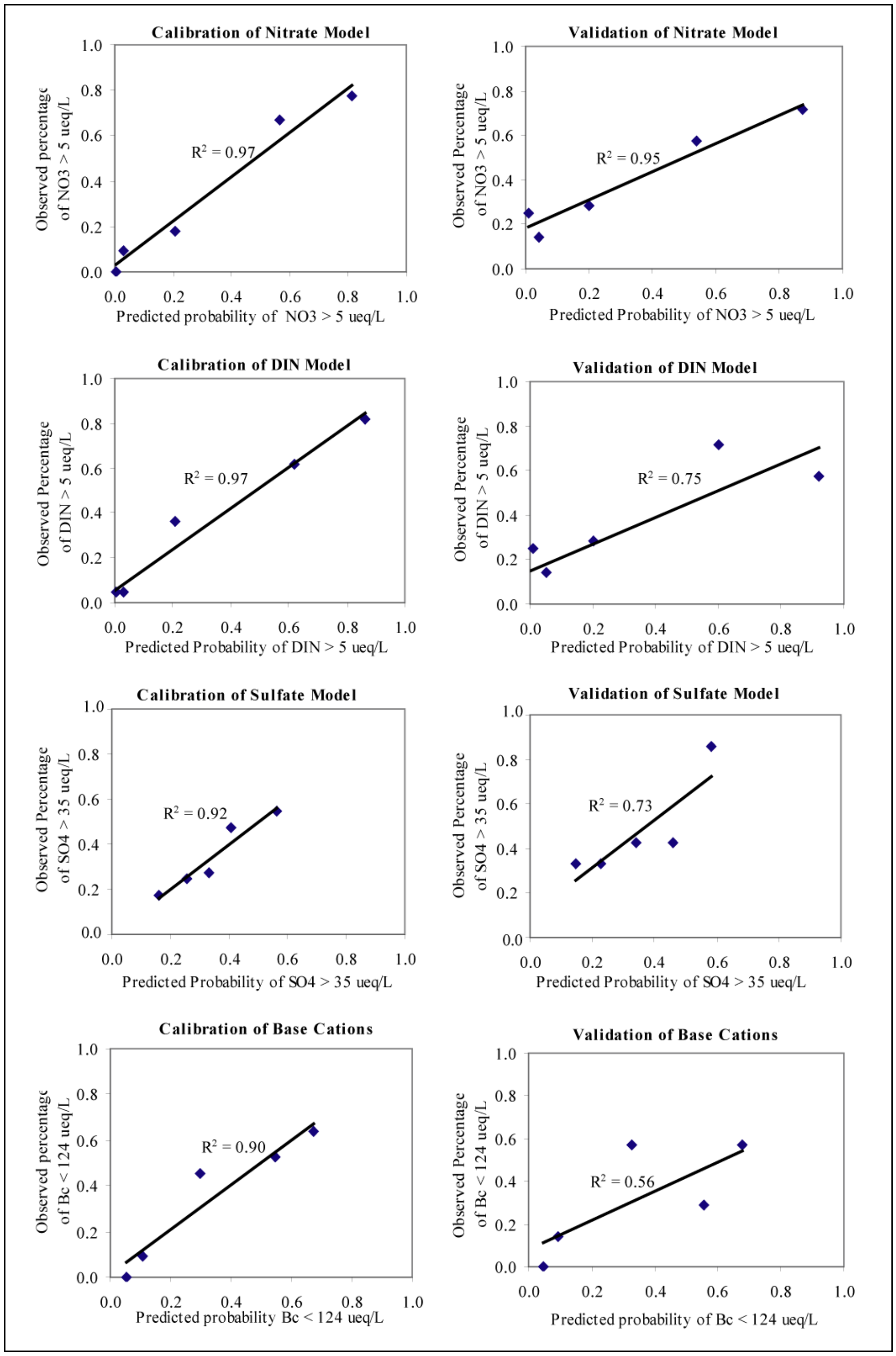

Figure 4-6. Percentage of observed concentration greater than threshold (DIN, NO_3, SO_4) and lower than threshold (Bc) versus the predicted probability of solute concentration greater than threshold (DIN, NO_3, SO_4) and lower than threshold (Bc) (calibration data, n=108, validation data, n=36).

DISCUSSION

The relation between characteristics of alpine/subalpine basins in the Rocky Mountains and NO_3, DIN, SO_4, and base cations in lakes is complex. The use of GIS and multivariate logistic regression to identify a subset of lakes that are most likely to have NO_3 concentration greater than 5 µeq/L yielded a high r^2 (r^2= 0.95) in model validation. However, the regional model for SO_4 was lower (r^2 = 0.75) and for base cations even lower (r^2 = 0.56) indicating that these models may not be as reliable. In general, the regional model had significant relations with a greater number of explanatory variables than the individual parks (Table 4-4), likely due to more statistical power for the region which had a greater number of lakes (n=144). The approach presented in this paper may be transferable to other remote and protected alpine/subalpine areas that are sensitive to acidic deposition, such as wilderness areas in national forests in the western U.S.

Spatial variation in solute concentrations and basin characteristics

Lakes across the region were identified based on a high predicted probability of having high NO_3 and SO_4 concentrations and low Bc concentrations. For the individual parks and for the Rocky Mountain region, the basin characteristics found to have an effect on predicted solute concentration probabilities, starting with the parameter that has the greatest influence are elevation, bedrock type, slope steepness, and estimates of atmospheric deposition. This is the first study to evaluate the effect of one basin characteristic over another in terms of solute concentration and these findings are discussed. These results are generally in agreement with previous work that relates basin characteristics with solute concentrations in the Rocky Mountains (Clow and Sueker, 2000; Sueker et al., 2001; Clow et al., 2002; Corbin, 2004).

For the Rocky Mountain region, increased elevation at the lake outlet was found to be significantly related (p-value < 0.01) to high NO_3 concentrations and low Bc concentrations in this study, and to low ANC concentrations (less than 100 μeq/L) in a companion study (Nanus et al., in review). ANC is a measure of the water's capacity to buffer acidic inputs and a measure of the concentration of solutes. As expected NO_3 was inversely related to ANC (spearman correlation, $p<0.01$), and lakes with low ANC typically had high NO_3 concentrations. An evaluation of the standardized coefficients, shows that elevation has the greatest influence in predicting the probability for high NO_3 (0.416) and low sum of base cations (0.364), over bedrock geology, slope, and NO_3 deposition for the region (Table 4-5). In the Rocky Mountains, low lake ANC concentrations and physical characteristics of high-elevation basins such as limited soils and biota, combined with the storage and release of contaminants in snowmelt runoff from deep snowpacks, make them particularly susceptible to acidic deposition (Williams et al., 1996, Baron and Campbell, 1997). This significance of elevation influencing solute chemistry in lakes is consistent with high lake elevation found to be a predictor of lower surface water ANC concentrations in two Colorado Wilderness Areas (Turk and Adams, 1983; Turk and Campbell, 1987), Yellowstone (Gibson et al., 1983; Nanus et al., 2005), Grand Teton National Park (Nanus et al., 2005), in wilderness areas of Nevada, Idaho, Utah, and Wyoming (Rutkowski et al., 2001) and in the Swiss Alps (Drever and Zorbrist, 1992). In this study, elevation was found to be positively related to NO_3 and inversely related to Bc not only for the region, but also within individual parks including ROMO, YELL, and for GRSA (NO_3 only). However, elevation was not found to be significantly related to solute concentrations in ROMO in previous work (Clow and Sueker, 2000). This difference may be because in the current study 50 lakes were included with a difference in elevation of approximately 1000 m between the high and low sites (Table 4-3), compared with only 9 basins with a difference of only 384 m in Clow and Sueker (2004). No relation was found in GRTE, similar to results of Corbin (2004).

Lake sensitivity to acidification from deposition of acidic solutes is a function of the contribution of geologic weathering products that act to neutralize acidic compounds. Geochemical processes, including mineral weathering and cation exchange, play an important role because they are the main source of base cations and ANC. For the Rocky Mountain region, bedrock geology with low buffering capacity (granite, gneiss, schist, quartzite) was significantly and positively related to high NO_3 concentrations and low base cation concentrations, and was inversely related to high SO_4. Significant relations were found between low ANC concentrations and GC1 (Nanus et al., in review). The standardized coefficents for NO_3 (0.356) and Bc (0.364) indicate that for the region, bedrock geology has the next greatest influence on solute chemistry after elevation. Results of bedrock geology for the regional model are consistent with findings of Clow and Sueker (2000) in ROMO and Corbin et al. (2004) in GRTE. In GLAC, significant relations were found between high base cation and high-buffering capacity bedrock types that included carbonates and calc-silicates, due to the potential for the production of base cations that balance acidic anions. SO_4

was inversely related to GC1 in the Rocky Mountain region, likely because granite and gneiss generally do not contain mappable sulfide bearing minerals. The relation between bedrock geology and solute concentrations may have been limited by the resolution of the geologic maps used for the quantification of basin characteristics, for example that may not include the presence of unmapped trace quantities of a sulfide mineral, such as pyrite, within the lake basins which could affect the relations with SO_4.

Watersheds with steep slopes, which typically have poorly developed soils and limited vegetative cover, were positively related to lake NO_3 concentrations in ROMO and in the region ($p< 0.05$). Previous studies also found significant relations between fraction of basins with slopes greater than 30 degrees and NO_3 concentrations in alpine/subalpine basins in ROMO (Clow and Sueker, 2000; Sueker et al., 2001). Corbin (2004) did not find a significant relation between median slope and NO_3 concentrations and neither did this study in GRTE. To further examine the role of vegetative cover in ROMO, the fraction of the basin that is unvegetated was positively related to lake NO_3 and negatively related to Bc, and the fraction of basin that was forested was inversely related to lake NO_3 concentrations and positively related to Bc, indicating that these two environments have a significant and opposite effect on lake solute concentrations. When atmospheric DIN deposition exceeds the total combined plant and microbial demand, it can lead to N saturation and increased rates of N leaching from soils to aquatic ecosystems (Aber et al., 1989). Lakes with high probability (> 67%) for NO_3 concentrations greater than 5 μeq/L were primarily located in ROMO and GRSA, which is in keeping with previous work that has found

N saturation is currently occurring in the Front Range of Colorado (Williams et al., 1996; Baron 2006).

Atmospheric deposition of DIN, SO_4, and hydrogen ion, has the potential to alter the chemistry of aquatic ecosystems, through nitrogen saturation and episodic acidification, thus increasing the sensitivity of lakes to future changes related to atmospheric deposition. This is the first study to include deposition estimates in a predictive model, and to relate lake solute concentrations with deposition estimates across the Rocky Mountain region. Average annual deposition maps show regions of high DIN and SO_4 deposition within national parks of the Rockies, particularly parks located in Colorado including ROMO and GRSA (Nanus et al., 2003). NO_3, DIN, and SO_4 deposition estimates were calculated for each basin and were included in the statistical analysis. Previous work found that deposition was not significantly related to ANC concentrations, which may be due to the fact that acidic deposition in the Western US is not sufficient to cause chronic acidification (Nanus et al., in review).

In this study, we show that lake NO_3 concentrations are positively ($p < 0.01$) related to NO_3 deposition, with the highest lake NO_3 concentrations and NO_3 deposition estimates at high elevations (> 2700 meters) in ROMO and GRSA. These findings are in agreement with Clow et al. (2002), which showed a similarity between regional patterns in NO_3 and SO_4 lake concentrations and concentrations in precipitation, with NO_3 concentrations highest in ROMO compared with seven national parks in the western US. NO_3, DIN, and SO_4 deposition in the Rockies increases with increasing elevation (> 2500 meters) compared with lower elevations (< 2500 meters), likely due in part to orographically enhanced precipitation amounts at high elevations (Nanus et al., 2003; Williams and Tonnessen, 2000). Our results showing that elevation has the greatest influence on predicted lake NO_3 concentrations at the regional scale, and that elevated lake NO_3 concentrations and NO_3 deposition are found in lakes located at the highest elevations of the study area may be partially explained by the landscape continuum hypothesis developed by Seastedt et al. (2004). Seastedt et al. (2004) suggest that high-elevation areas serve as source areas that export NO_3 to lower-elevation systems, and that high-elevation lakes are particularly vulnerable to atmospheric DIN inputs that are amplified by transport processes in portions of the basin.

The correlation between solute concentrations and atmospheric deposition has also been shown at a regional scale in Europe (Lepori et al., 2003) and at a global scale in the Northern hemisphere (Bergstrom, and Jansson, 2006). In the Swiss Alps, NO_3 and SO_4 concentrations in surface water correlated significantly (and positively) with N and S concentrations in wet deposition, with a stronger association for NO_3 in summer compared with winter (Lepori et al., 2003). Lepori et al. (2003) suggest that deposition may indirectly stimulate mineralization and nitrification, increasing the rate of NO_3 production, and though without additional work, quantification of the amount of NO_3 that comes directly from deposition is not possible.

Nanus et al. (in review) evaluated regional trends in nitrate isotopes for source identification in the five parks and found that NO_3 concentrations and $\delta^{15}N$ (NO_3) values are heaviest in lakes and precipitation from the Southern Rockies and at higher elevations, compared to lower elevations and the Northern Rockies. The correspondence of higher NO_3 concentrations and higher $\delta^{15}N$ (NO_3) values in precipitation with higher NO_3 and higher $\delta^{15}N$ (NO_3) values in lake waters, suggests that atmospheric deposition of NO_3 may affect the amount of NO_3 in lakes through either direct (wet deposition) or indirect processes (enhanced nitrification) (Nanus et al., in review). This correlation warrants additional investigation.

Limitations of Approach

There are inherent limitations with this type of a regional-scale predictive modeling approach. Scale related issues are relevant to this study because available information on characteristics distributed over a watershed may vary widely in their spatial resolution. For example, for a specific basin characteristic a process may be present at one level of GIS resolution but not at another, when evaluated within a 1 km^2 plot, a 10 km^2 catchment, or an entire National Park at 100 km^2. At the 1 km^2 watershed scale, local physical features and processes may dominate, while at the level of 100 km^2 many features may not be resolved. For example, the presence of unmapped carbonate rocks or sulfide-bearing minerals within the lake basins could affect the model results. The type of soil and the spatial extent and depth of soils together with initial conditions affects alkalinity and may affect solute concentrations. Thus, omitting soils in the regional model, because of a lack of detailed data that is consistent across parks, may have affected the results.

Environmental variables other than those included in this study also may contribute to variability in some lake solute concentrations and can be difficult to quantify; these variables could include recreational use, fires particularly in YELL and GLAC, and geothermal activity in YELL. This analysis predicts the probability of solute concentrations in relation to a set threshold based on the distribution of data. However, this analysis only includes chemical data sampled during baseflow. To capture seasonality, further seasonal sampling and application of a similar approach is recommended for ROMO and GRSA where lakes have the highest NO_3 concentrations, and low Bc.

CONCLUSIONS

The significant relations identified in this analysis between solute concentrations and basin characteristics, indicate that the GIS and multivariate logistic regression approach across a regional scale can be used to help resource managers identify lakes for future monitoring. A benefit of this type of an approach is that relations can be developed using existing water quality data and with topographic, geologic, and vegetation data in GIS. Final multivariate logistic regression models were selected that predict the probability of having a solute concentration for NO_3 above 5 μeq/L, for SO_4 above 35 μeq/L, and for sum of base cations below 124 μeq/L.

The multivariate logistic regression model for nitrate had the best statistical outcome, with the status of NO_3 in 95% of lakes correctly predicted in the validation data. Lakes with a high probability (defined here as greater than 67% predicted probability) of having NO_3 concentrations greater than 5 microequivalents per liter (μeq/L) are located in basins with elevations greater than 2,700 meters, maximum basin slope greater than 43 degrees, greater than 66% of the basin with low buffering capacity bedrock, and with greater than 1 kg/ha/yr NO_3-N deposition.

Elevation was found to have the greatest influence on lake solute concentrations followed by bedrock type, basin slope, and nitrate deposition. At high elevations, the significant correlation ($p < 0.01$) between lake nitrate concentrations and atmospheric nitrate deposition suggests that some lakes are showing a response to nitrate deposition, through either direct (wet deposition) or indirect (enhanced nitrification) processes. ROMO and GRSA have the most lakes with predicted NO_3 concentrations greater than 5 μeq/L and low base cation concentrations, and are coincident with areas that have increasing rates of DIN deposition. These findings may be used to improve long-term monitoring designs for high-elevation watersheds in the Rockies and the modeling approach may be transferable to other remote mountain areas of the US and the world.

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CHAPTER 5

RECOMMENDATIONS FOR FUTURE MONITORING

The GIS and logistic regression modeling approach described in this dissertation that were used to make regional-scale predictions of lake's that may be sensitive to atmospheric deposition of pollutants. This approach can be used as a tool to help resource managers improve long-term monitoring designs for both lakes chemistry and precipitation chemistry in protected areas within the Rocky Mountains. These recommendations are particularly relevant to the five National Parks in the Rockies where the data were gathered and the lake sensitivity models developed, but may also be useful in other parts of the region.

- Monitoring of sensitive lakes in the two Colorado parks, GRSA and ROMO, is especially important since: over one third of their lake populations (> 1 ha) have a model predicted high probability (> 66%) for elevated nitrate (NO_3) concentrations and low acid-neutralizing capacity (ANC) and base cation concentrations.
- These high elevations systems are receiving increased rates of inorganic N deposition (NADP, 2007).

The ultimate endpoints of concern for resource managers include (1) water quality (many of these lakes are Outstanding Natural Resource Waters), (2) aquatic species, including endangered species of fish and amphibians, and (3) potential for maintenance or restoration of ecological integrity in these dilute systems.

At the regional scale, lake chemistry models were validated as part of this project. Models for ANC concentrations and NO_3 concentrations in lakewater had the best statistical outcome with over 93% of lakes in the validation data correctly predicted. Based on the findings of this dissertation, the following recommendations are likely to help resource managers understand lake sensitivity to atmospheric deposition:

- Collect lake samples each season for a minimum of 5 years at one to two high-elevation lakes per park to evaluate seasonal variation in lake chemistry and the potential for episodic acidification and eutrophication of lake water in the future.
    - Therefore, selection criteria for lakes for long term monitoring include: location at high elevation (minimum basin elevation of 2,700 meters).
    - reasonable access during all seasons of the year
    - lake area should be in the range of 1-4 hectares
    - basin geology should include more than 66 percent of the bedrock with low buffering capacity (granite, gneiss, quartzite, schist),
    - basins with some slopes greater than 43 degrees
- Most at risk are basins that currently receive inorganic nitrogen in atmospheric deposition greater than 1 kg/ha/yr N.
- In the general vicinity of the indicator lakes, precipitation quantity and quality should be sampled at climate stations, with emphasis on

weekly sampling of precipitation chemistry. Collection of additional data on the amount and chemistry of seasonal snow near to the indicator lakes should also be considered.

- Analysis of lakewater and precipitation samples should be done in a laboratory that specializes in the analysis of dilute waters for pH, acid neutralizing capacity, conductance, and major ions. Full ion chemistry, along with appropriate QA/QC samples, should be included in the long term monitoring protocol.
- Water samples should be archived at temperatures of -80^{o}C.
- A periodic review and analysis of the long term data sets should be conducted by subject matter experts in research agencies or universities.

To increase the utility of the monitoring data and to evaluate long-term variability in alpine and subalpine lake chemistry in the five Rocky Mountain parks the following are recommended:

- Annual water-quality sampling of three to four high-elevation (>2700 meters) lakes for a minimum of 10 years is likely to provide the necessary data to evaluate long-term variability in lake sensitivity. Important considerations for selection of lakes to be sampled include model predictions of high probability (> 66%) for ANC concentrations less than 100 μeq/L, NO_3 concentrations predicted greater than 5 μeq/L and low base cation concentrations (appendix 1), and a representative spatial distribution within each park.

- Elevation was found to have the greatest influence on lake chemistry across the region. Therefore, establishing long term water-quality monitoring at an additional two to three lakes in each park at lower elevations would provide data to evaluate changes in lake chemistry over time with changing elevation. Long-term monitoring of lakes that were identified as having moderate sensitivity (probability 33% - 66%) for elevated NO_3 concentrations (> 5 μeq/L) and low ANC concentrations (< 100 μeq/L) would provide the data needed to detect gradual temporal trends and early warning of changes in the water-quality of these lakes.
- Long-term monitoring of selected lakes for a minimum of 10 years would aid in evaluating the influence of climate change and natural variability on lake sensitivity over time.

If refinement of the lake sensitivity models in this dissertation is needed, the following are recommended:

- Geostatistics can be used to evaluate the influence of landscape attributes and their position in the watershed relative to the proximity of the lake. Further study of lake sensitivity to atmospheric deposition using geostatistics may improve this analysis.

- Understanding the relation between nitrogen deposition and NO_3 concentrations in lakes is important to our predicting future lake chemistry and sustainability of aquatic biota and ecosystems. Data collected during this study of Rocky Mountain lakes show a significant correlation between lake NO_3 concentrations and nitrogen deposition, further suggesting that some lakes may already be showing a response to inorganic nitrogen (both nitrate and ammonium) in atmospheric deposition. Regional analysis of nitrogen and oxygen isotopes leads us to the hypothesis that nitrogen deposition may affect the amount of NO_3 in lakes through indirect processes such as enhanced nitrification. This hypothesis should be investigated through more detailed watershed experiments and mass balance analysis for nitrogen species.
- Further investigation into sources and sinks of atmospheric NO_3 and NH4 is recommended, using a more detailed isotopic analysis. Seasonal sampling and analysis of isotopes of NO_3 and NH4 at one to two high-elevation lakes in each park for a minimum of five years is recommended. These data can be used to evaluate the contribution of atmospheric nitrate and ammonium to watershed and lakewater nitrogen concentrations.

Of the five national parks studied the ones with the most risk factors for water quality changes due to atmospheric deposition are Grand Teton National Park and Rocky Mountain National Park. As such, long term deposition monitoring and lake monitoring at Rocky Mountain NP should continue, and perhaps, be expanded to determine the effects of nitrogen and sulfur deposition on lakes and their biota. Grand Teton NP has selected a few high elevation lakes for long term, vital signs monitoring. However, there is currently no consistent monitoring of wet deposition in this park. Both wet and dry deposition monitoring are recommended for Grand Teton NP. Great Sand Dunes National Park and Preserve shares many of the risk factors with other Rocky Mountain protected areas for water quality changes due to atmospheric deposition. However, data sets on deposition chemistry and lakewater chemistry are limited in this park. Expanded surveys and long term monitoring of both parameters are recommended for Great Sand Dunes.

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Appendix A. Lakes greater than 1 hectare in Glacier National Park, Grand Teton National Park, Great Sand Dunes National Park and Preserve, Rocky Mountain National Park, and Yellowstone National Park, including lake identification number, lake name, elevation of lake outlet, and associated predicted probabilities at acid-neutralizing capacity concentration < 100 μeq/L, nitrate concentration > 5 μeq/L, sulfate concentration > 35 μeq/L, and sum of base cations < 124 μeq/L.

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Glacier	1	Cameron	1660	1%	0%	66%	3%
Glacier	2	Waterton	1279	1%	1%	49%	2%
Glacier	3	small unnamed	1392	2%	0%	59%	2%
Glacier	4	Upper Kintla	1332	1%	0%	64%	2%
Glacier	5	Wurdeman	1605	1%	1%	51%	3%
Glacier	6	Kintla	1222	1%	0%	55%	2%
Glacier	7	Lake Nooney	1676	2%	2%	61%	3%
Glacier	8	small unnamed	1941	1%	3%	59%	4%
Glacier	9	small unnamed	1330	1%	0%	57%	2%
Glacier	10	small unnamed	1331	0%	0%	61%	2%
Glacier	11	small unnamed	1678	1%	1%	59%	3%
Glacier	12	large unnamed	1299	0%	0%	36%	4%
Glacier	13	large unnamed	2018	1%	1%	66%	5%
Glacier	14	Carcajou	1717	2%	1%	58%	4%
Glacier	15	Wahseeja	2085	5%	1%	54%	6%
Glacier	16	large unnamed	1852	9%	28%	19%	12%
Glacier	17	Pocket	2016	6%	2%	57%	6%
Glacier	18	Goat Haunt	1823	10%	19%	29%	10%
Glacier	19	Shaheeya	2155	1%	0%	57%	7%
Glacier	20	Miche Wabun	1817	3%	2%	62%	4%
Glacier	21	Lake Janet	1500	1%	1%	65%	2%
Glacier	22	Lake Francis	1602	2%	1%	63%	3%
Glacier	23	large unnamed	1448	2%	0%	60%	2%
Glacier	24	Kaina	2093	8%	2%	66%	5%
Glacier	25	large unnamed	1646	1%	0%	64%	3%
Glacier	26	large unnamed	2115	7%	4%	66%	6%
Glacier	27	small unnamed	1332	1%	0%	59%	2%
Glacier	28	small unnamed	2177	28%	21%	29%	14%
Glacier	29	Numa	1980	9%	6%	45%	7%
Glacier	30	Long Bow	2099	21%	21%	28%	15%
Glacier	31	Kootenia Lakes	1332	1%	0%	57%	2%
Glacier	32	large unnamed	2276	31%	30%	29%	18%
Glacier	35	large unnamed	1335	1%	0%	64%	2%
Glacier	36	Cosley	1476	2%	1%	52%	3%
Glacier	37	large unnamed	2029	7%	3%	64%	5%
Glacier	38	small unnamed	1339	1%	0%	60%	2%
Glacier	39	Bowman	1228	1%	1%	46%	2%
Glacier	40	Glenns	1482	2%	1%	47%	3%
Glacier	41	small unnamed	1180	1%	0%	64%	1%
Glacier	42	Whitecrow	1874	5%	1%	65%	4%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 µeq/L	Probability Nitrate >5 µeq/L	Probability Sulfate > 35 µeq/L	Probability Sum of Base Cations < 124 µeq/L
Glacier	43	large unnamed	1518	3%	2%	46%	4%
Glacier	44	Slide	1518	3%	2%	39%	4%
Glacier	45	Akokala	1443	5%	2%	34%	5%
Glacier	46	Otatso	2125	24%	34%	23%	17%
Glacier	47	Elizabeth	1491	4%	4%	36%	5%
Glacier	48	Stoney Indian	1928	5%	2%	60%	4%
Glacier	49	small unnamed	1443	2%	0%	64%	2%
Glacier	50	Bench	2067	6%	2%	56%	7%
Glacier	51	large unnamed	2131	14%	23%	23%	17%
Glacier	52	Mokowanis	1518	2%	2%	46%	3%
Glacier	53	Cerulean	1420	3%	1%	46%	3%
Glacier	54	small unnamed	1964	3%	1%	64%	4%
Glacier	55	Atsina	1757	2%	2%	61%	3%
Glacier	56	small unnamed	2081	25%	18%	21%	16%
Glacier	57	Redhorn	1862	3%	0%	64%	4%
Glacier	58	large unnamed	2214	26%	11%	25%	18%
Glacier	59	small unnamed	2138	6%	2%	64%	6%
Glacier	60	Margaret	1699	3%	2%	57%	4%
Glacier	61	Sue	2178	5%	11%	52%	8%
Glacier	62	small unnamed	1951	11%	14%	18%	16%
Glacier	63	large unnamed	1684	3%	5%	45%	4%
Glacier	64	small unnamed	2175	13%	0%	64%	6%
Glacier	65	Ipasha	1725	3%	2%	57%	3%
Glacier	66	Quartz	1346	2%	2%	33%	4%
Glacier	67	Poia	1763	9%	8%	23%	9%
Glacier	68	small unnamed	2182	4%	0%	60%	6%
Glacier	69	small unnamed	2208	6%	2%	61%	7%
Glacier	70	large unnamed	2248	7%	4%	60%	7%
Glacier	71	Gyrfalcon	2213	6%	2%	60%	7%
Glacier	72	Ptarmigan	2019	16%	4%	35%	9%
Glacier	73	small unnamed	1536	2%	0%	62%	2%
Glacier	74	Helen	1550	3%	2%	42%	4%
Glacier	75	Nahsukin	1649	3%	0%	59%	3%
Glacier	76	Swiftcurrent	1850	2%	0%	60%	4%
Glacier	77	large unnamed	1225	1%	0%	41%	2%
Glacier	78	Middle Quartz	1340	1%	0%	55%	2%
Glacier	79	large unnamed	1859	5%	0%	64%	4%
Glacier	80	Kennedy	2067	12%	10%	27%	13%
Glacier	81	small unnamed	1856	4%	0%	60%	4%
Glacier	82	large unnamed	1798	12%	2%	23%	9%
Glacier	83	Lower Quartz	1277	1%	0%	59%	2%
Glacier	84	Sherburne	1455	2%	1%	40%	4%
Glacier	85	large unnamed	1934	3%	0%	60%	4%
Glacier	86	large unnamed	1994	6%	0%	60%	5%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 µeq/L	Probability Nitrate >5 µeq/L	Probability Sulfate > 35 µeq/L	Probability Sum of Base Cations < 124 µeq/L
Glacier	87	large unnamed	1271	2%	0%	42%	3%
Glacier	88	large unnamed	1851	3%	5%	49%	5%
Glacier	89	Natahki	2004	19%	28%	19%	16%
Glacier	90	Iceberg	1858	3%	5%	50%	5%
Glacier	91	small unnamed	1963	18%	26%	19%	16%
Glacier	92	large unnamed	2001	2%	2%	61%	5%
Glacier	93	large unnamed	1202	3%	1%	29%	3%
Glacier	94	small unnamed	1893	3%	0%	60%	4%
Glacier	95	Grace	1202	2%	1%	30%	3%
Glacier	96	small unnamed	1468	1%	0%	58%	2%
Glacier	97	Swiftcurrent	1487	3%	2%	40%	4%
Glacier	98	Fishercap	1506	2%	1%	49%	3%
Glacier	99	Redrock	1548	2%	1%	52%	3%
Glacier	100	Governor Pond	1486	1%	0%	62%	2%
Glacier	101	large unnamed	1573	2%	1%	59%	3%
Glacier	102	Windmaker	1622	2%	1%	64%	3%
Glacier	103	small unnamed	1688	2%	1%	61%	3%
Glacier	104	small unnamed	2083	8%	0%	61%	5%
Glacier	105	large unnamed	2079	6%	0%	63%	5%
Glacier	106	large unnamed	1803	1%	1%	62%	4%
Glacier	107	Bullhead	1579	2%	1%	60%	3%
Glacier	108	Stump	1487	3%	2%	42%	4%
Glacier	109	Lake Josephine	1487	3%	2%	40%	4%
Glacier	110	Logging	1161	1%	0%	38%	2%
Glacier	111	Falling Leaf	2019	20%	20%	18%	17%
Glacier	112	Snow Moon	2030	20%	22%	16%	17%
Glacier	113	large unnamed	1163	1%	0%	63%	1%
Glacier	114	small unnamed	2272	45%	26%	15%	25%
Glacier	115	large unnamed	2164	31%	12%	17%	21%
Glacier	116	Grinnell	1505	1%	1%	50%	3%
Glacier	117	small unnamed	2125	24%	10%	16%	20%
Glacier	118	Upper Grinnell	1969	3%	3%	62%	4%
Glacier	119	small unnamed	2068	34%	29%	16%	18%
Glacier	120	Evangeline	1599	3%	3%	38%	5%
Glacier	121	Camas	1547	2%	0%	57%	3%
Glacier	122	Ruger	1769	6%	13%	21%	11%
Glacier	123	Cracker	1801	8%	13%	32%	7%
Glacier	124	Mud	1064	1%	0%	62%	1%
Glacier	125	large unnamed	2020	9%	20%	19%	16%
Glacier	126	St. Mary	1367	2%	1%	45%	3%
Glacier	127	Dutch Lakes 2	2094	23%	22%	24%	14%
Glacier	128	large unnamed	1410	1%	0%	61%	2%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 µeq/L	Probability Nitrate >5 µeq/L	Probability Sulfate > 35 µeq/L	Probability Sum of Base Cations < 124 µeq/L
Glacier	129	Dutch Lakes 3	2063	32%	7%	15%	19%
Glacier	130	Dutch Lakes 1	1737	11%	3%	26%	8%
Glacier	131	Lilly	1275	1%	0%	62%	2%
Glacier	132	large unnamed	2059	31%	34%	16%	19%
Glacier	133	large unnamed	1951	10%	7%	40%	8%
Glacier	134	large unnamed	2216	12%	3%	61%	7%
Glacier	135	small unnamed	2078	19%	5%	16%	19%
Glacier	136	Otokomi	1976	13%	17%	26%	12%
Glacier	137	Arrow	1241	2%	1%	45%	2%
Glacier	138	small unnamed	1486	0%	0%	63%	2%
Glacier	139	large unnamed	1455	1%	0%	62%	2%
Glacier	140	Goat	1968	11%	21%	20%	14%
Glacier	141	Trout	1190	2%	1%	35%	3%
Glacier	142	Hidden	1943	4%	4%	54%	5%
Glacier	143	small unnamed	1943	1%	0%	62%	4%
Glacier	144	small unnamed	1983	0%	0%	63%	5%
Glacier	145	large unnamed	1367	3%	2%	17%	7%
Glacier	146	Rogers	1156	2%	1%	37%	3%
Glacier	147	Avalanche	1190	1%	1%	40%	2%
Glacier	148	Red Eagle	1439	2%	1%	46%	3%
Glacier	149	large unnamed	2051	14%	17%	24%	14%
Glacier	150	Mary Baker	2012	8%	1%	62%	5%
Glacier	151	Twin Lakes 2	1786	3%	1%	62%	3%
Glacier	152	Twin Lakes 1	1789	3%	2%	63%	3%
Glacier	153	McDonald	956	0%	0%	51%	1%
Glacier	154	large unnamed	1443	1%	0%	63%	2%
Glacier	155	Johns	1010	2%	0%	27%	3%
Glacier	156	small unnamed	2247	2%	0%	63%	7%
Glacier	157	small unnamed	2240	12%	3%	61%	7%
Glacier	158	large unnamed	2248	6%	12%	32%	15%
Glacier	159	Snyder Lakes 1	1699	14%	24%	18%	11%
Glacier	160	small unnamed	2138	37%	20%	18%	20%
Glacier	161	large unnamed	1882	3%	1%	61%	4%
Glacier	162	Snyder Lakes 2	1594	11%	17%	19%	9%
Glacier	163	large unnamed	2140	17%	18%	17%	20%
Glacier	164	Gunsight	1620	3%	1%	51%	4%
Glacier	165	Akaiyan	2344	52%	24%	17%	26%
Glacier	166	Feather Woman	2298	41%	18%	19%	24%
Glacier	167	large unnamed	2074	12%	8%	28%	13%
Glacier	168	large unnamed	2110	16%	10%	25%	15%
Glacier	169	Howe 1	1252	0%	0%	61%	1%
Glacier	170	Howe 2	1252	1%	0%	63%	1%
Glacier	171	Ellen Wilson	1807	10%	18%	22%	11%
Glacier	172	small unnamed	1953	1%	0%	63%	4%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/I	Probability Nitrate >5 μeq/L	Probability Sulfate > 35μeq/L	Probability Sum of Base Cations < 124 μeq/L
Glacier	172	small unnamed	1953	1%	0%	63%	4%
Glacier	173	Fish	1263	0%	0%	63%	1%
Glacier	174	large unnamed	2036	3%	3%	60%	5%
Glacier	175	Medicine Owl	2075	13%	26%	17%	19%
Glacier	176	Lincoln	1402	5%	5%	22%	6%
Glacier	177	small unnamed	1840	11%	15%	21%	12%
Glacier	178	large unnamed	2154	25%	22%	24%	17%
Glacier	179	small unnamed	2126	38%	19%	17%	21%
Glacier	180	small unnamed	2101	4%	1%	63%	5%
Glacier	181	large unnamed	1929	5%	2%	50%	6%
Glacier	182	large unnamed	2135	13%	20%	27%	15%
Glacier	183	Medicine Grizzly	1696	4%	4%	39%	6%
Glacier	184	small unnamed	2027	33%	4%	17%	18%
Glacier	185	large unnamed	1904	7%	7%	30%	10%
Glacier	186	large unnamed	2083	13%	15%	32%	12%
Glacier	187	large unnamed	2035	13%	13%	27%	13%
Glacier	188	Running Crane	2240	14%	20%	19%	22%
Glacier	189	Morning Star	1757	6%	4%	35%	7%
Glacier	190	Lonely Lakes 1	2174	15%	33%	17%	22%
Glacier	191	Lonely Lakes 2	2207	15%	35%	17%	23%
Glacier	192	small unnamed	1985	30%	12%	17%	17%
Glacier	193	Harrison	1126	2%	1%	33%	3%
Glacier	194	Katoya	1941	3%	4%	24%	12%
Glacier	195	large unnamed	2138	26%	32%	17%	20%
Glacier	196	Lake Seven Winds	2124	4%	0%	59%	6%
Glacier	197	Pitamakan	2074	3%	3%	51%	7%
Glacier	198	small unnamed	1761	5%	0%	63%	3%
Glacier	199	large unnamed	1675	2%	0%	53%	4%
Glacier	200	large unnamed	1710	1%	1%	50%	4%
Glacier	201	Oldman	2026	5%	3%	51%	7%
Glacier	202	large unnamed	1882	9%	21%	20%	13%
Glacier	203	small unnamed	2056	23%	36%	17%	19%
Glacier	204	Lower Medicine	1488	4%	2%	29%	5%
Glacier	205	small unnamed	2067	17%	3%	40%	9%
Glacier	206	Boy	1929	5%	4%	41%	8%
Glacier	207	Young Man	2105	3%	4%	45%	9%
Glacier	208	Sky	2130	26%	27%	17%	21%
Glacier	209	small unnamed	1788	7%	0%	54%	4%
Glacier	210	Nyack Lakes 2	1495	6%	4%	26%	6%
Glacier	211	Beaver Woman	1788	7%	2%	45%	6%
Glacier	212	Pray	1574	6%	4%	25%	7%
Glacier	213	Two Medicine	1574	6%	4%	25%	7%
Glacier	214	No Name	1806	7%	5%	37%	7%
Glacier	215	Buffalo Woman	1856	10%	13%	19%	13%
Glacier	216	Halfmoon	1212	1%	0%	63%	1%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/I	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Glacier	217	small unnamed	1646	8%	11%	18%	10%
Glacier	218	Upper Medicine	1646	7%	12%	18%	10%
Glacier	219	large unnamed	1978	29%	15%	17%	17%
Glacier	220	Aurice	2233	32%	31%	17%	23%
Glacier	221	large unnamed	2089	26%	28%	17%	20%
Glacier	222	large unnamed	1242	1%	0%	63%	1%
Glacier	223	Cobalt	2003	7%	6%	24%	13%
Glacier	224	Lake Isabel	1742	7%	14%	23%	10%
Glacier	225	large unnamed	1623	1%	0%	63%	3%
Glacier	226	large unnamed	1826	8%	25%	18%	13%
Glacier	227	small unnamed	1826	6%	23%	19%	12%
Glacier	228	large unnamed	1760	9%	16%	21%	11%
Glacier	229	small unnamed	1558	2%	0%	63%	2%
Glacier	230	Jackstraw	1870	23%	24%	17%	15%
Glacier	231	Striped Elk	1918	11%	19%	21%	13%
Glacier	232	Lena	1953	19%	15%	17%	16%
Glacier	233	small unnamed	1588	1%	0%	63%	2%
Glacier	234	large unnamed	2299	38%	14%	17%	25%
Glacier	235	small unnamed	2151	41%	34%	17%	21%
Glacier	236	Green	1577	2%	0%	64%	2%
Glacier	237	small unnamed	1538	3%	0%	63%	2%
Glacier	238	Ole	1692	4%	0%	63%	3%
Glacier	239	small unnamed	2001	25%	19%	17%	17%
Glacier	240	large unnamed	1582	1%	0%	63%	2%
Glacier	241	large unnamed	1746	9%	17%	17%	12%
Glacier	242	small unnamed	1584	3%	0%	57%	3%
Glacier	243	Three Bears	1610	2%	0%	59%	3%
Glacier	244	small unnamed	1624	3%	0%	64%	3%
Grand Teton	1	South Boundary	2248	13%	1%	17%	24%
Grand Teton	2	Hechtman Lake	2394	17%	0%	71%	8%
Grand Teton	3	Two Ocean Lake	2102	7%	0%	33%	6%
Grand Teton	4	Cygnet Pond	2084	8%	0%	38%	5%
Grand Teton	5	Talus Lake	2947	66%	22%	24%	45%
Grand Teton	6	Swan Lake	2062	7%	0%	37%	5%
Grand Teton	7	Emma Matilda	2095	3%	0%	24%	8%
Grand Teton	8	Heron Pond	2065	6%	0%	36%	5%
Grand Teton	9	Dudley Lake	2512	25%	51%	18%	32%
Grand Teton	10	Christian Pond	2084	7%	0%	22%	8%
Grand Teton	11	Trapper Lake	2107	19%	13%	32%	15%
Grand Teton	12	Cirque Lake	2929	58%	56%	19%	48%
Grand Teton	13	Bearpaw Lake	2088	19%	11%	30%	15%
Grand Teton	14	Cow Lake	2083	7%	0%	36%	5%
Grand Teton	15	Halfmoon Lake	2064	2%	0%	35%	5%
Grand Teton	16	Leigh Lake	2094	12%	14%	23%	13%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Grand Teton	17	Mink Lake	2715	19%	32%	29%	31%
Grand Teton	18	Grizzly Bear	2810	59%	51%	18%	44%
Grand Teton	19	String Lake	2094	12%	12%	23%	13%
Grand Teton	20	Lake Solitude	2755	61%	38%	19%	40%
Grand Teton	21	Holly Lake	2868	78%	23%	18%	46%
Grand Teton	22	Mica Lake	2913	72%	62%	18%	48%
Grand Teton	23	Jenny Lake	2067	12%	12%	22%	13%
Grand Teton	24	Ramshead Lake	2894	70%	63%	10%	47%
Grand Teton	25	Lake of Crags	2916	70%	64%	10%	48%
Grand Teton	26	Arrowhead	2790	36%	34%	10%	43%
Grand Teton	27	Hedrick Pond	2047	7%	0%	42%	5%
Grand Teton	28	Moose Pond	2065	26%	4%	11%	15%
Grand Teton	29	Delta Lake	2748	65%	74%	18%	41%
Grand Teton	30	Amphitheater	2956	77%	43%	17%	49%
Grand Teton	31	Surprise Lake	2906	79%	8%	7%	47%
Grand Teton	32	Iceflow Lake	3247	86%	88%	30%	61%
Grand Teton	33	Schoolroom	3067	33%	1%	51%	30%
Grand Teton	34	Bradley Lake	2141	20%	23%	22%	18%
Grand Teton	35	Kit Lake	3145	86%	48%	38%	50%
Grand Teton	36	Snowdrift Lake	3051	72%	61%	22%	49%
Grand Teton	37	Taggart Lake	2104	19%	20%	16%	17%
Grand Teton	38	Lake Taminah	2761	58%	50%	18%	39%
Grand Teton	39	Timberline Lake	3141	83%	57%	12%	57%
Grand Teton	40	Rimrock Lake	3023	53%	58%	28%	45%
Grand Teton	41	Phelps Lake	2023	13%	11%	34%	11%
Grand Teton	42	Forget Me Not	2921	72%	42%	21%	48%
Grand Teton	43	Forget Me Not	2964	84%	1%	21%	50%
Grand Teton	44	Forget Me Not	2967	57%	0%	21%	50%
Grand Teton	45	Forget Me Not	2950	41%	22%	26%	44%
Grand Teton	46	Kelly Spring	2037	7%	0%	46%	5%
Grand Teton	47	Coyote Lake	3110	78%	24%	37%	42%
Grand Teton	48	Indian Lake	2988	66%	16%	51%	34%
Grand Teton	49	Marion Lake	2812	40%	5%	76%	15%
Grand Teton	1	No name	2310	15%	0%	63%	8%
Grand Teton	2	No name	2100	10%	0%	54%	5%
Grand Teton	3	No name	2070	4%	0%	42%	5%
Grand Teton	4	No name	2070	6%	0%	42%	5%
Grand Teton	5	No name	2459	12%	0%	67%	9%
Grand Teton	6	No name	2067	3%	0%	64%	5%
Grand Teton	7	No name	2067	5%	0%	66%	5%
Grand Teton	8	No name	2581	13%	0%	74%	11%
Grand Teton	9	No name	2797	15%	11%	56%	23%
Grand Teton	10	No name	2072	6%	0%	21%	6%
Grand Teton	11	No name	2108	10%	0%	35%	6%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Grand Teton	12	No name	2969	74%	61%	26%	47%
Grand Teton	13	No name	2231	8%	0%	37%	7%
Grand Teton	14	No name	2932	59%	36%	39%	36%
Grand Teton	15	No name	2758	57%	41%	23%	38%
Grand Teton	16	No name	2097	10%	0%	28%	7%
Grand Teton	17	No name	2127	7%	0%	37%	6%
Grand Teton	18	No name	3059	74%	58%	29%	48%
Grand Teton	19	No name	2084	5%	0%	36%	6%
Grand Teton	20	No name	2908	60%	50%	24%	44%
Grand Teton	21	No name	2124	7%	0%	35%	6%
Grand Teton	22	No name	2967	58%	58%	22%	48%
Grand Teton	23	No name	2915	56%	27%	23%	45%
Grand Teton	24	No name	3067	66%	58%	20%	54%
Grand Teton	25	No name	2104	7%	0%	35%	6%
Grand Teton	26	No name	2794	58%	43%	23%	37%
Grand Teton	27	No name	2961	73%	33%	7%	48%
Grand Teton	28	No name	2988	58%	41%	31%	40%
Grand Teton	29	No name	2157	11%	0%	67%	6%
Grand Teton	30	No name	2086	2%	0%	36%	5%
Grand Teton	31	No name	2085	8%	0%	67%	5%
Grand Teton	32	No name	2919	74%	13%	10%	38%
Grand Teton	33	No name	2078	9%	0%	33%	6%
Grand Teton	34	No name	2072	4%	0%	33%	6%
Grand Teton	35	No name	2942	72%	47%	18%	48%
Grand Teton	36	No name	2958	34%	9%	21%	48%
Grand Teton	37	No name	2988	69%	31%	19%	51%
Grand Teton	38	No name	2066	6%	0%	54%	7%
Grand Teton	39	No name	2814	74%	44%	8%	43%
Grand Teton	40	No name	2120	10%	0%	36%	6%
Grand Teton	41	No name	3011	86%	37%	11%	52%
Grand Teton	42	No name	2087	9%	0%	64%	6%
Grand Teton	43	No name	2098	11%	0%	66%	6%
Grand Teton	44	No name	2376	29%	28%	27%	22%
Grand Teton	45	No name	2929	58%	56%	19%	48%
Grand Teton	46	No name	2097	19%	12%	18%	20%
Grand Teton	47	No name	2898	51%	19%	31%	36%
Grand Teton	48	No name	2799	67%	51%	19%	43%
Grand Teton	49	No name	2967	76%	63%	18%	50%
Grand Teton	50	No name	2035	7%	0%	36%	5%
Grand Teton	51	No name	3060	77%	48%	18%	53%
Grand Teton	52	No name	2929	73%	49%	18%	48%
Grand Teton	53	No name	2918	55%	30%	21%	45%
Grand Teton	54	No name	2143	5%	0%	22%	10%
Grand Teton	55	No name	3118	87%	28%	12%	56%
Grand Teton	56	No name	1962	7%	0%	36%	4%
Grand Teton	57	No name	3066	69%	14%	31%	48%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate >35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Gr. Sand Dunes	2	No name	3483	0%	38%	73%	39%
Gr. Sand Dunes	3	Upper Sand Cr. Lk.	3580	87%	93%	34%	72%
Gr. Sand Dunes	4	Lower Sand Cr. Lk.	3496	88%	91%	30%	70%
Gr. Sand Dunes	5	No name	3455	0%	93%	35%	68%
Gr. Sand Dunes	6	No name	3729	0%	91%	34%	77%
Gr. Great Sand D	7	Upper Little Sand Lk.	3774	0%	92%	34%	78%
Gr. Sand Dunes	8	Lower Little Sand Lk.	3654	0%	68%	34%	75%
Gr. Sand Dunes	9	No name	3656	0%	3%	61%	42%
Gr. Sand Dunes	10	No name	3552	0%	43%	19%	69%
Gr.Sand Dunes	11	Medano Lake	3509	0%	75%	10%	70%
Gr. Sand Dunes	12	No name	3542	94%	66%	16%	71%
Rocky Mountain	1	No name	3364	93%	82%	16%	65%
Rocky Mountain	2	Mirror Lake	3355	93%	82%	16%	65%
Rocky Mountain	3	Lake Husted	3379	92%	44%	19%	66%
Rocky Mountain	4	Lake Louise	3360	86%	57%	20%	64%
Rocky Mountain	5	Lost Lake	3261	90%	37%	19%	61%
Rocky Mountain	6	Lake Dunraven	3432	85%	72%	19%	67%
Rocky Mountain	7	No name	3507	85%	76%	19%	70%
Rocky Mountain	8	Hazeline Lake	3395	79%	26%	25%	58%
Rocky Mountain	9	No name	4021	95%	57%	19%	84%
Rocky Mountain	10	No name	3670	95%	65%	19%	75%
Rocky Mountain	11	No name	3108	87%	16%	15%	52%
Rocky Mountain	12	Little Crystal Lake	3537	86%	22%	33%	65%
Rocky Mountain	13	Crystal Lake	3505	90%	90%	20%	70%
Rocky Mountain	14	No name	3633	95%	78%	20%	74%
Rocky Mountain	15	Lawn Lake	3349	91%	60%	24%	65%
Rocky Mountain	16	Fay Lakes	3386	91%	79%	20%	66%
Rocky Mountain	17	Fay Lakes	3449	50%	64%	20%	68%
Rocky Mountain	18	Spectacle Lakes	3492	94%	91%	20%	70%
Rocky Mountain	19	Spectacle Lakes	3490	92%	91%	20%	70%
Rocky Mountain	20	Ypsilon Lake	3234	83%	84%	20%	60%
Rocky Mountain	21	Chiquita Lake	3488	91%	68%	20%	70%
Rocky Mountain	22	Lake of the Clouds	3493	69%	11%	43%	49%
Rocky Mountain	23	No name	3632	96%	55%	21%	74%
Rocky Mountain	24	Poudre Lake	3268	85%	2%	25%	48%
Rocky Mountain	25	Fan Lake	2608	43%	21%	14%	34%
Rocky Mountain	26	Sheep Lake	2597	64%	3%	13%	35%
Rocky Mountain	27	Forest Lake	3430	91%	69%	22%	68%
Rocky Mountain	28	No name	3456	83%	25%	35%	59%
Rocky Mountain	29	Rock Lake	3160	82%	56%	23%	57%
Rocky Mountain	30	Arrowhead Lake	3387	88%	70%	23%	65%
Rocky Mountain	31	No name	2713	72%	3%	17%	38%
Rocky Mountain	32	Doughnut Lake	3430	91%	70%	22%	68%
Rocky Mountain	33	Inkwell Lake	3493	90%	75%	23%	69%
Rocky Mountain	34	Timber Lake	3374	87%	27%	34%	56%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35μeq/L	Probability Sum of Base Cations < 124 μeq/L
Rocky Mountain	35	Azure Lake	3625	88%	59%	22%	74%
Rocky Mountain	36	Highest Lake	3786	95%	66%	22%	79%
Rocky Mountain	37	Julian Lake	3381	92%	26%	15%	63%
Rocky Mountain	38	No name	3353	73%	37%	13%	65%
Rocky Mountain	39	No name	3792	95%	23%	18%	79%
Rocky Mountain	40	Hayden Lake	3392	90%	58%	18%	66%
Rocky Mountain	41	Lonesome Lake	3550	89%	58%	18%	72%
Rocky Mountain	42	Haynach Lakes	3374	90%	34%	13%	66%
Rocky Mountain	43	Hourglass Lake	3415	94%	30%	18%	67%
Rocky Mountain	44	Cub Lake	2640	15%	2%	18%	37%
Rocky Mountain	45	No name	3612	93%	57%	18%	74%
Rocky Mountain	46	Rainbow Lake	3577	91%	71%	18%	72%
Rocky Mountain	47	Spruce Lake	2942	56%	52%	28%	48%
Rocky Mountain	48	Loomis Lake	3120	72%	61%	30%	54%
Rocky Mountain	49	Fern Lake	2896	63%	48%	30%	45%
Rocky Mountain	50	Chickaree Lake	2832	77%	1%	9%	44%
Rocky Mountain	51	Odessa Lake	3044	69%	56%	31%	50%
Rocky Mountain	52	Bierstadt Lake	2873	57%	1%	18%	46%
Rocky Mountain	53	No name	2645	31%	1%	20%	37%
Rocky Mountain	54	Two Rivers	3236	60%	14%	31%	58%
Rocky Mountain	55	No name	2652	26%	1%	30%	37%
Rocky Mountain	56	Murphy Lake	3416	87%	30%	12%	67%
Rocky Mountain	57	Ptarmigan Lake	3487	88%	7%	29%	52%
Rocky Mountain	58	No name	3351	93%	40%	12%	65%
Rocky Mountain	59	Bear Lake	2903	78%	15%	30%	47%
Rocky Mountain	60	Emerald Lake	3083	73%	62%	30%	53%
Rocky Mountain	61	Dream Lake	3035	70%	60%	30%	51%
Rocky Mountain	62	Snowdrift Lake	3299	78%	28%	12%	63%
Rocky Mountain	63	Lily Lake	2727	54%	4%	10%	40%
Rocky Mountain	64	Lake Haiyaha	3111	87%	18%	28%	56%
Rocky Mountain	65	Loch Outlet	3103	79%	65%	32%	52%
Rocky Mountain	66	Bench Lake	3091	82%	15%	17%	48%
Rocky Mountain	67	Mills Lake	3030	77%	86%	31%	52%
Rocky Mountain	68	Jewel Lake	3030	77%	90%	35%	52%
Rocky Mountain	69	Glass Lake	3299	87%	79%	21%	62%
Rocky Mountain	70	Sky Pond	3316	86%	80%	21%	63%
Rocky Mountain	71	Shelf Lake	3430	91%	91%	35%	67%
Rocky Mountain	72	Solitude Lake	3490	92%	92%	35%	69%
Rocky Mountain	73	Pettingell Lake	3200	78%	23%	23%	53%
Rocky Mountain	74	Blue Lake	3399	94%	56%	34%	66%
Rocky Mountain	75	Lake Solitude	2965	66%	55%	18%	49%
Rocky Mountain	76	Blake Lake	3252	81%	94%	34%	61%
Rocky Mountain	77	Lake Nokoni	3286	77%	57%	22%	58%
Rocky Mountain	78	Peacock Pool	3431	87%	98%	45%	66%
Rocky Mountain	79	Chasm Lake	3592	89%	99%	36%	72%
Rocky Mountain	80	Frozen Lake	3530	81%	93%	34%	71%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 µeq/L	Probability Nitrate >5 µeq/L	Probability Sulfate > 35 µeq/L	Probability Sum of Base Cations < 124 µeq/L
Rocky Mountain	81	Lake Nanita	3286	81%	74%	18%	62%
Rocky Mountain	82	Lake Powell	3517	86%	46%	36%	58%
Rocky Mountain	83	Keplinger Lake	3570	93%	66%	50%	62%
Rocky Mountain	84	No name	3225	83%	84%	28%	59%
Rocky Mountain	85	Snowbank Lake	3512	94%	88%	36%	68%
Rocky Mountain	86	Lion Lake No. 2	3459	93%	82%	40%	64%
Rocky Mountain	87	No name	3420	90%	54%	51%	55%
Rocky Mountain	88	Lone Pine Lake	2987	59%	21%	27%	37%
Rocky Mountain	89	Lion Lake No. 1	3371	92%	80%	39%	61%
Rocky Mountain	90	Falcon Lake	3371	90%	90%	30%	63%
Rocky Mountain	91	Lake Verna	3102	62%	38%	31%	39%
Rocky Mountain	92	Spirit Lake	3136	70%	45%	47%	42%
Rocky Mountain	93	Thunder Lake	3217	74%	87%	35%	58%
Rocky Mountain	94	Fourth Lake	3154	72%	52%	44%	45%
Rocky Mountain	95	Sandbeach Lake	3134	77%	1%	29%	38%
Rocky Mountain	96	Copeland Lake	2536	60%	0%	11%	29%
Rocky Mountain	97	Fifth Lake	3304	80%	75%	33%	57%
Rocky Mountain	98	Box Lake	3270	91%	62%	33%	62%
Rocky Mountain	99	Eagle Lake	3289	74%	68%	37%	59%
Rocky Mountain	100	Frigid Lake	3600	90%	80%	25%	73%
Rocky Mountain	101	No name	3411	89%	10%	34%	52%
Rocky Mountain	102	Twin Lakes	3000	85%	9%	33%	51%
Rocky Mountain	103	Ouzel Lake	3047	68%	50%	41%	47%
Rocky Mountain	104	Pipit Lake	3478	90%	75%	27%	68%
Rocky Mountain	105	Bluebird Lake	3346	86%	77%	34%	63%
Rocky Mountain	106	No name	3369	87%	54%	39%	55%
Rocky Mountain	107	Junco Lake	3545	90%	61%	30%	68%
Rocky Mountain	108	Finch Lake	3021	83%	2%	13%	50%
Rocky Mountain	109	Adams Lake	3409	55%	5%	56%	37%
Rocky Mountain	110	Hutcheson	3408	88%	79%	32%	67%
Rocky Mountain	111	Cony Lake	3509	88%	92%	25%	70%
Rocky Mountain	112	Hutcheson	3308	72%	81%	33%	62%
Rocky Mountain	113	No name	3307	86%	69%	25%	63%
Rocky Mountain	114	Lake Helene	3220	54%	28%	28%	60%
Yellowstone	1	No name	2785	37%	4%	46%	15%
Yellowstone	2	Sedge Lake	2699	20%	4%	48%	13%
Yellowstone	3	No name	2627	16%	4%	44%	12%
Yellowstone	4	Crescent Lake	2614	13%	9%	46%	12%
Yellowstone	5	No name	2676	31%	6%	48%	13%
Yellowstone	6	High Lake	2674	31%	2%	49%	13%
Yellowstone	7	No name	2655	20%	3%	46%	12%
Yellowstone	8	No name	2591	27%	5%	48%	11%
Yellowstone	9	No name	1662	1%	0%	28%	3%
Yellowstone	10	Rainbow Lake	1792	1%	1%	37%	3%
Yellowstone	11	Sportsman Lake	2349	17%	2%	48%	8%
Yellowstone	12	No name	1960	3%	1%	42%	4%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	14	Crevice Lake	1687	6%	2%	9%	8%
Yellowstone	15	Big Beaver Pond	1938	4%	1%	40%	4%
Yellowstone	16	Cache Lake	2450	13%	4%	59%	10%
Yellowstone	17	No name	2255	11%	0%	40%	7%
Yellowstone	18	No name	2219	11%	0%	33%	7%
Yellowstone	19	No name	2284	8%	0%	31%	7%
Yellowstone	20	Geode Lake	1823	1%	1%	19%	6%
Yellowstone	21	No name	2838	40%	1%	38%	16%
Yellowstone	22	No name	1797	10%	0%	15%	6%
Yellowstone	23	No name	2839	37%	2%	38%	16%
Yellowstone	24	No name	1773	1%	0%	26%	4%
Yellowstone	25	No name	2219	1%	0%	29%	7%
Yellowstone	26	No name	1779	5%	0%	24%	4%
Yellowstone	27	No name	1991	3%	0%	33%	5%
Yellowstone	28	McBride Lake	2007	20%	8%	6%	16%
Yellowstone	29	No name	2013	13%	2%	16%	8%
Yellowstone	30	No name	2210	7%	1%	30%	7%
Yellowstone	31	No name	2225	9%	1%	51%	7%
Yellowstone	32	No name	2011	4%	0%	32%	5%
Yellowstone	33	Fawn Lake	2373	4%	0%	56%	8%
Yellowstone	34	No name	2015	6%	0%	31%	5%
Yellowstone	35	Floating Is Lake	2010	3%	1%	36%	5%
Yellowstone	36	No name	2331	18%	0%	51%	8%
Yellowstone	37	No name	2335	9%	1%	51%	8%
Yellowstone	38	Junction Lake	1904	7%	0%	25%	5%
Yellowstone	39	No name	1889	22%	8%	6%	13%
Yellowstone	40	Small Lake	2765	36%	7%	56%	15%
Yellowstone	41	No name	1886	7%	0%	19%	6%
Yellowstone	42	No name	2325	16%	0%	43%	8%
Yellowstone	43	No name	1886	2%	1%	12%	8%
Yellowstone	44	No name	2102	38%	2%	5%	20%
Yellowstone	45	No name	2321	11%	6%	58%	8%
Yellowstone	46	No name	2227	14%	0%	36%	7%
Yellowstone	47	No name	2320	17%	0%	43%	8%
Yellowstone	48	Swan Lake	2215	5%	0%	39%	7%
Yellowstone	49	Hals Lake	2335	15%	2%	59%	8%
Yellowstone	50	Ruddy Duck	2199	9%	0%	36%	6%
Yellowstone	51	Trumpiter	1862	2%	0%	31%	4%
Yellowstone	52	No name	2219	5%	0%	36%	7%
Yellowstone	53	No name	2190	7%	0%	36%	6%
Yellowstone	54	No name	2211	10%	0%	36%	6%
Yellowstone	55	No name	2199	9%	0%	36%	6%
Yellowstone	56	No name	1885	7%	0%	22%	5%
Yellowstone	57	No name	1893	2%	0%	31%	4%
Yellowstone	58	No name	2217	6%	0%	43%	7%
Yellowstone	59	No name	2301	7%	0%	43%	7%
Yellowstone	60	Divide Lake	2201	9%	1%	56%	7%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	61	Lost Lake	2046	3%	0%	19%	9%
Yellowstone	62	Buck Lake	2113	10%	0%	44%	6%
Yellowstone	63	No name	2067	5%	0%	26%	6%
Yellowstone	64	No name	2074	4%	0%	32%	5%
Yellowstone	65	Trout Lake	2119	8%	2%	40%	6%
Yellowstone	66	No name	2077	2%	0%	32%	5%
Yellowstone	67	No name	2303	11%	0%	54%	7%
Yellowstone	68	No name	2029	9%	1%	38%	5%
Yellowstone	69	No name	2567	15%	7%	63%	11%
Yellowstone	70	No name	2750	29%	31%	40%	24%
Yellowstone	71	No name	2592	14%	7%	63%	11%
Yellowstone	72	No name	2790	49%	78%	20%	39%
Yellowstone	73	No name	2688	22%	9%	64%	13%
Yellowstone	74	Gallatin Lake	2688	14%	15%	63%	13%
Yellowstone	75	No name	2243	3%	0%	35%	7%
Yellowstone	76	No name	2309	15%	9%	45%	11%
Yellowstone	77	Ace of Hearts Lake	2464	11%	0%	44%	9%
Yellowstone	78	Obsidian Lake	2356	8%	1%	43%	8%
Yellowstone	79	No name	2468	15%	0%	44%	9%
Yellowstone	80	No name	2270	10%	1%	53%	7%
Yellowstone	81	No name	2473	11%	1%	44%	10%
Yellowstone	82	No name	2391	18%	1%	45%	8%
Yellowstone	83	Middle Trilobite	2674	25%	11%	62%	13%
Yellowstone	84	Trilobite Lake	2544	16%	9%	64%	11%
Yellowstone	85	Echo Lake	2699	12%	7%	65%	13%
Yellowstone	86	No name	2384	19%	0%	59%	8%
Yellowstone	87	No name	2250	10%	1%	49%	7%
Yellowstone	88	Beaver Lake	2251	10%	1%	49%	7%
Yellowstone	89	Rosa Lake	2748	19%	5%	62%	14%
Yellowstone	90	Grizzly Lake	2288	10%	1%	54%	8%
Yellowstone	91	No name	2401	12%	0%	48%	9%
Yellowstone	92	Lake of the Woods	2362	9%	1%	42%	9%
Yellowstone	93	No name	2533	22%	0%	58%	10%
Yellowstone	94	North Twin Lake	2301	5%	1%	49%	7%
Yellowstone	95	South Twin Lake	2295	8%	1%	49%	7%
Yellowstone	96	No name	2008	3%	0%	41%	5%
Yellowstone	97	No name	2010	3%	0%	42%	5%
Yellowstone	98	No name	2418	16%	0%	53%	9%
Yellowstone	99	No name	2413	14%	0%	49%	9%
Yellowstone	100	Grebe Lake	2445	11%	1%	40%	9%
Yellowstone	101	Cascade Lake	2435	10%	1%	39%	9%
Yellowstone	102	No name	2341	17%	0%	46%	8%
Yellowstone	103	Nymph Lake	2283	12%	1%	49%	7%
Yellowstone	104	No name	2711	29%	1%	43%	13%
Yellowstone	105	No name	2285	6%	0%	47%	7%
Yellowstone	106	Wolf Lake	2438	15%	1%	42%	9%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 µeq/L	Probability Nitrate >5 µeq/L	Probability Sulfate > 35 µeq/L	Probability Sum of Base Cations < 124 µeq/L
Yellowstone	107	No name	2288	9%	0%	46%	7%
Yellowstone	108	No name	2543	14%	0%	38%	11%
Yellowstone	109	Mirror Lake	2727	20%	0%	38%	16%
Yellowstone	110	Ribbon Lake	2382	12%	0%	38%	8%
Yellowstone	111	Wapiti Lake	2569	19%	1%	42%	11%
Yellowstone	112	Ice Lake	2405	9%	0%	32%	12%
Yellowstone	113	No name	2638	31%	0%	44%	12%
Yellowstone	114	No name	2594	13%	0%	41%	11%
Yellowstone	115	Clear Lake	2382	11%	0%	37%	8%
Yellowstone	116	No name	2299	10%	0%	53%	7%
Yellowstone	117	Solfatara Lake	2503	11%	0%	40%	10%
Yellowstone	118	No name	2553	19%	0%	46%	11%
Yellowstone	119	Wrangler Lake	2393	5%	0%	37%	8%
Yellowstone	120	Wapiti Lake	2505	5%	0%	9%	30%
Yellowstone	121	No name	2528	18%	0%	40%	10%
Yellowstone	122	Turn Lake	2505	15%	1%	37%	11%
Yellowstone	123	No name	2489	12%	0%	38%	10%
Yellowstone	124	No name	2340	10%	1%	40%	8%
Yellowstone	125	No name	2344	14%	0%	31%	10%
Yellowstone	126	Cygnet Lake #1	2527	5%	0%	43%	10%
Yellowstone	127	Cygnet Lake #2	2527	7%	0%	42%	10%
Yellowstone	128	Cygnet Lake #3	2527	7%	0%	42%	10%
Yellowstone	129	West Tern Lake	2505	13%	1%	34%	12%
Yellowstone	130	No name	2536	4%	0%	41%	10%
Yellowstone	131	Cygnet Lake #4	2527	7%	0%	42%	10%
Yellowstone	132	Cygnet Lake #5	2527	7%	0%	42%	10%
Yellowstone	133	No name	2512	8%	1%	29%	14%
Yellowstone	134	White Lake	2509	13%	1%	37%	11%
Yellowstone	135	No name	2518	2%	0%	11%	27%
Yellowstone	136	No name	2490	7%	1%	39%	10%
Yellowstone	137	No name	2529	18%	1%	38%	11%
Yellowstone	138	Harlequin Lake	2094	9%	1%	65%	5%
Yellowstone	139	No name	2394	5%	0%	39%	8%
Yellowstone	140	No name	2539	21%	0%	39%	10%
Yellowstone	141	No name	2542	9%	1%	39%	11%
Yellowstone	142	No name	2076	5%	2%	64%	5%
Yellowstone	143	No name	2551	22%	1%	43%	11%
Yellowstone	144	No name	2749	13%	4%	43%	14%
Yellowstone	145	Frost Lake	2897	27%	0%	46%	17%
Yellowstone	146	No name	2358	4%	0%	51%	8%
Yellowstone	147	No name	2394	15%	0%	36%	8%
Yellowstone	148	No name	2399	16%	0%	36%	9%
Yellowstone	149	No name	2393	11%	0%	37%	8%
Yellowstone	150	No name	2301	12%	0%	51%	7%
Yellowstone	151	Mary Lake	2510	15%	0%	44%	10%
Yellowstone	152	No name	2228	9%	0%	47%	7%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	153	No name	2530	14%	0%	44%	10%
Yellowstone	154	No name	2522	8%	0%	44%	10%
Yellowstone	155	No name	2200	12%	0%	55%	6%
Yellowstone	156	No name	2281	13%	0%	54%	7%
Yellowstone	158	No name	2357	13%	0%	34%	8%
Yellowstone	159	No name	2199	3%	0%	57%	6%
Yellowstone	160	Indian Pond	2364	6%	0%	21%	12%
Yellowstone	161	Big Bear Lake	2437	18%	1%	74%	10%
Yellowstone	162	No name	2358	5%	0%	35%	8%
Yellowstone	163	Big Bear Lake	2438	18%	3%	73%	11%
Yellowstone	164	Turbid Lake	2388	14%	2%	41%	9%
Yellowstone	165	Beach Springs	2358	13%	0%	36%	8%
Yellowstone	166	Dryad Lake	2530	11%	0%	42%	10%
Yellowstone	167	No name	2202	0%	0%	57%	6%
Yellowstone	168	Feater Lake	2201	1%	0%	57%	6%
Yellowstone	169	Hot Lake	2244	9%	2%	56%	7%
Yellowstone	170	Beach Lake	2483	14%	1%	37%	11%
Yellowstone	171	Goose Lake	2201	4%	3%	60%	6%
Yellowstone	172	No name	2538	11%	1%	45%	10%
Yellowstone	173	No name	2197	5%	0%	57%	6%
Yellowstone	174	Lower Basin	2587	29%	0%	45%	11%
Yellowstone	175	Crater Lake #4	2286	6%	1%	61%	7%
Yellowstone	176	No name	2283	12%	1%	61%	7%
Yellowstone	177	Bridge Bay	2357	13%	0%	34%	8%
Yellowstone	178	No name	2483	15%	1%	39%	11%
Yellowstone	179	No name	2489	22%	0%	44%	10%
Yellowstone	182	Gooseneck Lake	2234	10%	2%	58%	7%
Yellowstone	184	No name	2558	24%	1%	45%	11%
Yellowstone	185	No name	2444	16%	0%	60%	9%
Yellowstone	186	No name	2393	5%	0%	35%	8%
Yellowstone	187	No name	2376	8%	0%	35%	8%
Yellowstone	188	Sylvan Lake	2564	21%	1%	50%	11%
Yellowstone	189	Delacy Lake	2593	18%	4%	53%	13%
Yellowstone	190	Delacy Lake	2593	14%	3%	51%	13%
Yellowstone	191	Mallard Lake	2454	13%	5%	56%	9%
Yellowstone	192	No name	2605	17%	2%	52%	12%
Yellowstone	193	No name	2614	14%	4%	58%	12%
Yellowstone	194	No name	2538	11%	1%	50%	10%
Yellowstone	196	No name	2571	19%	3%	58%	11%
Yellowstone	197	Chickadee Lake	2534	6%	1%	50%	10%
Yellowstone	198	No name	2547	14%	0%	50%	11%
Yellowstone	199	Teal Lake	2568	20%	3%	55%	12%
Yellowstone	200	Nuthatch Lake	2547	12%	0%	50%	11%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	202	No name	2553	15%	0%	50%	11%
Yellowstone	204	No name	2413	14%	7%	59%	9%
Yellowstone	206	No name	2357	5%	0%	38%	8%
Yellowstone	207	Scaup Lake	2411	14%	7%	58%	9%
Yellowstone	208	No name	2373	5%	1%	51%	8%
Yellowstone	209	Summit Lake	2605	14%	1%	66%	13%
Yellowstone	210	Shoshone Lake	2374	9%	8%	55%	10%
Yellowstone	211	Delusion Lake	2383	6%	0%	25%	12%
Yellowstone	212	No name	2354	11%	1%	48%	8%
Yellowstone	213	Pocket Lake	2486	12%	3%	60%	10%
Yellowstone	214	No name	2355	14%	0%	42%	8%
Yellowstone	215	No name	2388	6%	0%	45%	8%
Yellowstone	216	Hidden Lake	2388	6%	0%	45%	8%
Yellowstone	217	No name	2544	21%	0%	71%	11%
Yellowstone	218	No name	2393	15%	0%	45%	8%
Yellowstone	219	No name	2401	3%	0%	45%	9%
Yellowstone	220	No name	2386	9%	0%	45%	8%
Yellowstone	221	No name	2548	1%	0%	68%	11%
Yellowstone	222	No name	2396	17%	2%	70%	9%
Yellowstone	223	No name	2429	20%	0%	50%	9%
Yellowstone	224	Riddle Lake	2412	8%	0%	52%	9%
Yellowstone	225	No name	2374	10%	2%	60%	8%
Yellowstone	226	No name	2412	2%	0%	52%	9%
Yellowstone	227	Glade Lake	2942	36%	16%	49%	18%
Yellowstone	228	Madison Lake	2502	2%	12%	69%	10%
Yellowstone	229	Alder Lake	2360	11%	1%	43%	8%
Yellowstone	230	No name	2375	8%	3%	69%	8%
Yellowstone	231	Buffalo Lake	2342	13%	0%	61%	9%
Yellowstone	232	No name	2359	7%	0%	31%	11%
Yellowstone	233	No name	2373	10%	0%	44%	8%
Yellowstone	234	Lewis Lake	2371	9%	5%	57%	9%
Yellowstone	236	No name	2371	13%	1%	54%	8%
Yellowstone	237	No name	2359	5%	0%	41%	9%
Yellowstone	240	No name	2359	3%	0%	44%	8%
Yellowstone	241	No name	2358	4%	0%	44%	8%
Yellowstone	242	No name	2359	4%	0%	44%	8%
Yellowstone	243	No name	2359	9%	1%	43%	8%
Yellowstone	244	No name	2358	3%	0%	37%	10%
Yellowstone	245	No name	2362	8%	0%	43%	8%
Yellowstone	246	No name	2458	19%	0%	48%	9%
Yellowstone	247	No name	2363	13%	0%	43%	8%
Yellowstone	248	No name	2358	6%	2%	43%	8%
Yellowstone	249	No name	2359	2%	0%	43%	8%
Yellowstone	250	Aster Lake	2489	10%	1%	56%	10%
Yellowstone	251	No name	2368	15%	1%	44%	8%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	252	No name	2385	1%	2%	44%	8%
Yellowstone	253	No name	2359	2%	0%	43%	8%
Yellowstone	255	No name	2363	2%	0%	43%	8%
Yellowstone	257	Heart Lake	2272	9%	3%	46%	8%
Yellowstone	258	Trail Lake	2363	6%	2%	42%	9%
Yellowstone	259	Outlet Lake	2371	14%	1%	47%	9%
Yellowstone	260	No name	2702	20%	0%	47%	13%
Yellowstone	261	No name	2670	32%	0%	76%	13%
Yellowstone	263	No name	2133	8%	2%	55%	6%
Yellowstone	264	No name	2143	6%	0%	55%	6%
Yellowstone	265	Little Robinson	1978	4%	1%	56%	5%
Yellowstone	266	Lake Wyodaho	2067	7%	1%	55%	5%
Yellowstone	267	No name	2167	8%	1%	55%	6%
Yellowstone	268	No name	1977	1%	0%	48%	5%
Yellowstone	269	Ranger Lake	2122	6%	1%	34%	8%
Yellowstone	270	No name	2382	1%	0%	41%	8%
Yellowstone	271	No name	1951	4%	0%	48%	4%
Yellowstone	272	No name	1953	6%	0%	48%	4%
Yellowstone	273	No name	2627	18%	2%	46%	12%
Yellowstone	274	No name	2314	17%	1%	55%	8%
Yellowstone	275	No name	2383	3%	0%	41%	8%
Yellowstone	276	Basin Creek Lake	2252	11%	1%	56%	7%
Yellowstone	277	No name	2835	36%	1%	46%	16%
Yellowstone	278	No name	1977	2%	0%	47%	5%
Yellowstone	279	No name	1971	1%	0%	46%	5%
Yellowstone	280	No name	1960	4%	0%	46%	4%
Yellowstone	281	No name	1977	2%	0%	47%	5%
Yellowstone	282	No name	2994	34%	2%	51%	20%
Yellowstone	283	No name	2784	33%	0%	46%	15%
Yellowstone	284	No name	1977	2%	0%	47%	5%
Yellowstone	285	No name	1963	6%	0%	46%	4%
Yellowstone	286	Robinson Lake	1977	3%	0%	36%	6%
Yellowstone	287	Lilypad Lake	1953	3%	0%	43%	5%
Yellowstone	288	No name	1975	3%	0%	47%	5%
Yellowstone	289	Forest Lake	2262	6%	0%	37%	9%
Yellowstone	290	Beula Lake	2256	9%	2%	60%	7%
Yellowstone	291	No name	2317	15%	0%	53%	8%
Yellowstone	292	No name	2180	6%	0%	63%	6%
Yellowstone	293	Mariposa Lake	2729	27%	2%	51%	14%
Yellowstone	294	No name	1943	2%	0%	46%	4%
Yellowstone	295	Hering Lake	2257	7%	2%	56%	8%
Yellowstone	296	No name	1971	2%	0%	44%	5%
Yellowstone	297	No name	2285	7%	0%	56%	7%
Yellowstone	298	Phoneline Lake	1935	2%	0%	43%	4%
Yellowstone	299	No name	2097	8%	0%	43%	5%
Yellowstone	300	No name	2497	21%	1%	60%	10%
Yellowstone	301	No name	2247	10%	1%	63%	7%

National Park	ID	Lake Name	Elevation (meters)	Probability ANC < 100 μeq/L	Probability Nitrate >5 μeq/L	Probability Sulfate > 35 μeq/L	Probability Sum of Base Cations < 124 μeq/L
Yellowstone	302	No name	2095	7%	0%	43%	5%
Yellowstone	303	Winegar Lake	1963	3%	0%	34%	6%
Yellowstone	304	Tanager Lake	2124	8%	0%	39%	6%
Yellowstone	305	No name	1962	2%	0%	44%	4%
Yellowstone	306	No name	1977	6%	0%	44%	5%
Yellowstone	307	South Boundary	2247	8%	1%	61%	7%
Yellowstone	157	Yellowstone Lk	2353	5%	9%	24%	13%