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Assessment of Atmospheric Nitrogen Deposition: Possible Effects on Alpine Ecosystems Above 9000 Feet in Grand Teton National Park

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ASSESSMENT OF ATMOSPHERIC NITROGEN DEPOSITION: POSSIBLE EFFECTS ON ALPINE ECOSYSTEMS ABOVE 9000 FEET IN GRAND TETON NATIONAL PARK

by

Jennifer Hansen

A thesis submitted in partial fulfillment
of the requirements for the degree

of

MASTER OF SCIENCE

in

Ecology

Approved:

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2012

ABSTRACT

Assessment of Atmospheric Nitrogen Deposition: Possible Effects on Alpine Ecosystems Above 9000 Feet in Grand Teton National Park

by

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Utah State University, 2012

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Atmospheric N deposition is becoming a stressor on ecosystems in the western U.S. There are few National Atmospheric Deposition Program (NADP) monitoring sites and little is known about N deposition impacts on terrestrial ecosystems in the Intermountain West. Alpine ecosystems may be particularly sensitive to changes in N inputs because of the shallow soils, short growing seasons, and sparse plant cover. This study focused on N deposition effects on an alpine ecosystem in Grand Teton National Park located along a modeled N deposition gradient (Moose Basin high, Paint Brush Medium, Rendezvous Mtn. low) and across contrasting edaphic conditions using a two-factorial design. At each location, we estimated N deposition and measured soil moisture and temperature across edaphic conditions, soil parameters (total and extractable N, available N, net mineralization, and nitrification potential), and plant community characteristics (species richness, species composition, percent cover, plant and root biomass, N content, and above and belowground plant components). These response variables were used to test whether there is a north to south N deposition gradient, if N deposition and N status are affected by soil moisture content, and whether soil and/or plant properties were affected by N deposition and edaphic conditions and if the response variables can serve as indicators as early warning signs of N saturation. The Tetons receive 1.42 kg N ha^{-1} yr^{-1} with more in winter (0.85-1.17 kg N ha^{-1} yr^{-1}) than during the summer (0.25 kg N ha^{-1} yr^{-1}). Soil moisture content was related to snowpack accumulations and melt but did not affect N status. Moose Basin (i.e., high N deposition) showed characteristics of an N-rich site shown by higher soil N content and extractable soil NH_4^+, higher nitrification potential, low C:N ratios, more aboveground biomass, and higher foliar N content compared to the RDV location. Rendezvous Mountain (i.e., low N deposition) showed characteristics of an N-poor site having lower soil extractable N, high C:N ratios in soil and roots, and low N mineralization potential. Paint Brush was highly variable in soil and plant characteristics and most clearly showed differences between wet and dry sites. In terms of N status, it was intermediate and shared similarities with both N-poor and N-rich sites. This study shows that it is important to consider both soil and plant indicators (i.e., total and extractable N in soils, soil nitrification potential, above and belowground biomass, and N content) together to assess N status. The C:N ratio of plants and soils was less informative than anticipated. Species richness and composition was a less sensitive indicator of N-induced change and should be considered for long-term assessments only.

(184 pages)

PUBLIC ABSTRACT

Assessment of Atmospheric Nitrogen Deposition: Possible Effects on Alpine Ecosystems Above 9000 Feet in Grand Teton National Park

by

Jennifer Hansen, Master of Science

Certain forms of nitrogen (N) in the atmosphere are pollutants with effects that mimic fertilizer application. If there is too much N, it can become a stressor, and the ecosystem may undergo drastic changes (e.g. certain plant species may decline or disappear). The N load at which a system starts exhibiting negative effects is dependent on the type and location of the ecosystem. Alpine ecosystems (i.e. above 9000 feet in Wyoming) may be particularly sensitive to low levels of atmospheric N input because of short growing seasons, sparse plant cover, and shallow soils that limit their ability to absorb the extra N. It is therefore very useful to have early warning signs of changes in ecosystem N dynamics.

The National Atmospheric Deposition Program (NADP) monitors air quality and pollutant inputs with precipitation using instruments set up at various sites across the U.S., but there are only a limited number of NADP locations in the western U.S., with few locations at high elevation. Therefore, for many locations in the Rocky Mountains, N deposition is often modeled from the few available NADP monitoring sites. Even less is

known about N deposition impacts on ecosystems in the Intermountain West, especially in the sensitive alpine ecosystems.

This study focused on N deposition effects on an alpine ecosystem located in the Grand Teton National Park. Modeling of N deposition for the Rocky Mountains has predicted a north to south gradient in the Grand Teton National Park. The objective of this project was to evaluate whether small changes in atmospheric N deposition had detectable effects on alpine plant communities and soils of Grand Teton National Park. First, we wanted to see if there was an actual N deposition gradient from north to south in the park by locating our measurements at three locations predicted to receive of high (Moose Basin), medium (Paint Brush Divide), and low (Rendezvous Mountain) N deposition. Secondly we wanted to investigate whether any of these alpine systems were already showing signs of excess available N. This was achieved by gathering information on how much N was coming from the atmosphere to each location, and by looking at various plant and soil parameters indicative of the N content.

Atmospheric N bonds to water molecules in the air and returns to the Earth with precipitation events such as rain and snow. Alpine ecosystems in the Grand Teton National Park receive most of their annual precipitation in the form of snow. Thus, snow distribution and relative melt rate dictates where the water accumulates in this landscape, creating areas that are either wet or dry in the summer months. This is important to plant and soil communities. Since atmospheric N inputs follow water, we thought that the wet areas would have different amounts of available N compared to dry areas, which in turn, would cause differences in plant and soil properties. To see if there were differences in wet versus dry areas, we compared ways that plants and soils use and/or store N across

this presumed N deposition gradient. In summary, we followed N from the atmosphere, into the plants, then the roots, and then to the soil.

Our results show that the Tetons receive modest amounts of atmospheric N (< 2 kg ha^{-1} yr^{-1}) mostly in winter (85%) with very small amounts coming in during the summer (15%, <0.5 kg ha^{-1} yr^{-1}) months. We confirmed that snow pack accumulation and snow melt created wet and dry sites, but these sites were not different in terms of N status.

At the three study locations in Grand Teton National Park, there were small, but significant differences in N availability that were expressed in plant and soil properties. Moose Basin (i.e., high N deposition) showed characteristics of an N-rich site having more N in the soil, more plant biomass, and more N in the plants, while Rendezvous Mountain (i.e., low N deposition) showed characteristics of an N-poor site with less N in the soil, and less ability to process N in the soil. Paint Brush (i.e. medium N deposition) shared similarities with both N-poor and N-rich sites.

This study shows that even small changes in atmospheric N input can cause fundamental changes in ecosystem characteristics. In order to detect these changes, it is important to look at both plant and soils together. We found that some characteristics such as total and extractable N in soils, soil nitrification potential, above and belowground biomass, and N content can be used as early warnings signs of ecosystem N overload.

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Jennifer Hansen

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CHAPTER 1

INTRODUCTION

Air pollution in the U.S. remains a growing problem (Fenn et al., 2003 a and b) despite legislation to improve air quality and human health (Heidorn 1979). In the 1970s, acid deposition was discovered as a cause of a major decrease in diversity in both aquatic and terrestrial ecosystems (Scott et al., 1989). NOx and NHx have been found to cause a decline in both ecosystem diversity and productivity because they act as a fertilizer for plant life and overload the system with N. Airborne forms of anthropogenic N (specifically NOx and NHx) react with water vapor in the atmosphere and deposit at different locations within ecosystems, affecting the structure and function of these ecosystems. NOx originates mostly from automobile and industrial emissions. Sources of atmospheric ammonia deposition (NHx) include decomposing excrement from livestock (Vitousek et al., 1997), volatilization from the production and application of fertilizer (Fenn et al., 2003 a), NHx loss from burning biomass (Asman et al., 1998), and the natural N cycle of soils (Dawson 1977; Bouwman et al., 1997). Ninety percent of NOx emissions come from northern latitude countries such as the U.S., Canada, Western and Eastern Europe, the former USSR, China, Japan, and the Middle East (Olivier et al., 1995).

Nitrogen deposition is predicted to increase 25% worldwide by the year 2020 (Peterjohn et al., 1996) causing some ecosystems to become overloaded with N. Excessive N in the soil may lead to so-called N-saturation where N availability exceeds the demands of microbes and plants, leaving an abundance of N in the soil matrix that can cause ecosystem function to decline (Vitousek et al., 1997; Langan 1999). Most ecosystems have evolved in and adapted to N-limited environments since N_2 gas, the most abundant form of N in the atmosphere is difficult to transform into useful forms for plant and soil microbial life (Fenn et al., 1998).

N Deposition in the Eastern vs. Western U.S.

The Eastern U.S. receives around 7-17 kg N $ha^{-1}yr^{-1}$ (NADP, 2005) of wet N deposition that is a regional problem because of the dense human population in the eastern U.S. The Western U.S. receives 1-6 kg N $ha^{-1}yr^{-1}$ (NADP, 2005) of N deposition and has a large amount of land that is being exposed to low levels of N deposition. There are “hot spots” in some localized areas downwind from major metropolitan areas such as the Colorado Front Range and the Los Angeles Basin where N deposition rates are respectively 9 and 35 kg N $ha^{-1}yr^{-1}$ (Bowman and Steltzer 1998; Nanus et al., 2003, Fenn et al., 1998, 2003 b). Typically in the Western U.S., the closer the ecosystem is to a large city, the greater the likelihood for ecosystem damage (Meixner et al., 2002). However, large agricultural areas in the Western U.S. are a cause for concern because of the amount of NH_x that originates from them (Fenn et al., 2003 b).

There have been few studies of anthropogenic atmospheric deposition in the arid western U.S. Western soils were originally thought to be less affected by acid deposition compared to soils in the Eastern U.S. because the soils have high base saturation, pH, and acid buffering capacities (Belnap and Eldridge 2001; Nagy et al., 2005). Soil types in the Western U.S. (excluding the Pacific Coast) are primarily made up of Aridisols, Mollisols, some Inceptisols and Andisols, and a small amount of Alfisols and Entisols. Most of these soils have a high base saturation (USDA staff, 1999). Since there is less precipitation in the Interior Western U.S., the soils tend to hold much more Ca^{2+} and commonly contain $CaCO_3$, causing the pH of these soils to be higher than the pH of the soils in the Eastern U.S., creating an acid buffering system for the arid soils. Soil types in the Eastern U.S. are primarily made up of Ultisols, Inceptisols, Spodosols (in the northeast), and Alfisols (in the mid-eastern U.S.) (USDA staff, 1999). These soils typically receive more rainfall precipitation causing nutrients such as Ca^{2+} to leach out and soil pH to be lower, thus making them more vulnerable to acid deposition than western soils.

N Deposition Effects on Ecosystem Structure and Function

Nitrogen is considered a limiting factor for plant growth in most ecosystems (Aber et al., 1989; Körner 1989; Peterjohn et al., 1996; Näsholm 1997; Friedland and Miller, 1999) especially alpine and tundra ecosystems. N deposition causes a general N enrichment of soils (Fenn et al., 1998 and 2003 b), which can cause changes in ecosystem structure and function such as decreased diversity of mycorrhizal communities (Egerton-Warburton and Allen, 2000; Van Der Eerden, 1998), increased growth of invasive species (Weiss, 1999; Brooks, 2003), lichen community shifts (McCune et al., 1998; Geiser and Neitlich 2006), forest expansion into grasslands (Köchy and Wilson, 2001), altered cryptobiotic crusts in desert systems (Belnap et al., 2003), and shifts in alpine plant communities (Bowman and Steltzer, 1998; Bowman, 2000).

Excessive N in the soil may lead to so-called N-saturation (Aber et al., 1989; Vitousek et al., 1997; Langan, 1999). According to Aber et al., (1989, 1998), a series of events take place in plant and soil microorganisms when N input to the ecosystem increases: plant communities respond to increased N by an initial increase in foliar N, net primary productivity (NPP), and an overall increase in aboveground and root biomass (Aber et al., 1989). Accelerated tree growth in forests causes the trees to sequester N initially, but long-term chronic inputs of N in later stages of N-saturation cause tree growth to decline, sometimes resulting in tree mortality (Aber et al., 1998; Fenn et al., 2003 b). Increases in foliar N lead to a decline in foliar and litter C:N ratio resulting in an increase in litter decomposition, net increase in N mineralization in the soil, a build up of NH_4^+, stimulation of NO_3^- production , and excess NO_3^- in the soil matrix that may leach from the soil and into aquatic ecosystems. In the later stages of older litter decomposition, breakdown rates may decrease significantly with increased N. This is because higher levels of N reduce the efficiency of ligninolytic enzymes inside microbes that break down lignin, causing it to build up in the soil resulting in much slower humus formation (Magill and Aber, 1998). In the last stages of N-saturation, there is so much NO_3^- and NH_4^+ in the ecosystem (relative to background levels of N), that N availability far exceeds the demands for plants and microbes and N may become toxic or inhibit growth (Aber, 1992).

Too much N can lead to several progressive outcomes:

1) Some plant communities will increase their cover range (i.e. spread out), decrease their cover range (i.e. cover smaller areas), or disappear all together (Rusek, 1993; Hungate et al., 1997; Suding et al., 2006) as plants inherently differ for their abilities to process and uptake N (Miller et al., 2007a; Ashton et al., 2008).

2) Competitive interactions among plant communities will change (i.e., since N is no longer a limiting factor other factors become limiting), resulting in composition and cover changes (Miller et al., 2007a and b).

3) Relationships among organisms will shift (Suding et al., 2006; Nemergrut et al., 2008).

4) Root mycorrhyzal relationships will decline resulting in less uptake of nutrients other than N by plants (Van Der Eerden, 1998).

5) Plants will become deficient in other resources such as phosphorus, calcium, etc.

The cover range and composition of plant communities reflects the fact that N is a limiting factor in biomass production. Most plants have evolved to grow with limited amounts of N. Theodose et al., (1996) suggest that excess N availability may be a mechanism used by rare plants as a way to coexist with competitive dominant species, thus increasing or maintaining their cover range. The premise is that with elevated soil N, N is no longer limiting and there is no competition for N, allowing rare plants to exist among dominant species that would normally outcompete them. Nitrophilic plants may thrive simply by their natural ability to uptake higher amounts of N compared to plants with inherently lower uptake rates of N. However, in the saturation stages of N deposition, with too much NO_3^- inside plant tissues, too many free radicals will form and the plant's enzymes designed to handle these radicals will become overloaded, which may result in phytotoxicity and an overall reduction in cover range (Näsholm, 1997).

Competitive interactions among plant communities will shift. Too much N may cause plant diversity and species richness to decrease because plants inherently differ in their tolerance of N concentration and type of N uptake (Ashton et al., 2008; Miller and Bowman, 2003; Miller et al., 2007a; Suding et al., 2006). Lesham (1996) conducted a study on NO (an N air pollutant resulting from combusting fossil fuels) in legume plants that demonstrated that production of ethylene (the compound that causes fruits to mature and ripen) was inhibited with low concentrations of gaseous NO, and stimulated with high concentrations of gaseous NO. In natural ecosystems, this stimulation of fruit ripening with high atmospheric N concentrations could be a mechanism for strong competition that may result in a mono-specific ecosystem with little to no diversity (Bowman et al., 1993; Vitousek et al., 1997). Some plants preferentially take up only NH_4^+, resulting in weak competitive abilities compared to plants that take up both NH_4^+ and NO_3^-. These specialized plants that only take up one form of N could undergo population decline because they will be subject to exploitative competition from other plants that are able to take up different forms of N (Ashton et al., 2008; Miller et al., 2007a; Näsholm 1997).

If plant communities change, then so will soil microbial and animal communities. Organisms that rely on certain plant communities for resources may disappear or go extinct if those plant species decline (Fenn et al., 2003 b). Mycorrhizal fungal biomass may be altered because lignin-degrading enzymes may decline and microbial communities may shift and start to metabolize different substrates (Frey et al., 2004; Ashton et al., 2008; Goulding et al., 1998). This shift in microbial biomass and substrate metabolism may be reflected in changes to soil nutrient dynamics that in turn will affect plant growth, plant function, and root symbiotic relationships (Bever et al., 1997; Fenn et al., 1998; Goulding et al., 1998). It can cause an alteration of plant community structure and function by shifting nutrient availability to plant roots and other soil biota, or change the pH of the soil all together (Fenn et al., 2003 b; Ashton et al., 2008).

With excess N, other soil nutrients may become deficient. An increase in net nitrification follows the early stages of N-saturation creating an excess of NO_3^- in the soil matrix (Bowman et al., 2006). Nitrate and ammonium that has not been taken up by plants or microbes can interact with cation exchange sites and mobilize nutrient cations such as Ca^{2+}, K^+ and Mg^{2+} which will then leach from the soil in association with mobile NO_3^- and become lost from the ecosystem. This can result in an overall decline in nutrient availability (Magill et al., 2004). In acid soils, NO_3^- can also mobilize toxic cations such as Al^{3+} that are harmful to plants (Asner et al., 2001).

Since N deposition has become a concern in the early 1990's, scientists have found ecosystems in the Western U.S. that are already showing early signs of N-saturation (Fenn et al., 2003 b). Scientists have attempted to come up with criteria to indicate that an ecosystem has become saturated with N (Aber et al., 1989; Aber 1992, Stoddard 1994; Peterjohn et al., 1996). This may include (but is not limited to) signs such as:

1) High rates of net nitrification and mineralization.

2) N concentrations in the soil that show little seasonal changes (little variability) because microbial communities have shifted dominance.

3) Rapid N leaching from soil with additional N inputs, because the system is becoming N overloaded and cannot handle any more uptake of N.

4) Denitrification with elevated atmospheric N deposition

5) Elevation of N concentration in nearby aquatic systems because of N leaching from soils, resulting in mountain lakes becoming acidic and/or eutrophicated.

The chaparral and forested areas of Southern California near Los Angeles are characterized by high base saturation, low C:N ratios, and coarse-textured soils and receive 20-35 kg N ha^{-1} yr^{-1} and are showing high net nitrification rates with N cycles strongly dominated by NO_3^- (Fenn et al., 2003 b). This could have a detrimental effect on competition for flora that favor the uptake of NH_4^+ as opposed to NO_3^- , and could ultimately lead to a monoculture of organisms (Bowman et al., 1993; Vitousek et al., 1997).

The Colorado Front Range which receives 3-5 kg N ha^{-1} yr^{-1} is showing signs of higher N mineralization among old-growth Engelmann spruce (*Picea engelmannii*) forests on the eastern side compared to the western side, which receives lower inputs of N (Baron et al., 2000; Rueth et al., 2003). This increase in the availability of N could affect growth and biomass allocation for some plants.

In a N-saturated system, seasonal patterns of microbial community abundance and activity may be altered. In an alpine ecosystem, soil microbial communities typically mineralize N during snowmelt, before the flora becomes active, thus playing a key role in the N cycle of the ecosystem. Net mineralization by microbes declines just after snowmelt when soils are saturated with water and plants begin to grow (Brooks et al., 1996). Nitrogen availability peaks in July and August with a decline in microbial biomass during the growing season and plants take in up to 93% of available N (based on ^{15}N tracer studies). Mineralization then peaks again just after plant senescence resulting in a pool of available N (Jaeger et al., 1999; Lipson et al., 1999). In a N-saturated system, the microbe populations do not decline during seasonal changes nor do they shift dominant mineralization/nitrification communities (Peterjohn et al., 1996; Jaeger et al., 1999).

Rapid discharge of N as a sign of N-saturation remains a topic that needs further investigation as NO_3^- leaching responses may vary among communities and ecosystems. Magill and co-workers (2000 and 2004) published the results from a 15-year study conducted in the Harvard Forest of Massachusetts intended to investigate the effects of N-saturation and accelerated N leaching as a system becomes saturated. Ammonium-nitrate was added chronically to the forest floor in hardwood and pine plots in a low, medium, and high regime (5-15g N m^{-2} yr^{-1}) over a 15-year period. They found two interesting results. First, red pine forests (*Pinus resinosa*) have a lower ability to retain N than hardwood forests (such as oak, birch, maple, black cherry, and beech). As a consequence, mean soil solution concentrations were around 20 mg-L^{-1} in the pine forests compared to 10 mg-L^{-1} in the hardwood forest, even though both communities received the same amount of fertilization. This demonstrates how different communities and ecosystems respond to excess N and N leaching. Second, dissolved organic nitrogen (DON) in the high fertilization plots was significantly higher than in the control plots. Roughly 35% of the dissolved inorganic N (DIN) was recovered in the high fertilization plots as leachates. The rest of the missing N was explained by incorporation into soil organic matter or immobilization by soil microbial communities. Formation of DON from DIN is an important mechanism for N assimilation to the soil system (Dail et al., 2001). Looking at ratios of DON and DIN in solution could be a way of assessing NO_3^- leaching potential because tree roots, foliage, and wood in all of the plots showed little ability to retain N anymore. The high fertilization plots also showed an increase in tree mortality of 56% and biomass accumulation had stopped altogether.

Denitrification can be stimulated by an overload of N. A study by Mohn and co-workers (2000) conducted in Switzerland demonstrated that soil redox potential, soil temperature and pH were all controlling factors that affected denitrification. Elevated N to these soils caused an increase in all of the above factors, which caused denitrification to increase from 1.7 to 2.9 kg N $ha^{-1}yr^{-1}$. They noted that the increases in denitrification were most intense with rain events carrying atmospheric N deposition.

High N concentration in soil water causes N to leach from terrestrial systems into nearby aquatic systems resulting in eutrophication and acidification. High concentrations of NO_3^- (compared to background levels of NO_3^-) have been found in streams of the Colorado Front Range, montane watersheds in Southern California, and in parts of the southwestern Sierra Nevada Mountain Range (Fenn et al., 2003 a). High-elevation lakes in the Colorado Front Range and many other places in the Rocky Mountains are showing signs of lower pH. This is due to N-saturated alpine systems located at higher elevations leaching N into watersheds that drain into these lakes (Baron et al., 1991, 1994). These same high elevation lakes are exhibiting large changes in diatom communities which have become an ecological marker for eutrophication (Baron et al., 2000).

N Deposition on Alpine Ecosystems

N deposition can affect ecosystems in different ways because plant and soil communities in these different ecosystem types may vary in their response to changes in nutrient availability (Seastedt and Vaccaro, 2001) depending on substrate type, topography, hydrology, or successional stage (Güsewell et al., 2002). Alpine ecosystems are among the most sensitive to chemical and climatic change because they have short growing seasons, strong seasonal variation in moisture and temperature, shallow and poorly developed soils, variable terrain with an abundance of different microclimates, sparse vegetation, and low overall productivity. These ecological factors naturally limit the uptake of N by plants (Baron, 1992; Fisk et al., 1998; Burns 2003). Alpine ecosystems are therefore typically affected by atmospheric N deposition earlier than ecosystems at lower altitudes (Rusek, 1993). Fenn and co-workers (2003 a and b; Fenn and Poth, 2004) observed that ecosystems with canopies tend to 'catch' atmospheric pollutants and retain them in their canopy. Since alpine ecosystems occur above the treeline, there is no canopy interception, resulting in more N deposition input directly to the soil. Alpine ecosystems are already predisposed to poorly retaining N because of very low net primary productivity (NPP) (Williams and Tonnessen, 2000) making them that much more sensitive to the direct effects of atmospheric N deposition.

Soil and snow accumulate in alpine zones on the leeward side of any kind of obstruction to the wind such as buolders or depressions (Rehder, 1976). Since atmospheric N tends to follow weather patterns and prevailing winds, N that deposits with winter snowfall will tend to follow the same topographic deposition patterns as snow accumulation. Leeward areas have higher soil depth because wind blows and deposits dust, debris, and soil in the depressions. Leeward areas likewise hold much more snow than more exposed areas. In places where there is more moisture from snow accumulation, there is typically more N deposition. These topographic patterns result in N-rich sites where plant and soil microbial communities may already be altered or shifted. Exposed areas in the alpine are typically drier and have very limited plant cover.

Topography controls where the moisture is found in alpine ecosystems, and moisture plays a major role in the presence or absence of plants and microbes (Fisk et al., 1998). Thus, N cycling and responses to excess N are expected to be different in dry versus wet alpine sites (Isard, 1986; Bowman, 1992; Fisk et al., 1998).

N Deposition in the Rocky Mountains

The Rocky Mountains make up much of the Intermountain West of the U.S. and are found as far south as New Mexico and extend as far north as northern British Columbia in Canada, covering some 4,800 kilometers (3,000 miles). The location of the alpine life zone depends on the Earth's latitude (Körner, 2003), and in the Rocky Mountain Range, it is typically found at or above 2745-3700 meters (9,000-11,500 feet).

The U.S. Geological Survey (see Ingersoll et al., 2008; 2009) is currently conducting snow pack surveys at various key locations during peak snow pack times to assess and model atmospheric deposition in the Rocky Mountain region, including NO_3^- and NH_4^+. The atmospheric deposition data thus obtained has formed the basis for modeled GIS maps that graphically show how atmospheric deposition is spatially distributed across different locations in the Rocky Mountain region (Nanus et al., 2003). The GIS maps show the "hot spots" of high N deposition entering the system as snow. One such "hot spot" of N deposition is found within the Niwot Ridge in the Rocky Mountains in the Colorado Front Range which gets as much as 9 kg N $ha^{-1}yr^{-1}$ of N deposition (Bowman and Steltzer, 1998; Nanus et al., 2003). Though this is not a high amount compared to the eastern U.S., it is enough to call this system "critically loaded" (defined as N deposition amount above the threshold levels that negatively affect natural resources). Williams and Tonnessen (2000) created an N deposition model for the Rocky Mountains and suggested a guideline for a “critical load” for these systems around 4 kg N ha^{-1} yr^{-1}.

N Cycling and Soil Responses at Niwot Ridge, Colorado

The Colorado Front Range of the Rocky Mountains is arguably the most affected by atmospheric N deposition for the Rocky Mountains because the majority of Colorado’s population resides there creating a large metropolitan area (Baron et al., 2000). However, the Colorado Front Range is also the most studied area for the effects of N deposition than any other part of the Rocky Mountains. The Institute of Arctic and Alpine Ecology was established in 1952 by The Mountain Research Institute and University of Colorado and includes a research station at Niwot Ridge. Studies on N deposition in alpine ecosystems of the Rocky Mountains started in the 1990’s and continue today (Ives, 1980). The LTER at Niwot Ridge studies patterns and controls of nutrient cycling, trace gas dynamics, plant primary productivity and species composition, geomorphology, and paleoecology in order to track global climate change (Benedict, 1968; Mancinelli and Keigley, 1983; Bowman et al., 1995; Rusch and Sievering, 1995).

Land masses in the alpine zone that develop a consistent snow cover early in the season tend to remain unfrozen during the winter because the snow acts as an insulating blanket. These snowpacks have high rates of microbial N immobilization because the soil microbes remain active during the winter months, so available NO_3^- in the soil is minimal during early spring snowmelt, and available NH_4^+ is mostly immobilized (Brooks et al., 1996; Edwards et al., 2007). However, land masses with low or inconsistent snowpack freeze during the winter and the microbes go dormant, resulting in an abundant amount soil of NO_3^- during early spring snowmelt (Brooks et al., 1998). A system with too much N will show equal amounts of available soil NO_3^- regardless of snow pack accumulation (Aber et al., 1989; Aber, 1992; Stoddard, 1994; Peterjohn et al., 1996).

Snow packs in the alpine zone hold up to 80% of annual precipitation and up to 50% of annual N deposition (Edwards et al., 2007). Inorganic N from anthropogenic N deposition and natural cycles is stored in snow in the winter and during early snowmelt there is a net ionic flux input of NO_3^- into aquatic systems. The first 25% of the snowmelt has higher concentrations of N than later during the growing season when plants come out of dormancy. This is because the NO_3^- trickles down through the snow pack where it concentrates at the bottom and flows out via snowmelt into the groundwater. Stream chemistry may show NO_3^- leaching into aquatic systems from terrestrial systems during early snowmelt which happens just before the growing season that lasts until early June as a result of percolation of meltwater into soil systems. N uptake by plants during snowmelt constitutes 12% of season-long uptake for graminoids and 7.4% for forbs (Bowman,1992; Bilbrough et al., 2000). As plants come out of dormancy the ecosystem has a high retention of N and the snow pack is depleted of N by up to 70% by mid-June (Reddy and Caine, 1990; Bowman, 1992). However, when alpine ecosystems are continually receiving atmospheric N deposition, this could change the spatial heterogeneity of the snow pack N supply (Bowman, 1992).

Total N storage in the alpine ecosystem at Niwot Ridge is 7000-8000 kg N ha^{-1} where 85-95% is stored in organic matter. N storage in living plants account for 1-2% of total N with the majority represented as root biomass. Less than 1% of N storage is in microbial biomass with up to 5% of microbial N turnover per day (Fisk et al., 1998). Turnover of N in alpine systems is a result of topographic heterogeneity in which soil moisture content, NPP, and microbial processes are all affected. Topography dictates where soil moisture is found. As a result plant biomass, NPP, and N allocation are higher in wet sites (where snow accumulates) than in dry sites (where snow does not accumulate). In dry sites, root biomass accounts for the main form of NPP and N allocation. Microbial processes such as respiration and N transformation are higher in wet sites than in dry sites of alpine ecosystems. However, gross N mineralization and immobilization patterns are similar in wet and dry sites suggesting that substrate quality or quantity does not contribute to differences observed in N availability and cycling among wet and dry communities. This further supports the idea that it is topographic and climatic heterogeneity, rather than substrate quality or quantity that dictate where N pools and communities are found throughout alpine ecosystems (Fisk et al., 1998). A N-saturated system could change this dynamic where both wet and dry communities receive higher amounts of N that could shift alpine plant productivity from N limitation to P limitation (Burns, 2003) and topography will no longer dictate where N pools are found.

Arctic tundra and alpine ecosystems are different as far as plant communities go. Soil microbial communities in these different ecosystems are very similar in that several of the same species of bacteria are ubiquitous and commonly distributed among different ecosystems and soil types than many other organisms (Mishustin, 1975; Anderson, 1977; Mancinelli, 1986). The break-down of plant litter by these microbes is heavily influenced by plant species, soil type, pH, climate, temperature, moisture, and microbial density (Witkamp, 1966; Ljungholm et al., 1979; Howard and Howard ,1980; Mancinelli 1986).

Soil microbial communities of alpine ecosystems exhibit seasonal cycling reflected in changes in biomass and community compositional shifts (Lipson et al., 2002). Overall, microbial biomass generally tends to decrease initially after spring snowmelt, fluctuate dynamically in the summer growing season, then increase steadily in the fall and peak in the winter just before snowmelt under snow packs (Lipson et al., 1999; Ley et al., 2004). However, some microbial populations such as N-fixing microbes are high in the spring when there is a high demand for N, and low in the fall when there is a low demand for N (Mancinelli, 1984). In the winter, microbial communities shift to microbes that utilize different enzymes than summer microbes (Lipson et al., 2002). Lipson and co-workers (1999) observed that there was a pulse of extractable soluble protein in the soil just after snowmelt, which was shortly followed by a peak of NH_4^+ and amino acids, suggesting a decline in microbial biomass and an increase in plant available N. They concluded that proteins, enzyme activity, and amino acids in the soil represent a peak of available N that is linked to the decline of microbial biomass and a shift in microbial communities just after snowmelt.

During the summer months in alpine ecosystems, soil microbial biomass exhibits a general decline compared to winter and spring biomass. There are several explanations for this decline in microbial biomass such as: protozoal grazing, community shifts due to substrate changes, temperature changes, and shifts in moisture regime (Lipson et al., 1999). Moisture could be the major controlling factor for this decline in biomass as water greatly influences diffusion and availability of soluble compounds (Stark and Firestone, 1995).

During the winter, microbial communities in the alpine soil are more abundant under consistent snow packs that insulate and allow the soil to remain unfrozen (Fisk et al., 1998). Winter freeze-thaw cycles in inconsistent snow-covered areas reduce bacterial populations and diversity significantly by disrupting cell structure (Mancinelli, 1986; Lipson and Monson, 1998). Spore-forming bacteria dominate variable snow-covered areas during this time of year because they can withstand the harsh winter conditions better than non spore-forming bacteria (Mancinelli, 1986).

Net mineralization of organic N from the organic matter left from senescent plants tend to be much higher during the snow covered period (2-6 g N m^{-2}, Brooks et al., 1996) than during the growing season (1.2 g N m^{-2}, Fisk and Schmidt, 1995) suggesting microbes are utilizing labile C sources during seasonal snow cover. This net mineralization of N creates a large pool of inorganic N in the soil just before snowmelt suggesting these microbe communities play a vital role in available N to plants for the growing season (Brooks et al., 1996).

Chronic N input to ecosystems such as anthropogenic N deposition, leads to an overall decrease in ecosystem function. Soil microbial communities respond to excess N by a significant decrease in activity and biomass (Lipson and Monson, 1998; Lipson et al., 1999; Schmidt et al., 2004) leading to slower rates of organic matter breakdown, decreases in some ecosystem functions (such as declines in mychorrizae/root symbiotic reliationships), and microbial dominant community shifts (Nemergrut et al., 2008). Mancinelli (1986) found that nitrifier and nitrogen-fixing microbe populations were significantly inhibited with increased NO_3^- in the soil, while actinomycete populations increased. Shifts in microbial species composition such as these may lead to a positive feedback loop of higher rates of NO_3^- leaching from the ecosystem. The dominant microbial communities, together with excess N, will result in greater N mineralization and nitrification rates. As these populations grow in biomass with excess N, so does the demand for carbon. A higher turnover of organic N in the soil releases more inorganic N to the soil matrix resulting in additional soil N on top of the atmospheric N deposition (Bowman and Steltzer, 1998; Steltzer and Bowman, 1998).

Shifts in microbial biomass from N deposition seem to be governed by seasonal changes. The negative effects of excess N appear to be most prominent during the summer growing season when competition is high and communities are more vulnerable to decline. There is a tendency to sequester N in microbes in the fall and winter as plants senesce and competition between the two decreases. However, this is an unlikely long-term sink for N because excess N decreases microbial biomass too much and it decreases the rate at which microbial biomass can respond to C inputs (Fisk and Schmidt, 1996; Schmidt et al., 2004).

There are no studies that demonstrate what will happen to alpine floral communities if microbial communities shift in response to excess N. Plant/microbe competition and interactions may change significantly causing the alpine ecosystem to decrease in nutrient cycling functionality, therefore compromising the ecosystem's abilities to sequester N and C properly.

Plant Responses to N Deposition at Niwot Ridge

One hypothesis for plant community responses to increases in resources is that resource-poor communities typically respond by increasing species diversity or species richness, and a resource-rich community responds by a decrease in diversity due to competitive exclusion for light (Goldberg and Miller, 1990; Bowman, 2000; Seastedt and Vaccaro, 2001). Declines in NPP, fine root biomass, foliar biomass, and increases in NO_3^- leaching are predicted signs of N-saturation in alpine ecosystems of the Colorado Front Range as they move from the current stage of 1, to stages 2 and 3 in Aber's N-saturation model (Aber et al., 1989; 1998; Burns, 2003). Gough and co-workers (2000) conducted a study across a wide variety of plant communities in several different life zones including the alpine zone to see how they would respond to N additions. They found an overall uniform decline in species richness in each community, irrespective of inherent productivity.

There are several mechanisms of how alpine plants take up, utilize, and store N in order to compete and coexist with the rest of the plant community. A unique aspect about alpine ecosystems is that when observing community abundance and dominance structure, the rate of NO_3^- uptake is not as significant as differences in allocation to roots, shoots, and biomass production. Alpine plants with more N dedicated to roots, shoots, and biomass are more abundant than those plants with less N allocated to the same areas. However, with increased soil N, this seems to be changing. Rare plants show higher rates of N uptake than dominant plants. This may be a mechanism for rare plants to coexist with dominant species keeping them from going extinct. Dominant species have much slower N uptake rates, but have higher root:shoot ratios and higher overall biomass, resulting in higher abundance and frequency. This suggests that dominant species can utilize and allocate N much more efficiently than rare species giving them a competitive advantage (Theodose et al., 1996). In a N-saturated system, this interaction may change N uptake mechanisms which may help one plant species to increase in biomass and abundance, resulting in community and dominance shifts (Rusek, 1993).

Bowman and co-workers (1993) compared wet and dry alpine meadows at Niwot Ridge for nutrient constraints on NPP. He found that dry meadows responded positively to N fertilizations, suggesting they were N-limited. The wet meadows showed signs of both N and P limitations suggesting that when N is abundant, then P becomes the limiting factor. Bowman concluded that the wet sites may be exhibiting levels of N close to saturation induced by P limitations. He also noted that graminoid biomass increased significantly while forbs biomass decreased significantly with treatment in the wet plots, further supporting the hypothesis that light competition becomes a limiting factor when resources are abundant. Theodose and Bowman (1997) conducted a similar study that also concluded that increases in N from anthropogenic sources show the highest impact on limited communities such as dry alpine meadows, by changing the composition of the plants and favor the invasion of more competitive species such as grasses. These shifts in species composition may be already occurring in alpine tundra systems in central Europe (Rusek, 1993).

In summary, as N increases, diversity and biomass increase initially, then diversity decreases as opportunist plants invade and increase cover, then dominance shifts to species able to utilize the excess N (Tilman, 1982; Begon et al., 1990; Bowman et al., 1993).

Alpine plants have developmental constraints, such as preformation of next year's buds during the growing season; and cellular development constraints, such as stunted growth in order to cope with the harsh environment typical of alpine environments. These may be mechanisms that evolved to deal with fluctuating nutrient concentrations in the soil (Bowman et al., 2003). Alpine plants tend to preferentially allocate N to storage before foliage (Chapin, 1980; Körner, 1989; Bowman and Conant, 1994) and are capable of sequestering and storing N in aboveground foliage only when soil N pools are increased (Chapin et al., 1990). Several species of alpine plants store N in belowground biomass and only when soil N increases sufficiently, will they allocate N to foliage and even start to exhibit luxury consumption (Jaeger and Monson, 1992; Lipson et al., 1996; Mullen et al., 1998). Therefore, foliar N and P concentrations do not reflect immediate soil N availability and cannot be used as a significant marker for ecosystem N-saturation (Bowman et al., 2003). Root:shoot ratio and overall biomass may be a better indicator of plant responses to increased N rather than foliage alone. However, examining plant biomass alone does not accurately reflect what the entire ecosystem is going through. Looking at both plant biomass and soil characteristics may better describe how N is moving through the ecosystem.

N Deposition and Acidification Effects on Watersheds and Aquatic Systems

Precipitation has increased by 300 mm per year from 1967 to 1996 in the alpine tundra of Niwot Ridge, and N deposition has nearly doubled from 1989-1992 (Williams et al., 2003). Seasonal snowmelt releases a pulse of NO_3^- ions that have accumulated in the snow pack over winter into the ecosystem that acidifies aquatic systems by lowering the acid neutralizing capacity (Williams et al., 2001). There is a positive feedback of N deposition in that it causes net mineralization and nitrification to increase, resulting in more loss of NO_3^- into aquatic systems on top of the seasonal NO_3^- flux, further acidifying these systems (Bowman and Steltzer, 1998; Williams and Tonnessen, 2000).

Talus slopes are exposed rock and gravel slides in alpine ecosystems with little snow pack to cover them. They have little vegetation to retain N and allow rapid precipitation infiltration and movement to surface waters. Talus slopes are considered important sources of NO_3^- and are therefore associated with higher NO_3^- concentrations in surface waters draining from them (Williams and Brooks, 1997).

Lake sediment cores from three aquatic sites along the Front Range show a long-term shift in diatom communities that is consistent with increased atmospheric N deposition since around 1939 (Wolfe et al., 2001). This shift in species has been correlated to agricultural practices and it has been shown that the diatom species respond quickly to N additions (McKnight et al., 1990).

Acidification of lakes and streams is a possible explanation for the decline of several Rocky Mountain amphibian species such as the tiger salamander (*Ambystoma tigrinum*), boreal toad (*Bufo boreas*), and the chorus frog (*Pseudacris triseriata*) (Harte and Hoffman, 1989; Carey, 1993; Kiesecker, 1996). Harte and Hoffman (1989) did an in situ study in the Elk Mountains of Colorado where they exposed tiger salamander eggs to different pH conditions in a body of water that is known to have seasonal fluxes of acidification from NO_3^- leaching in the early spring (when salamander eggs are present). They found that at a pH of 5.6, there was a 100% mortality rate of the zooplankton *Diaptomus coloradensis,* which was a major food source of the aquatic food web. While salamander eggs were not directly affected by the low pH, a drastic decrease in resources from the altered pH caused salamander mortality to increase significantly.

N Deposition in Grand Teton National Park, Wyoming

Most studies on alpine ecosystems of the Rocky Mountains come from the Colorado Front Range and specifically from Niwot Ridge. Though these studies have been very important in demonstrating how an alpine ecosystem is affected by N deposition, they do not fully represent the entire Rocky Mountains. There are few studies that compare N deposition in ecosystems of the Intermountain West. The flora, soils, and geology vary from region to region throughout the Rocky Mountains (Habeck, 1987; Burns, 2003) so N deposition may affect these ecosystems differently.

The flora of alpine ecosystems across the Rocky Mountains have similarities in that the species are similar based on physiognomic and species characteristics. They also differ because of dispersion selection based on barriers such as the Wyoming Desert that separates the Colorado Rockies from the Wyoming Rockies. The Teton Mountain Range has vegetation characteristics more in common with the mountain ranges found in Montana, Idaho, and Southern Canada than it does with the mountain ranges found in Colorado, Utah, and New Mexico based on these dispersal barriers (Hadley, 1984). The dissimilarities are not large enough to disallow a comparison between Wyoming mountains and Colorado mountains because the Wyoming Desert and Colorado Plateau had near-periglacial Pleistocene conditions which acted as a corridor for dispersal of alpine species resulting in modern day island biogeographic regions in the alpine communities of the Rocky Mountains (Mears, 1981). The differences and similarities in physiognomy of the mountains of the Rockies make each one of them unique study areas; therefore more studies in different places of the Rocky Mountain Range may contribute a better understanding of the Rockies as a whole, compared to intensive studies of a few sites. Since most atmospheric N deposition studies occur at Niwot Ridge along the Colorado Front Range, our study used comparisons of the Colorado Front Range to the Teton Mountain Range with the physiognomic differences in mind because there are few N deposition studies elsewhere in the Rocky Mountains.

The state of Wyoming has only 8 National Atmospheric Deposition Program (NADP) monitoring stations. The Grand Teton National Park (GTNP) is located in the northwest corner of the state. The closest deposition monitoring station is in Yellowstone National Park, 50 miles to the north and located at low elevation. Consequently, the station at Yellowstone National Park may not accurately reflect N deposition at high-elevation alpine sites in GTNP. The USGS has 5 snow survey sites located throughout the Grand Teton Mountains located at Rendezvous Mountain, Garnet Canyon, (both located inside GTNP), Teton Pass, Togwotee Pass, and Four Mile Meadow (located outside the boundaries of GTNP). These sites may be slightly more accurate for studying N deposition in GTNP. However, the Rendezvous Mountain site is the only site located at a high enough elevation to yield usable data for alpine N deposition.

Monitoring N status in “arid” alpine soils of the Rocky Mountains

Monitoring the status of N in soil is a very complex issue because N is an essential nutrient for all life forms (Fenn and Poth, 2004) and the N cycle depends on several biotic and abiotic factors such as N pools and distribution (Bowman, 1992), plant function, microbial activity (Lipson et al., 1999), and soil properties such as temperature, moisture regime, and pH.

Methods to determine soil N status consists of static and dynamic measurements. Static measurements are usually a "snapshot" in time of soil properties, such as C and N pools, C:N ratios, or KCl extractable inorganic N (Jaramillo et al., 2003). Dynamic measurements of soil N status integrate N releases over a period of time either in the field or in the laboratory under more controlled conditions. This usually entails placing probes or resin bags in the field for periods of time, use of ^{15}N isotope tracers in soils, or conducting field or lab incubations with soil samples to monitor N mineralization and nitrification rates (Sala et al., 2000; Hanselman et al., 2004).

Assessing N-saturation or excess N typically involves looking at indicators such as high levels of nitrate reductase activity of enzymes (Norby et al., 1989; Magill and Aber, 1998), accumulation of N or amino acids in plant foliage (Ohlson et al., 1995; Stams and Schipholt, 1990), increased emissions of N trace gases from soil (Castro et al., 1994; Sitaula et al., 1995), high soil NO_3^- concentrations and NO_3^- leaching rates, and increases of NO_3^- concentrations in stream water (Riggan et al., 1985; Aber et al., 1989; Stoddard, 1994).

Since there are so many factors involved in determining soil N status, it is relevant to consider geographic and climatic variables during the examination of any soil study. Fenn and Poth (1998) created a list of ecological indicators of N status for the arid soils of the western U.S. The Intermountain west of the U.S. (which includes alpine soils of the central Rocky Mountains) is considered arid based on general precipitation patterns and lower moisture availability inland compared to coastal (Nagy and Grabherr, 2009;

Körner, 2003). Alpine zones in the Rocky Mountains can indeed be considered relatively dry. As the Rocky Mountain range typically receives less moisture and more radiation than other mountain ranges in the western U.S. Within the Rocky Mountain Range, the Central Rockies have the least amount of humidity relative to the maritime effects on the Northern Rockies, and the Gulf moisture influencing the Southern Rockies (Kittel et al., 2002). Another significant factor is that topographically high places in alpine systems (as opposed to the depressions that retain snow pack) occur on the tops of watersheds where water tends to flow off the mountain and water reserves tends to be low due to shallow soils (Burns, 2003). This leaves roughly half of the alpine ecosystem drier than lower ecosystems that have perennial flows and that are rich in organic matter and are characterized by more soil development. For this reason, alpine ecosystems are often compared to desert and arid environments (Körner, 2003).

The list of potential ecological indicators for soil N status summarized by Fenn and Poth (1998) includes:

- Soil C:N ratios
- NO_3^- : NH_4^+ ratios in soil
- Soil NO_3^- (in soil solution or in soil extracts)
- Nitric oxide (NO) emission levels from soil
- N-mineralization and nitrification rates
- Foliar nitrogen:phosphorus (N:P), C:N, and N:cation ratios
- Accumulation of NO_3^- in foliage of understory and overstory species
- Plant response to N fertilization
- Streamwater NO_3^-concentrations and fluxes

- Base saturation and soil pH

Because of the extensive prior research at Niwot Ridge, several ecological indicators of N cycling in alpine soils have been used in the Rocky Mountains. These show several similarities to Fenn and Poth's (1998) list:

- Inorganic N concentrations in the root zone of the soil (Bowman et al., 2006)
- Changes in vegetation species composition and abundance (Phillippi et al., 1998)
- N in aboveground biomass vegetation (Bowman et al., 2006)
- Measure of N cycling in soil
- Use of N^{15} tracers to evaluate N sinks and plant uptake of N
- N leaching from system (Binkley and Vitousek, 1989)
- Calculations of critical N load to vegetation species and communities that have been found to be sensitive to changes in N through the use of N amendments (Fenn et al., 2003 a and b; Bowman et al., 2006).

This study has taken into consideration most, but not all of the above ecological indicators of N status for alpine soils. Measurements such as NO emissions, NO_3^- levels in soil and stream water, and use of N^{15} tracers were not considered because of financial and logistical constraints.

Objectives

Our project focused on one of the modeled "hot spots" in the Rocky Mountain Range, Grand Teton National Park, that receives between 0.1-2.5 kg N ha^{-1} yr^{-1} of N deposition (Nanus et al., 2003).

The objective of this project was to evaluate the effects of atmospheric N deposition on alpine plant communities and soils of Grand Teton National Park. This study investigated potential indicators of changes in N status (i.e. early warning signs). There is evidence from prior research in the Rocky Mountains that responses by alpine vegetation to increased N input not only depend on current N status, but may also be affected by environmental conditions such as snow distribution, microclimate, and wet vs. dry soil conditions (Brooks et al., 1996; Fisk et al., 1998). If we can establish early warning signals of ecosystem change by examining soil functions and plant community structure (i.e. change in function), then it might be possible to avoid the negative effects of excess N before expensive restoration is necessary.

Specific study objectives for my master's thesis are:

1) Assess N deposition input along a presumed deposition gradient from north (high) to south (low) based on the maps from Nanus et al., (2003).

2) Assess the current N status in the alpine ecosystems of Grand Teton National Park through a series of soil and plant indicators.

3) Examine the edaphic conditions of each site and assess their influence on N status.

4) Assess the influence of the N deposition regime and edaphic conditions on plant and soil characteristics such as: community composition, aboveground and belowground biomass, and changes in soil N status to see whether soil indicators are more responsive than vegetative indicators.

My research project will address the following hypotheses:

Hypothesis 1: N Deposition Gradient: N deposition follows a north to south gradient, with the highest N input at the north end of the park (i.e. Moose Basin) and the lowest N input at the south end of the park (i.e. Rendezvous Mtn.) as proposed by the Nanus et al., (2003) maps (resolution 1km).

Hypothesis 2(a): Soil Responses: Soils in the high N deposition sites are already exhibiting changes in N status by having more available N, higher nitrification potential, and lower C:N compared to the low N deposition sites.

Hypothesis 2(b) Plant Responses: Plants in the high N deposition sites are responding to changes in N status compared to plants located in the low N deposition sites. They will be characterized by higher aboveground and belowground plant biomass (i.e. foliage biomass and root biomass), more N in foliar and root content, and lower species richness and composition in high N deposition locations.

Hypothesis 3: Soil Microclimate: The effect of N deposition on plants and soil characteristics is influenced by edaphic conditions such that wet edaphic conditions (that capture more snow and remain wet throughout the year) will behave as N rich sites compared to dry edaphic conditions (such as topographic areas where snow does not accumulate). This will be reflected in higher N concentrations, less plant species, higher nitrification potential (due to more microbes found in wet soils), and lower C:N in aboveground and belowground biomass.

CHAPTER 2

METHODS

Physiography

The Rocky Mountain Range spans a distance of 4,800 kilometers starting in New Mexico and ending in the northern most region of British Columbia in Canada. The United States is divided into different physiognomic regions based on terrain, soil texture, rock type, geologic structure, and history, with the Rocky Mountain Range being among one of the physiognomic regions (Fenneman, 1931). The Teton Mountains are part of the Rocky Mountain Range and are located in the “central Rockies” region (Fenneman, 1931). They represent the eastern most extent of several Pacific Northwest flora making it a corridor for plant dispersal.

The Teton Mountains are located in the northwest corner of the state of Wyoming (Fig. 2.1). They reside within Grand Teton National Park on the eastern slope and the Jedidiah Smith Wilderness area within the Caribou-Targhee National Forest on the western slope of the Teton Range. Yellowstone National Park is to the north of the Teton Mountains and the city of Jackson Hole, Wyoming, is located to the south of the mountain range.

Climate

The Teton Mountains have a north-south orientation making them perpendicular to the prevailing westerly winds. Wind flow patterns indicate that the state of Wyoming is covered by air from the Pacific most of the time with a small percentage of cold air from masses that move down from Canada (Curtis and Grimes, 2004). Storm systems typically come down from the northwest dropping precipitation on the north end of Grand Teton National Park first, then the south end. The Teton Mountains represent an orographic barrier that creates a rain shadow with much of the precipitation falling on the western slopes and creating semi-arid conditions on the eastern slopes.

The Wyoming climate is semi-arid. Precipitation during the growing season averages 250 mm and comes in spring and early summer, mostly as afternoon thunderstorms that develop daily. The alpine zone of the Teton Range receives over 7 meters of snow annually. Average maximum temperature ranges around 16°- 18° C, with average summer lows in the range of -1°and 4.5° C and occasional freezes. Temperatures rarely exceed 37.8° C above 1830 m. Winter temperatures above 2900 m range from -32° to -20.5° C.

Geology

Thousands of 7-7.5 magnitude earthquakes over the last 13 million years on the Teton fault caused the crust of the earth to break and rise up, forming the Teton Mountains. The Teton fault is 65 kilometers long from north to south, and 24 kilometers wide from east to west. It lies on the eastern slope of the Teton Range which explains the abrupt, steep eastern face of the Tetons and the long, gentle slope of the western face of the Tetons. The east side of the Teton fault rises up while the west side of the fault sinks and forms the Jackson Hole valley (Smith and Siegel, 2000).

From 2 million to 300,000 years ago, Jackson Hole Valley was a convergence point for three major glaciers from the north, north-east, and west to form the Yellowstone ice cap that was 3200 m in elevation. Alpine glaciers formed in the canyons of the Teton Mountains and carved U-shaped valleys that deposited and transported sediment to the Jackson Hole valley. When the glaciers receded 14,000 years ago, massive flooding and fast erosion of Paleozoic and Mesozoic formations exposed deep layers of older bedrock and formed many glacial lakes (Good and Pierce, 1996).

The oldest rock formations that have been eroded and exposed from the processes of the last 13 million years occur highest on the Teton Range, and are estimated to be 2.4-2.8 billion years old. These rocks typically consist of gneisses, schists, and granites. Other rocks that form the Teton Mountains are around 2500 million years old and consist of precambrian limestones, various shales, metamorphosed sandstones, and interbeded volcanic deposits (Harris et al., 1997).

Study Sites

Three locations in Grand Teton National Park were chosen based on a modeled atmospheric N deposition gradient (High, Medium, Low) from Nanus et al., (2003) and GIS maps (1 km resolution). Sites were selected based on elevation between 2810-3070 m (difference of 260 m), slope less than 15º, and the following vegetation community types based on GIS maps (datum NAD 83) from resource management at Grand Teton National Park headquarters: alpine dwarf shrubland, dry graminoid upper elevation, herbaceous rockland slopes, herbaceous alpine wetland meadows, tundra-dry alpine, sparsely vegetated limestone pavement, and glacier/snow. The sites are: Moose Basin to the north (high N deposition), Paint Brush Canyon (medium N deposition), and Rendezvous Mountain to the south (low N deposition; Fig. 2.2; see appendix 1 for plot list and UTMs).

Within each location, study sites were established at contrasting edaphic conditions. Edaphic plots were selected based on visual assessment of snowpack, local topography, and vegetation type from the GIS maps (Fig. 2.3). Wet sites contained soils that were physically wet or moist to the touch throughout the summer, had a snow field draining into it, or were described on the GIS vegetation community maps as herbaceous alpine wetland meadows, or glacier/snow. They were typically low topographic depressions where snowmelt accumulated and soils were the most developed. Dry sites were chosen based on visual assessment of exposed areas with little snow accumulation and having vegetation described as dry graminoid upper elevation, herbaceous rockland slope, tundra, sparsely vegetated limestone pavement, and sparsely vegetated exposed hillside from the GIS vegetation maps. Dry sites were typically higher topographic areas with sparse vegetation and poorly developed, shallow soils.

Moose Basin (MB) is located at the north end of Grand Teton National Park at the north fork of Webb Canyon. The trail is an official park service trail and the trailhead is located at the Lower Berry patrol cabin on the northwest side of Jackson Lake. Moose Basin wet and dry plots are located adjacent to each other within 25 meters.

Paint Brush Canyon (PB) is located in the middle of the park between Cascade Canyon and Leigh Canyon. The trail is an official park service trail and the trailhead is located at the String Lake picnic area. The sites are located in the Upper Paint Brush Canyon below Paint Brush Divide. The dry plots are located down the first shelf when walking off the main Upper Paint Brush trail to the Grizzly Lake unofficial trail. The wet plots are located on the second shelf below in a wetland drainage area. The wet and dry plots are approximately 200 m apart.

Rendezvous Mountain (RDV) is located at the south end of the park, and is accessed by the tram at Teton Village owned and operated by Jackson Ski Resort. The wet plots for Rendezvous Mountain are located off the official trail and to the north along a drainage where the trail starts to ascend up to the western ridge. The dry plots are located just above the drainage on top of the western ridge off the trail and to the north. The wet and dry plots are approximately 200 m apart. GIS UTMs (NAD 83) for each plot were recorded and are summarized in the appendix table 2.1.

Soils

The soils of Grand Teton National Park are derived from limestones, a variety of shales, metamorphosed sandstones, and interbeded volcanic deposits as parent materials (Love, et al., 1989). The Tetons were formed 13 million years ago as opposed to the rest of the Rocky Mountains that were formed 65-100 million years ago. Therefore, the Tetons are typically craggy rock faces at higher elevations (Smith and Siegel, 2000). Some parts of the Tetons are made of quartzite bedrock which is highly resistant to erosion. The soils are considered “new” because of the geological “youngness” of the Tetons, geological parent material structure, elevation, climate, and glaciation periods.

Elevation and climate in the Tetons creates short growing seasons and severe cold and freezing. There is not enough time between freeze and thaw for parent material and organic matter to break down fast enough to create and develop pedogenic soil horizons except in collection basins and protected areas. Some areas are saturated with water during spring run-off and most of the growing season. This inhibits soil formation as well (Buol et al., 2003). Glaciation periods tilled sections of the soils and mixed them into a horizonless mixture (Mahaney, 1975).

Inceptisol/Entisol and Mollisol soil orders make up respectively 25-50% and 10-25% of the soils found in the alpine zone of the Tetons with 25-50% designated as rock outcrop. Mollisols form in semi-arid climates typically under grasslands and therefore are less relevant to this study. Inceptisols and Entisols are typically horizonless soils found on flood plains, delta deposits, or steep slopes where parent material has either recently accumulated, or the soil horizon formation has been limited because of constant shifting of the soils, or unfavorable climatic conditions. The parent material is typically limestone, wind-blown sand, or loess. This study only considers the top 15 cm of the soil layers because it is considered the rooting zone where most chemical reactions take place.

Moose Basin (MB) soils are part of the Rock outcrop-Sheege-Starman association and Midfork-Spearhead association (Soil Survey of Teton County, Wyoming, 1975; Young et al., 1982). The parent materials consist of calcareous sedimentary rocks that weathered and formed the soils via stream and slope alluvium. The pH of the MB soils were 5.93 for the wet plots, and 6.82 for the dry plots. Sheege soil series are very shallow and well drained soils that formed from limestone residuum and are found on mountainsides between 3-30% slopes. The Starman series is also a very shallow, well-drained soil that formed from limestone residuum. They are found on hillsides and mountain slopes with a slope range of 3-70%. A typical pedon of 0-10 cm of the Sheege-Starman association consists of a pale brown, very stony loam that turns dark brown when moist. It has a moderately fine granular structure that is soft, very friable, sticky, and slightly plastic that contains fine roots. It is 25% pebbles, and 20% cobble that is calcareous and moderately alkaline. At 10-20 cm, the Sheege-Starman association is a pale brown, very stony clay loam that turns dark brown when moist. It has a weak, fine, subangular blocky structure that is slightly hard, friable, sticky, and plastic. It is 25% gravel, 35% cobble and stone. It is slightly calcareous. At the eight inch mark, it is hard, fractured limestone. The Midfork Series is very deep, well-drained, and formed from calcareous alluvium and colluvium from deposited sediment from landslides. Midfork soil series are typically found on alluvial fans and mountainsides of a slope range of 15-70%. Spearhead series are also a very deep, well-drained soil formed in collivium, alluvium, and glacial till typically found on mountainsides with a slope ranging between 3-70%. A typical pedon of the Midfork-Spearhead association for the top 0-10 cm consists of a brown, very stony loam that is dark brown when moist. It has a moderately fine granular structure that is soft, very friable, slightly sticky and slightly plastic with many fine, medium, and coarse roots. The soil consists of 40% pebbles and 10% cobble and is mildly alkaline. The next 10-25 cm of the Midfork-Spearhead association consists of a brown very stony loam that is dark brown when it is moist. It has a mix of weak to medium subangular blocky structure and a weak medium fine granular structure that is soft, very friable, sticky, and slightly plastic. It has fine and coarse roots with mostly medium roots. It typically is 20% pebbles, 10% cobble, and 10% stone and is mildly alkaline.

Paint Brush Canyon (PB) sites have soil associations that consist of Rock outcrop-Teewinot-Moran soil series and a small amount of Rubble land. The parent materials of these soils consist mostly of granite and gneiss, that formed slope alluvium, colluvium, and till during the glaciation periods. The pH for PB was 5.74 for the wet plots and 6.21 for the dry plots. The Teewinot soil series consist of shallow, well-drained soils that formed from granite residuum. They are found on mountainsides with a slope range of 20-70%. Moran soil series consist of very deep, well-drained soils formed from colluvium and alluvium found on mountainsides with a slope range of 3-70%. A typical pedon of Rock outcrop-Teewinot-Moran for the first 0-38 cm is a very cobbly grayish brown sandy loam that is very dark grayish brown when moist. It has a moderately fine granular structure that is soft and very friable. It is 25% pebbles 10% cobble, and 5% stones having bedrock at 25-50 cm.

Rendezvous Mountain (RDV) soils are part of the Rock outcrop-Sheege-Starman association (as described for the Moose Basin Sites) and Starman-Owlcan association. The parent materials of these soils consists of calcareous shale and shallow, hard sandstone that formed in colluvial and alluvial outwash weathered from underlying bedrock. The pH for RDV was 6.34 for the wet sites and 6.65 for the dry sites. The Owlcan series consists of a very deep, well-drained soil formed from alluvium found on alluvial fans and mountainsides with a slope range of 30-70%. A typical Starman-Owlcan pedon usually has a couple of centimeters of forest duff on top, but this was not the case in the study plots. The first 0-2.5 cm is a dark grayish brown loam that is a very dark brown when moist. It has a moderate, very fine granular structure that is soft, very friable, slightly sticky, and slightly plastic. The next 2.5-13 cm is a pinkish grey loam that is dark brown when moist. It has a moderate and fine subangular blocky structure that is slightly hard, friable, sticky and plastic. The next 13-28 cm of the Starman-Owlcan association is a brown clay loam that is dark brown when moist. It has a moderate and fine subangular blocky structure that is slightly hard, friable, sticky, and plastic (Soil Survey of Teton County, Wyoming, 1975).

Experimental Design

Within each of the 3 contrasting N deposition sites (Fig. 2.2), plot pairs were established at 3 locations and on contrasting edaphic conditions (wet/dry, Fig. 2.3) yielding a 2-factorial design (3 deposition levels, 2 edaphic conditions) with three replications for a total of 18 plot pairs (36 plots total). The wet and dry plot pairs were located within 200 meters of each other (Fig. 2.3). Each plot measured 2.5 m x 2.5 m and was separated by a 1 m buffer zone where microclimate measurements were made. A one meter by one meter inner plot was set up within each 2.5 m x 2.5 m plot for the non-destructive observation of ecological community changes (Fig 2.4). Plot pairs consisted of an untreated control and an N addition plot. Treatments were not part of this study. Results from the 18 unrelated control plots were used in this study.

Measurements

This study assessed how alpine plant and soil communities respond to changes in N deposition by collecting information on N deposition, soil microclimate, vegetation, and soil properties (Fig. 2.4).

N deposition

The atmospheric N input in two of the three sites was estimated from snow pack surveys conducted at maximum snow pack level in conjunction with the USGS. Plot level measurements of atmospheric N deposition in wet and dry edaphic conditions were measured using ion exchange resin tubes (IERT see Fig. 2.6) that were deployed over summer (“Fenn collectors” Fenn and Poth, 2004) or an entire year (“Johnson collectors” Susfalk and Johnson, 2002; Johnson et al., 2005).

To estimate winter N input, snow pack surveys at the maximum snow pack time were conducted at Rendezvous Mountain (3/15/06 and 3/13/07) and Moose Basin (3/27/06) in coordination with George Ingersoll of the USGS. After a training session with the USGS, a team skied into MB and dug a snow pit, measured pH, snow water equivalent (SWE, cm), temperature, snow pack crystals, and packed a snow sample out that was sent to the USGS to be analyzed for H^+, Ca^{2+}, Mg^{2+}, Na^+, K^+, NH_4^+, Cl^-, SO_4^{2-}, NO_3^- (μeq L^{-1}), and DOC (mg L^{-1}). USGS personnel conducted the snow pack survey for RDV. There are no snowpack values for PB and MB 2007 sites because access to the sites was restricted due to avalanche danger.

To estimate summer N input, Fenn collectors (Fenn and Poth, 2004) were installed just outside the plots (two per plot with one blank per plot pair) (Figs. 2.4 and 2.6) to capture input of NO_3^- and NH_4^+ from wet and dry atmospheric N deposition during summer. The collectors were set out for a 70-90 day period (7/8/2006-10/3/2006, and 7/11/2007-10/9/2007) during the alpine growing season.

Fenn collectors (Fig. 2.6) consisted of 30.5 cm diameter funnels with 35.6 cm by 2.5 cm tube connected to the funnel that held 25-30 grams of Rexin® ion exchange resin that was 300 analytical grade, mixed bed, strong acid/base of medium porosity with mesh size of 16-50 and a total exchange capacity of more than 1.5 meq g^{-1}(Fisher Scientific, New Jersey). Blanks consisted of a sealed tube with resin inside that did not have a funnel associated with it. The blanks were set up in the field alongside the collectors to gather information about changes in resin (if any) under the influence of environmental factors.

To compare winter and summer N input from the snow pack surveys and the Fenn collectors to annual N input estimates, Johnson collectors (Susfalk and Johnson, 2002; Johnson et al., 2005) were installed just outside the plots (two per plot with one blank per plot pair) (Figs. 2.4 and 2.6). Johnson collectors gathered NO_3^- and NH_4^+ input from snowmelt, and wet and dry atmospheric N deposition over a one year period (7/8/2006-7/20/2007).

Johnson collectors consisted of a 5 cm by 17 cm PVC tube inserted into a coupler that had 18-20 g of Rexin® resin sandwiched between 2 sheets (approximately 6-8 cm^2, 20 µm grade) of Nitex® nylon bolting cloth (Wildlife Supply Company, New York). Two to three tent stakes were taped to the coupler to hold the collector off the ground which was left in the field for one year. Blanks consisted of a sealed collector device with resin inside placed alongside the collectors.

After retrieval from the field, the ion exchange resin from each collector was added to 100 ml of a 2 M KCl solution and placed on a shaker for one hour. The KCl solution was then filtered using Whatman 42 filters and sent to the Utah State University (USU) Analytical Laboratory for analysis of NO_3^- and NH_4^+ Using a Lachat Instruments© Flow Injection Analyzer QuikChem 8000.

Edaphic Conditions

To compare edaphic conditions among the sites, soil moisture and temperature measurements were taken in each plot pair. Soil moisture regime was estimated from ECH20 probes (Decagon Devices Inc©, Washington) installed at 10 cm soil depth within the one meter buffer zone between plot pairs (Fig. 2.4). ECH20 probes are flat 10 cm by 2.5 cm plastic probes that measure electrical conductivity and translate these into volumetric soil moisture content using factory-established calibrations. Readings were taken during each visit to the sites in 2006 and 2007 with a hand-held device (ECH20 Check soil moisture monitor) and expressed as percent moisture and millivolts. In addition, two data loggers (Em5b by Decagon$^{©}$) were installed in early summer of 2006 at one wet and one dry edaphic site at each of the three locations. They were programmed to record moisture data every 1.5 hours and data were downloaded in the field using a hand-held computer. Due to data logger malfunction, the only reliable data was obtained at the MB wet plot (7/10/2007-10/9/2007) and the RDV wet plot (7/24/2006-9/30/2006).

Soil temperature was measured with Tidbit data loggers (Onset$^{©}$ Corp. Bourne, Massachusetts) installed in early summer 2006 buried around 15 cm soil depth in the 1 m buffer zone (Fig. 2.4). The data loggers recorded soil temperature from 7/06/2006 to 7/20-2007 at 1.5 hour intervals and were downloaded in the field using an optical shuttle™ (Onset Corp. Bourne, Massachusetts) when sites were visited during the summer and fall of 2006 and 2007.

Soil N Status

To determine soil N status, three soil cores were taken in each plot using a PVC pipe that was 5 cm diameter by 17 cm length. The first of the three soil cores was used for bulk density, root biomass, and total C and N. The second soil core was used for extractable NO_3^- and NH_4^+ and soil moisture content. The third soil core was used for nitrification potential. Because soil depth was highly variable and soils did not always fill the core completely, three additional measurements were taken for each core to estimate soil volume sampled. The measurements were (see Fig 2.5): 1) ground level (GL): measurement of how far the core stuck out of the ground after the core was pounded into the ground. 2) TOP: the measurement of the actual soil inside the core to the top of the PVC. This measurement was to account for compaction from pounding the core into the ground and was taken BEFORE the core was removed from the ground. 3) BOTTOM:

Measurement of how far the soil stuck out from the bottom (if at all) of the core AFTER the core was removed from the soil. Based on these values, the soil volume sampled was estimated as:

Volume = (core length – GL + Bottom) x Core surface area.

For bulk density, one of the soil cores was dried at 65° C for 24 hours and weighed. The soil from the core was sieved into a coarse and fine fraction (<2 mm) after removing roots. A small sub-sample (1-2 g) of the fine fraction from each bulk density core was crushed with a mortar and pestle and sent to the USU Analytical Lab for determination of percent C and N (LECO Truspec® CN elemental determinator St. Joseph, Michigan). Percent C and N were converted into grams per m^2 for the top 10 cm of the soil based on the bulk density and volume of the soil cores.

Extractable inorganic N (NH_4^+ and NO_3^-) in the second soil core was determined by in situ KCl extractions using 100 ml of 2 M KCl. The soil core was homogenized in a container in the field and a subsample of 6-10 grams of field moist-soil was added to the KCl solution. The bottles of soil and KCl solutions were kept in portable coolers. Within 30 hours of arrival in the field laboratory, bottles were shaken for approximately one hour and filtered into 25 ml vials using Whatman 42 filter papers. Extracts were frozen until analysis for NH_4^+ and NO_3^- using a Lachat Instruments© Flow Injection Analyzer QuikChem 8000. All soil concentrations were expressed on a dry weight basis. The remaining soil from this soil core had large rocks manually removed and was weighed, then dried at 65° C for 24 hours, and weighed again to estimate gravimetric soil moisture content.

Mineralization and nitrification potential was determined on soil samples from the third soil core removed from each plot. Each soil core was kept cool in portable coolers until processed upon arrival in the lab. Roots and rocks where removed manually or with a 2 mm sieve. Four, 8-gram samples were placed in test tubes and incubated at 24° C for 30 and 60 days at field moisture content with two replicate samples per incubation period. Samples were aerated at mid incubation by opening the containers and allowing gasses to escape and oxygen to enter. After the designated incubation period, 40 ml of 2 M KCl was added to each test tube, and tubes were placed on a shaker for 45 minutes then allowed to settle for one hour. Samples were filtered with Whatman 42 filters and the extract was sent to USU Analytical Labs for NH_4^+ and NO_3^- analysis. Inorganic N concentrations from the field extracted soil samples were used as initial (t_0) values, and nitrification potential was calculated as the increase in soil NO_3^- concentration over the 30-day and 60-day incubation time. All rates were expressed on a soil dry weight basis.

Nutrient availability supply rate in the soil was assessed by installing PRS™™ (Plant Root Simulator, Western Ag Innovations, Canada) at 10 cm soil depth in each plot. The probes were small, flat ion exchange probes (3 cm x 15 cm) (Hangs et al., 2004), and each plot measurement was derived from eight probes per plot (4 anion and 4 cation exchangers). The PRS™ probes were deployed during two summers (from 7/8/2006 to10/3/2006, and from 7/20/2007 to10/6/2007) and one winter (from 10/3/2006 to 7/20/2007) to estimate nutrient releases in summer and winter, respectively. Because the burial time varied among sites and seasons, nutrient release rates were standardized to 100 days for comparison. Probes were sent to Western Ag Innovations in Canada to be analyzed for 14 ions, including $NH_4^+$and NO_3^-.

Plant Responses

To asses potential vegetation differences among locations and edaphic conditions, species richness, aboveground biomass, and belowground biomass were measured.

Plant cover, composition, and species richness surveys were conducted within a 1 m x 1 m undisturbed subplot in the center of each 2.5 m x 2.5 m plot. Overall percent cover of mixed vegetation based on foliage cover was visually recorded, along with percent bare ground, percent lichens, percent moss, and percent rock (further subdivided into rock, cobble, and gravel). Cover estimates were based on Grossman et al., (1998) national vegetation classification system for cover scale. However, increments of 5% were used instead of 15% because the alpine vegetation is so sparse (see appendix Table 2.2). Each plant occurring in the 1 m x 1 m subplot was identified and a specimen was taken outside the plot (when needed) to compare to herbarium specimens for a positive identification. Each species of plant within the plot was assigned a percentage (with 0-5% increments as described above) for its relative contribution to plant cover compared to other species in the plot. The median of this percent cover was then used to obtain species diversity and richness.

Aboveground plant biomass was determined from destructive sampling using two 25 cm x 25 cm PVC frames that were randomly tossed into the sampling area of each plot (Fig. 2.4). After recording the percent plant cover within the frames, all vegetation within the frames was clipped completely to bare ground and separated into live and dead portions. The samples were placed in a drying oven at 65° C for 24 hours and the weight of aboveground live and dead biomass was recorded. The samples were then ground into powder using a Wiley mill and mortar and pestle, and a subsample was sent to the USU Analytical Labs for a C and N analysis (LECO).

Belowground root biomass from the first soil core was determined by manually removing the roots from the soil and separating the mineral material by floating the roots in water. The roots were then dried at 65° C for 24 hours and weighed. The volume of the core and weight of the roots were used to estimate the root biomass per square meter for the top 10 cm of the soil horizon. The roots were then ground into a fine powder with a mortar and pestle and a sub-sample was sent to the USU Analytical Labs for total C and N analysis using LECO.

Statistical Analysis

The study design is a 2 factor (factor1: nitrogen deposition gradient. factor 2: edaphic conditions.) sample type. A mixed model ANOVA on the response variables for differences between locations and edaphic conditions was conducted. The fixed effects are: (1) The N deposition gradient because they were not chosen at random; they were chosen based on the Nanus et al., GIS maps (Nanus et al., 2003); and (2) Edaphic conditions (wet/dry communities) because there are other types of communities than just wet and dry. The random effects are: The 3 areas in each location (i.e. replicates), and the locations of the wet/dry plots (i.e. within the wet/dry communities, the plots were randomly placed).

Two-way ANOVAs were performed on the various plot-level measurements to test for significant differences among sites along a N deposition gradient (hypothesized high, medium, and low sites), differences between wet and dry sites, and to test whether there were significant interactions between edpahic conditions and gradient effects for the measurements considered.

The study examined whether there is a gradient from north to south in the Grand Teton National Park, whether this is reflected in site characteristics and N status differences, and to what extent responses differ with edaphic conditions within the alpine ecosystem.

To asses N input, a test for hypothesis 1, the data from Fenn collectors, Johnson collectors, and snow pack surveys were considered in the analysis. The snow pack survey gradient could not be statistically compared because there is only one data point for MB in 2006, two data points for RDV for 2006 and 2007, and no data points for PB because of inaccessibility during the harsh winter months in the Tetons.

To assess differences in N status and test hypothesis 2, soil and plant N indicators were compared. To characterize potential differences in N status, soil N content, C:N ratios, the amount and form of KCl extractable inorganic N, nitrification potential, and available N (i.e., PRS™ probes), as well as plant N contents and distribution were considered in the analysis.

To test for differences in plant responses to N input/status in the wet versus dry plots, and to test hypothesis 3, Shannon-Weiner indices for species richness, abundance, and evenness were calculated based on the plant cover estimates. Biomass (in grams), plant N content, C:N ratio, and percent cover were compared using two-away ANOVA.

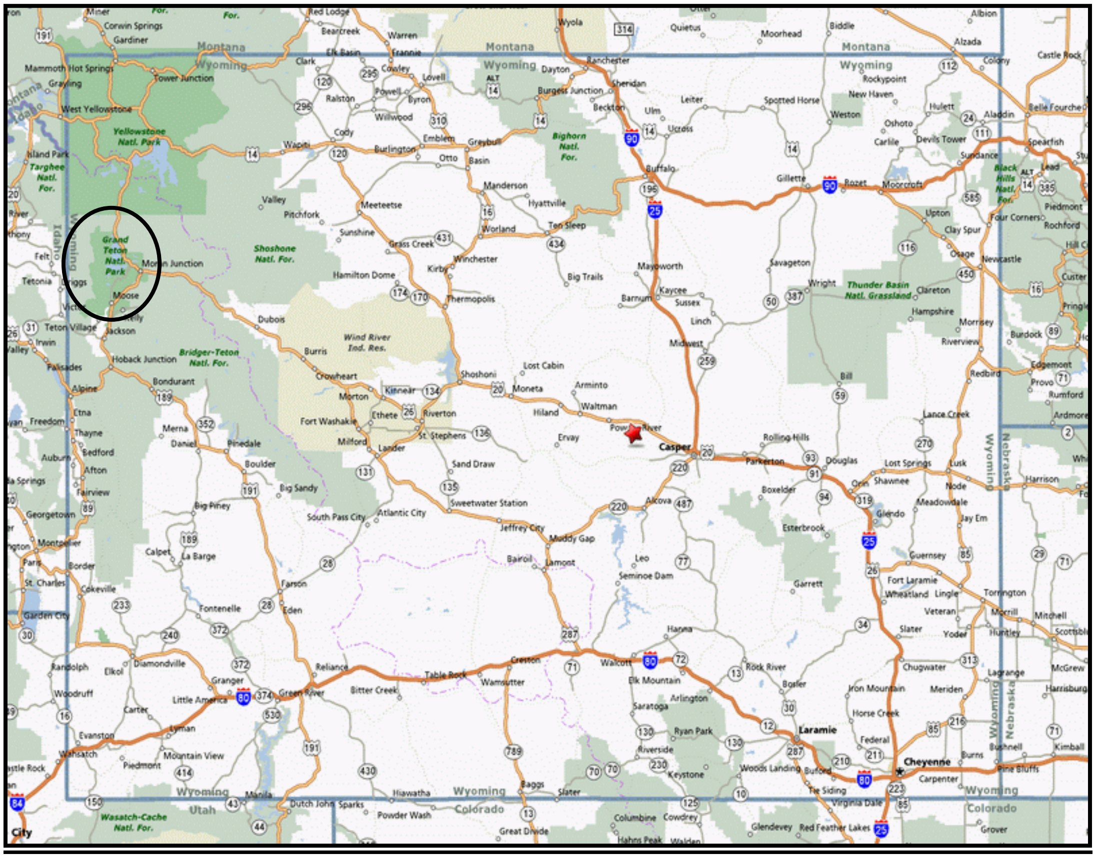


Fig. 2.1. Map of Wyoming illustrating where Grand Teton National Park is located (in black circle)

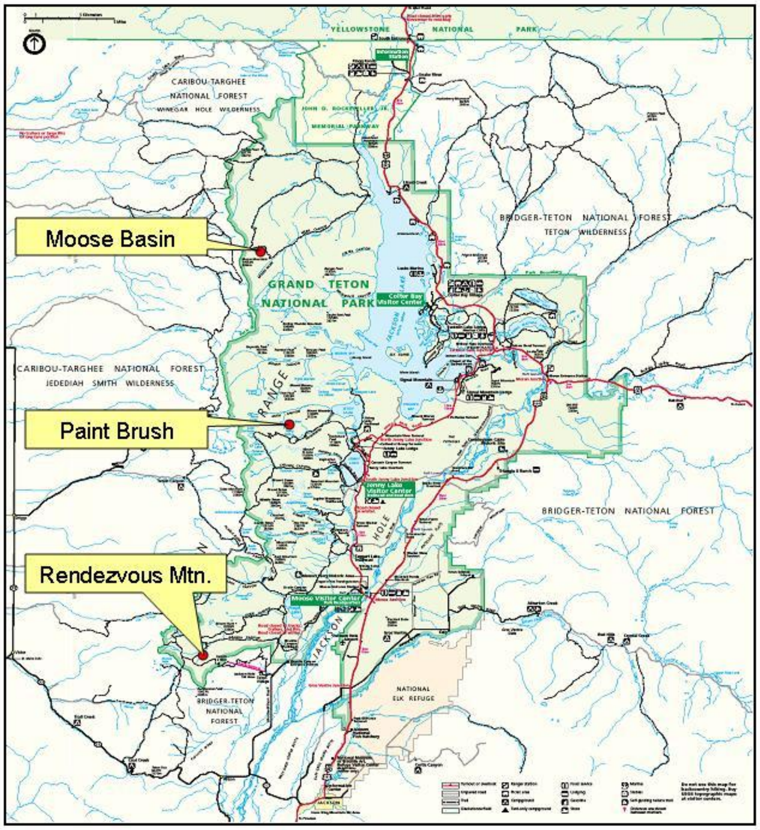


Fig. 2.2. Map of Grand Teton National Park with field site locations, Moose Basin, Paint Brush Canyon, Rendezvous Mountain.

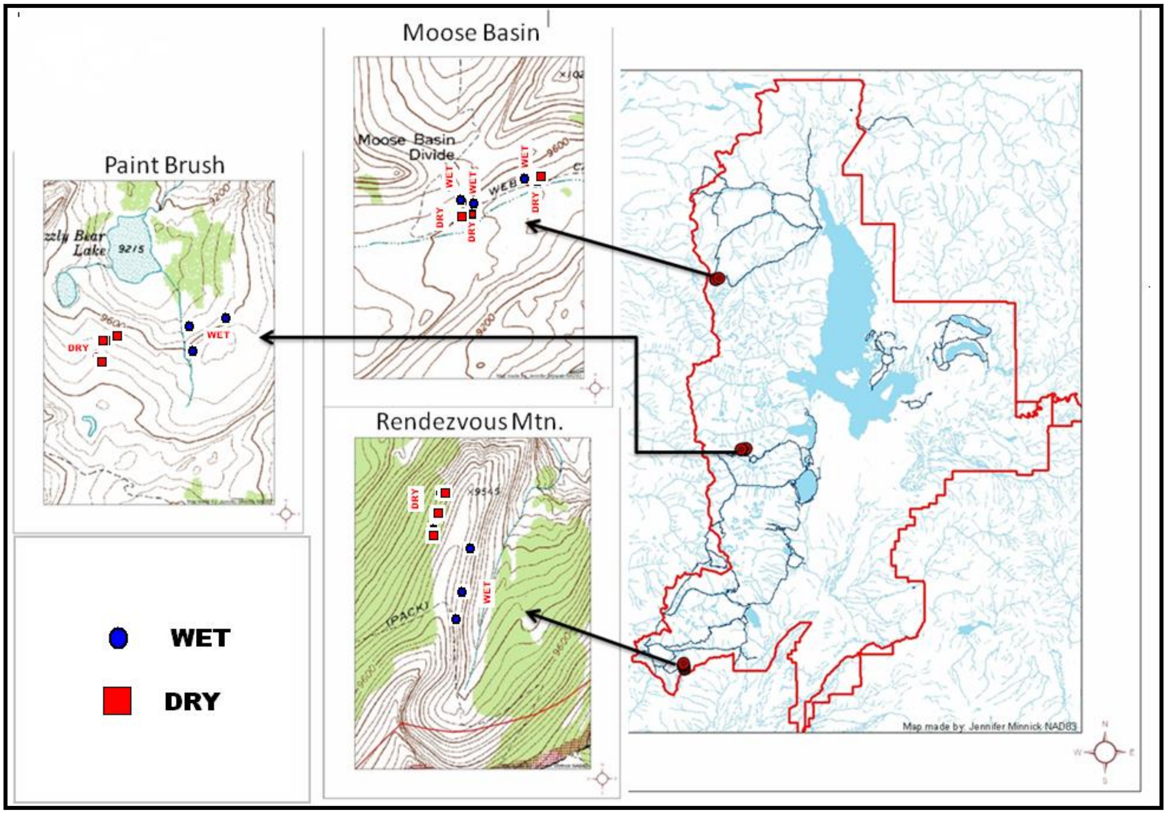


Fig. 2.3. Plot location layout. Wet and dry plots were located within 200 m of each other. Areas were located approximately 200-800 m apart at each location. Maps were made using NAD83 projection.

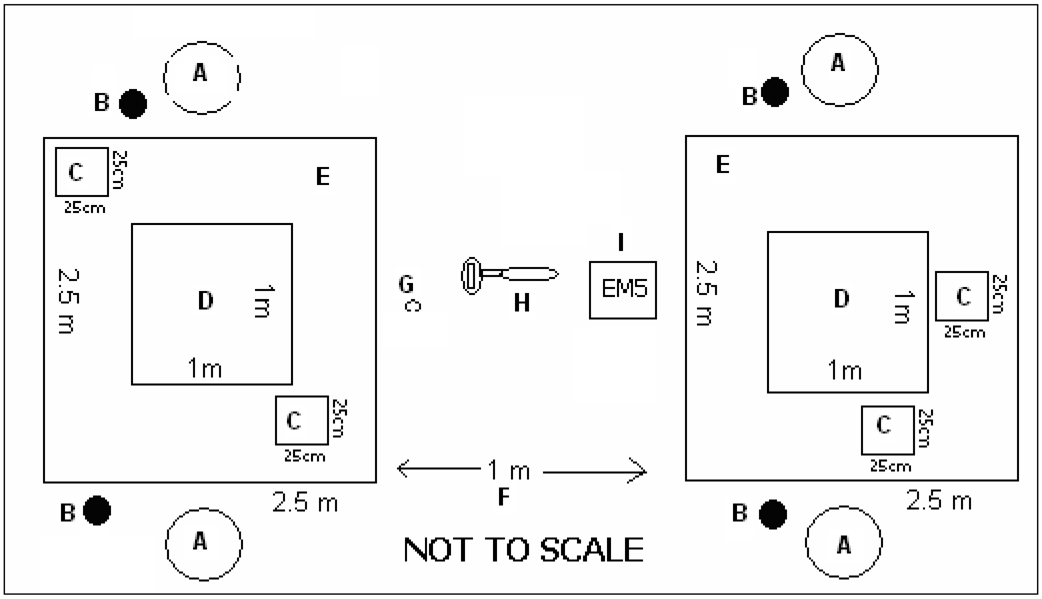


Fig. 2.4. Plot pair layout: A: Fenn Collectors B: Johnson collectors C: 25 cm x 25 cm PVC frames for destructive biomass sampling D: 1 m x 1 m undisturbed subplots for species richness surveys and treatment regime. E: 2.5 m x 2.5 m plot F: 1m buffer zone between plot pairs G: Onset© tidbit temperature data logger H: Decagon© ECH2O moisture probe. I: Decagon© Em5b soil moisture data logger (2 per site)

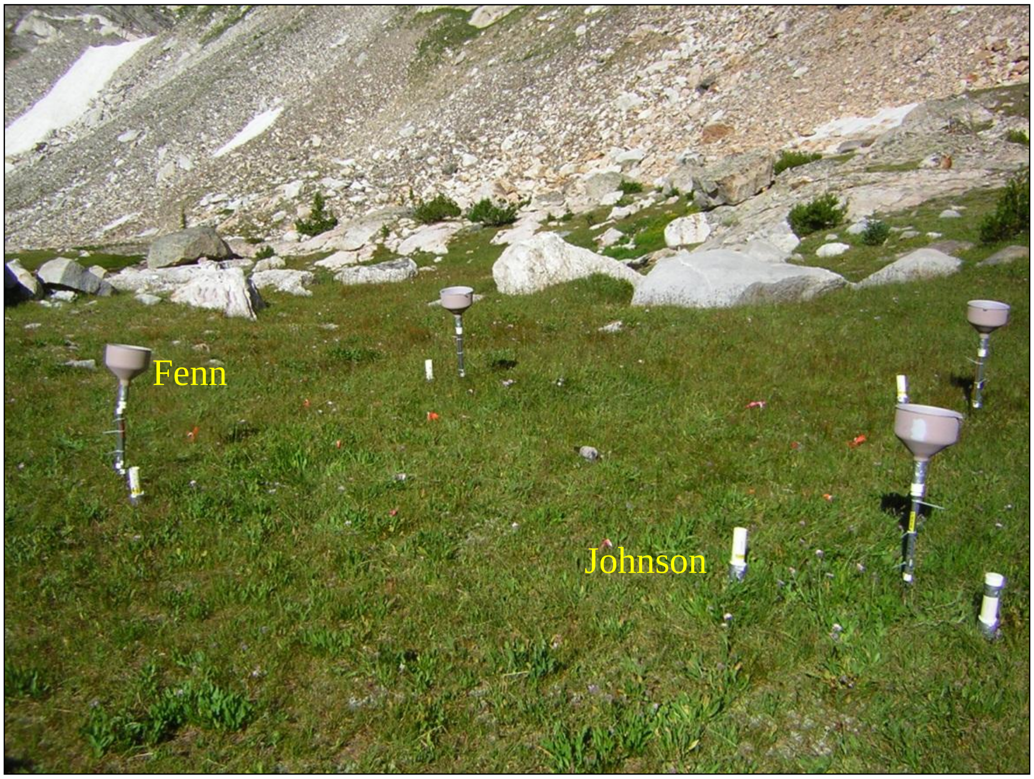


Fig. 2.5. Ion exchange resin tubes (IERT); Fenn and Johnson collectors set up in the field at paint brush wet site.

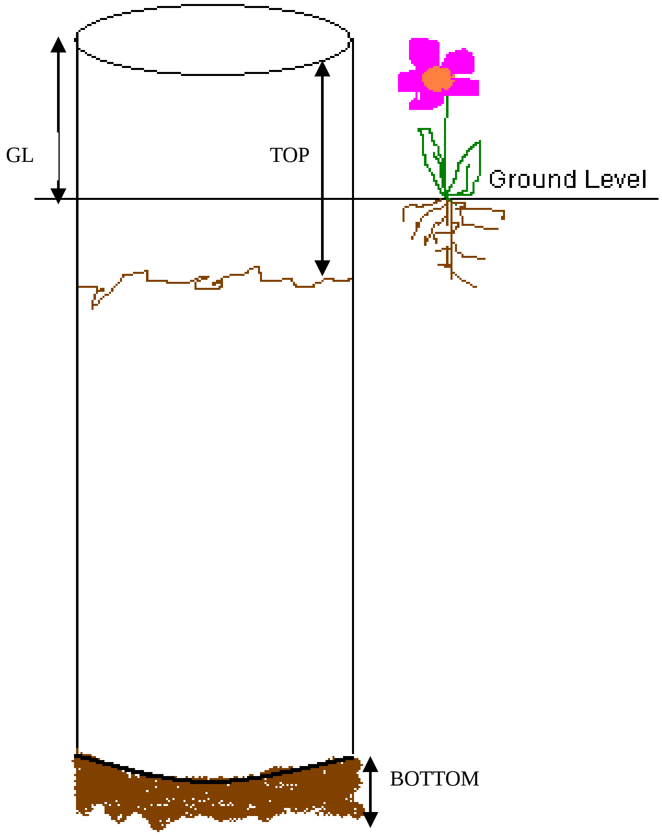


Fig. 2.6. Measurements of soil cores from each plot. GL: ground level. when the core was pounded into the ground, this was the distance the core went in or the measurement of how far the core stuck out of the ground. 2) TOP: the measurement of the actual soil inside the core to the top of the PVC. This measurement was to take compaction into account from pounding the core into the ground and was taken BEFORE the core was removed from the ground. 3) BOTTOM: Measurement of how far the soil stuck out from the bottom (if at all) of the core AFTER the core was removed from the soil.

CHAPTER 3

NITROGEN DEPOSITION INPUTS IN GRAND TETON NATIONAL PARK

The Teton Mountains run in a north-south direction with most of the precipitation coming from the northwest and moving toward the southeast (Curtis and Grimes, 2004). The working hypothesis in this study is that the north end of the national park receives more atmospheric N deposition than the south end of the park, resulting in the possible gradient observed in the modeled N deposition maps developed by Nanus et al., (2003).

To test these proposed differences in N deposition, three different measurements were used: 1) snow pack surveys for winter 2006 (two sites only) and winter 2007 (one site only); (Fig. 3.1); 2) N inputs over a 90-day period in summer 2006 and 2007 using exchange resins (Fenn collectors; see fig. 2.6 in chapter 2); 3) Annual N input from summer 2006 to summer 2007 using exchange resins (Johnson collectors; see fig. 2.6 in chapter 2).

In the alpine zone of the Tetons, the four seasons were separated by months: December, January, February, and March were considered the winter months; April, May and June were spring; July, August, and September were summer; and October and November were the fall months. For the year 2006, the Teton study sites received a total summer precipitation of 10.74 cm (4.23 inches) for MB, 14.66 cm (5.77 inches) for PB, and 12.55 cm (4.94 inches) for RDV. The 2006 winter precipitation was 96.04 cm (37.81 inches) for MB, 71.86 cm (28.29 Inches) for PB, and 82.65 cm (32.54 inches) for RDV. The 2007 summer precipitation was 18.85 cm (7.42 inches) for MB, 20.27 cm (7.98 inches) for PB, and 16.00 cm (6.30 inches) for RDV. The 2007 winter precipitation was

60.22 cm (23.71 inches) for MB, 56.95 cm (22.42 inches) for PB, and 65.71 cm (25.87 inches) for RDV. Based on the 2006 and 2007 data, winter precipitation comprises 84%-89% of the annual precipitation (84% in MB, 79% in PB, and 89% in RDV). Therefore, the predominant form of precipitation in the Tetons comes in the form of snow.

According to long-term annual precipitation from the PRISM data (from 1971-2000), MB receives slightly more annual mean precipitation (174 cm) than PB (168 cm) and RDV (166 cm), with most in the form of snow. This could account for the observed higher N deposition in MB than in RDV.

The snow pack survey showed a winter N input at MB of 1.70 kg N ha^{-1} with NH_4^+-N comprising 59% of the total winter N input. RDV showed 0.96 kg N ha^{-1} with NH_4^+-N comprising 53% of total N in 2006. For the 2007 snow pack survey, there was 0.85 kg N ha^{-1} at RDV with NH_4^+-N comprising 49% of the total winter N (Fig. 3.1). Moose Basin was inaccessible in 2007 due to avalanche danger.

The long term USGS data showed an average winter input of 0.12-0.81 kg N ha^{-1} as NH_4^+-N and 0.20-0.72 kg N ha^{-1}as NO_3^--N (Ingersoll et al., 2008 and 2009) for the central Rocky Mountains region, with an 11-year average (1993-2004) for Rendezvous Mountain of 0.41 kg N ha^{-1} for NH_4^+-N and 0.43 kg N ha^{-1}as NO_3^--N (Ingersoll et al., 2009). The measured total N input snowpack data from MB and RDV fall within the long term averages for the snow pack data reported by the USGS of RDV, and can thus be considered representative (Fig. 3.1).

Our data would support a north to south gradient for snow N input, but since snow pack surveys only represent two data points, data from more sites would need to be collected to ascertain such a gradient.

Fenn collectors at the three locations were set out for an average of 90 days to collect summer atmospheric N deposition for 2006 and 2007. In general, summer N input was small (0.2-0.3 kg N ha^{-1}) with NH_4^+-N comprising 60% of the total summer N input. There is no evidence of a significant north to south gradient in summer N input for either inorganic N or NH_4^+-N and NO_3^--N separately. Overall, when the 2006 and 2007 summer N inputs are compared to winter inorganic N inputs from the snowpack survey, most of the N coming into the system comes during winter rather than the summer months with slightly more in the form of NH_4^+ (Fig. 3.2).

The summer inorganic N inputs for both 2006 and 2007 show no significant differences between wet and dry sites or significant interactions between location and edaphic conditions (Fig. 3.3). Since topography dictates where snow is deposited in alpine zones (Fisk et al., 1998), it was hypothesized that the wet sites, which accumulate more snow, would receive more inorganic N than the dry sites, especially in winter. This is possibly because the wet sites in the winter accumulate more snow pack via wind-blown deposition on leeward sides of topography (Rehder, 1976). During the summer months however, the wet and dry sites receive the same distribution of precipitation and are no longer affected by the snow. Therefore, no significant differences in N input between wet and dry sites would be expected in the summer months.

Johnson collectors were set out at the field sites for approximately 1 year (from 7/8/06-7/20/07) to estimate annual N input and to compare values obtained from summer plus snow N input for the three sites. Table 3.1 summarizes the N input from the Johnson collectors. Note how much more NH_4^+ -N there is compared to NO_3^- -N. The N flux derived from the Johnson collectors would suggest an annual inorganic N input of 15-23 kg N ha^{-1} yr^{-1}, an amount well above the critical load estimated by Williams and Tonnessen (2000) of 4 kg N ha^{-1} yr^{-1}. When compared to winter snow pack inputs (Fig. 3.4), the Johnson collectors seem to capture significantly more NH_4^+-N (8-10 kg N ha^{-1} yr^{-1}) than the snow pack survey indicates. While the NH_4^+ levels seem to be higher than expected, the NO_3^- levels seemed to slightly coincide to the reported data of the Nanus (et al., 2003) deposition maps that show the highest input of NO_3^--N in the Rocky Mountains is 0.46-0.68 kg NO_3^- -N ha^{-1} yr^{-1} along the Colorado Front Range, the Park Range in Colorado, and the Wasatch Range in Utah. The snow pack survey and the summer NO_3^- -N input were summed for MB and RDV and then compared to Johnson collector values. The Johnson collectors are still two to three times higher for NO_3^--N (2.42 kg N ha^{-1} yr^{-1} vs. 0.76 kg N ha^{-1} yr^{-1} for MB, and 1.17 kg N ha^{-1} yr^{-1} vs. 0.56 kg N ha^{-1} yr^{-1} for RDV Fig. 3.5). Based on the Nanus maps of NO_3^--N deposition (Nanus et al., 2003), MB receives 0.46 kg NO_3^--N ha^{-1} yr^{-1}, PB 0.34 kg NO_3^--N ha^{-1} yr^{-1}, and RDV 0.12 kg NO_3^--N ha^{-1} yr^{-1}. Our NO_3^--N input estimates for the summer plus winter compared well to the trends from 1993-2004 for the Central Rocky Mountain region with mean NO_3^- -N inputs of 0.41 kg NO_3^- -N ha^{-1}yr^{-1} reported by Ingersoll (2008). Though the Johnson collector NO_3^--N data was higher than what Nanus (2003) and Ingersoll (2008) reported, it still compared to literature more than the NH_4^+-N. Therefore, further analysis was performed on the Johnson NO_3^--N inputs.

There were no significant differences in NO_3^--N inputs estimated from the Johnson collectors by site, gradient, or edaphic condition. The NH_4^+-N data on the other hand, seem suspect and need investigation as to what might be responsible for these high estimates. Therefore, no further statistical analysis was performed on the NH_4^+-N data from the Johnson collectors.

Resins have been used since the mid-1930s (Adams and Holmes, 1935) to mimic soil colloid function, and more recently, plant roots (Hangs et al., 2004). Our study used the resins to capture N before it reached the soil to estimate atmospheric N input as Fenn and Johnson did as a new experimental utilization of these resins (Fenn and Poth, 2004; Susfalk and Johnson, 2002; Johnson et al., 2005).

There are several possible explanations for the unusually high NH_4^+ N input values derived from the Johnson collectors: The resins could have been breaking down from long exposure; there could have been external source contamination to the resins; the resin affinities for NH_4^+ could have been greater than for NO_3^-, so more NH_4^+ was being captured by the resins; the extraction methods used could have yielded greater extraction recovery of NH_4^+ than NO_3^-; or other unaccounted for phenomena could have taken place in this study as well.

Our study used resin beads that had amide active groups which function as site exchangers (Skogley and Dobermann, 1996). Several studies indicate that these amine groups, when placed in the field for extended periods of time, can start to break down resulting in overestimates of NH_4^+ (Binkley and Matson 1983; Kjønaas 1999; Langlois et al., 2003; Fenn and Poth, 2004). This may have happened to the Johnson collectors in this study since they were placed in the field for close to one year and were subject to wet/dry and freeze/thaw cycles.

One way to quantify and account for the degree of resin breakdown is to include “blanks” which are resin collectors sealed with plastic caps set out into the environment.

These do not collect anything but reflect background “noise” originating from the resins themselves (Skogely and Dobermann, 1996; Langois et al., 2003; Simkin et al., 2004). Blanks were set out in our study for both Johnson and Fenn collectors. The average blank extraction results for the Johnson collectors was between 0.15-0.65 µg/g for NO_3^--N, while NH_4^+-N values were 0.92-2.66 µg/g. In the Fenn collectors, the average blank extractable NO_3^--N was 0-0.15 µg/g and 0.10-0.16 µg/g for NH_4^+-N. Both collectors exhibited more extractable NH_4^+-N than NO_3^--N which could be a result of resin breakdown from wet/dry and freeze/thaw cycles (Mamo et al., 2004). The Johnson collectors showed as much as ten times more resin modification than the Fenn collectors. Exposing the resin for 6-12 months with freeze/thaw and wet/dry cycles may have increased the NH_4^+ values and provided a potential for microbial transformation into NO_3^-. This resin breakdown may partially explain the higher NH_4^+ numbers exhibited in the Johnsons collectors, but the extractable NH_4^+-N is far above the average extractable NH_4^+-N found in the literature on these types of resins and collectors (Susfalk and Johnson, 2002; Langois et al., 2003; Fenn and Poth, 2004).

Contamination from soil dust or organic matter that blows into the collector can potentially inoculate the resins with soil microbial entities (Susfalk and Johnson, 2002). Microbes ingest the organic N inside the collector and transform the atmospheric N contaminants and amine groups released from the resins. This further adds to mineralized inorganic N in the collectors. The Johnson collectors were placed less than five cm above the ground with the open top no more than 25 cm from the ground, whereas the bottom of the Fenn collectors were at least 15-20 cm above the ground. The Johnson collectors may have been close enough to the ground to capture wind-blown soil particles and organic matter to create some contamination within the tubes.

Mamo et al., (2004) conducted a study on the integrity and affinities of strong acid-base mixed bed resins (resins similar to the one used in this study) in freeze-thaw and wet-dry conditions. They found that freeze-thaw cycles did not affect the adsorption efficiency of the resins, but wet-dry cycles caused the resin beads to swell and shrink causing desorption of N by 3.3%. Skogely and Dobermann (1996) examined affinities for resin adsorption and found that the affinity for NH_4^+ on the exchange site on synthetic resins ranked 19th out of 22 cations present in the soil, especially when the active group is sulfonated (Langois et al., 2003). The affinity for NO_3^- ranked 3rd out of 19 anions present in the soil. If NO_3^- indeed has a stronger affinity to the resin beads than NH_4^+, the Johnson collectors should have collected and reflected more NO_3^- in the KCl extracts than NH_4^+. Perhaps the wet/dry cycles described by Mamo et al., (2004) influenced the affinities of NO_3^- and NH_4^+ in the field differently causing desorption of NO_3^- but not NH_4^+ during field incubations resulting in higher extractable NH_4^+-N. However, this cannot account for the unusually high NH_4^+-N deposition values measured.

Lab studies are sometimes done to quantify adsorption efficiency and recovery of adsorbed N with KCl extractions. Typically, a solution carrying a known amount of the ion of interest is poured over the resins. The throughfall solution is analyzed to determine adsorption efficiency for the ion. The ion is then extracted from the resins. This gives the investigator an idea of the resin affinities for the ions, how much of the ion slips by the resin (i.e., adsorption efficiency), and how much of the ion does not get extracted from the resin (i.e., recovery). Langois et al., (2003) found that the recovery for NO_3^- was anywhere between 50%-100% and the recovery for NH_4^+ had much more variability, ranging from -146% (i.e. more background level than recovered) to 22.5% (Binkley and Matson, 1983; Skogely and Dobermann, 1996; Langois et al., 2003, Reno, 2006; Simkin et al., 2004). Langois et al., (2003) also found that the recovery for NO_3^- was influenced by the resin pretreatment (i.e. rinse with KCl solution before deploying to the field), the brand of resin, and the extraction solution used; whereas NH_4^+ recovery was not affected by this. However, as discussed earlier, resins made of amino compounds could release NH_4^+ and interfere with analysis (Binkley and Matson, 1983; Langois et al., 2003). This could explain the background "noise" found in the blanks with more NH_4^+ than NO_3^-, but it does not seem to account for the differences between the Fenn vs. Johnson collectors for the unusually high amounts of NH_4^+ observed.

Overall, this study took into consideration the 2 M KCl pretreatment (it was decided not to pretreat the resin), the type of resin to use to collect atmospheric N, comparison of results to other similar studies using similar methods for collecting N, and comparison of results to NADP stations set up throughout the Rocky Mountain Region. The phenomena of unusual amounts of NH_4^+, but comparable amounts of NO_3^- is perplexing and an explanation for it goes beyond the scope of this project. Mixed-bed resins may not be an accurate way to estimate NH_4^+-N deposition when collectors are placed in harsh field conditions for prolonged periods of time such as ours were.

In summary, our measurements show that the Tetons receive 1.42 kg N ha^{-1} yr^{-1} of N deposition (0.25 kg N ha^{-1} in the summer months and 1.17 kg N ha^{-1} in the winter months) which coincides well with the Nanus et al., (2003) estimate of 0.1-2.5 kg $Nha^{-1}yr^{-1}$for the Tetons. However, this seems to be a little low compared to the hot spot found along the Colorado Front Range of the Rocky Mountains which receives upwards of 9 kg N ha^{-1} yr^{-1} (Williams and Tonnessen, 2000; Williams et al., 2003; Burns, 2003; Blett et al., 2004; Bowman et al., 2006). The input pattern for N in the Teton Mountains follows a similar trend observed in the rest of the Rocky Mountain Range, with more N coming in the form of snow.

Table 3.1 Nitrogen input from Johnson collectors set out for standardized 365 days (kg N ha^{-1} yr^{-1})

Site Name	NH_4^+-N	NO_3^--N	Inorganic N
MB	9.28	2.42	11.7
PB	10.2	0.80	11.0
RDV	7.62	1.17	8.79

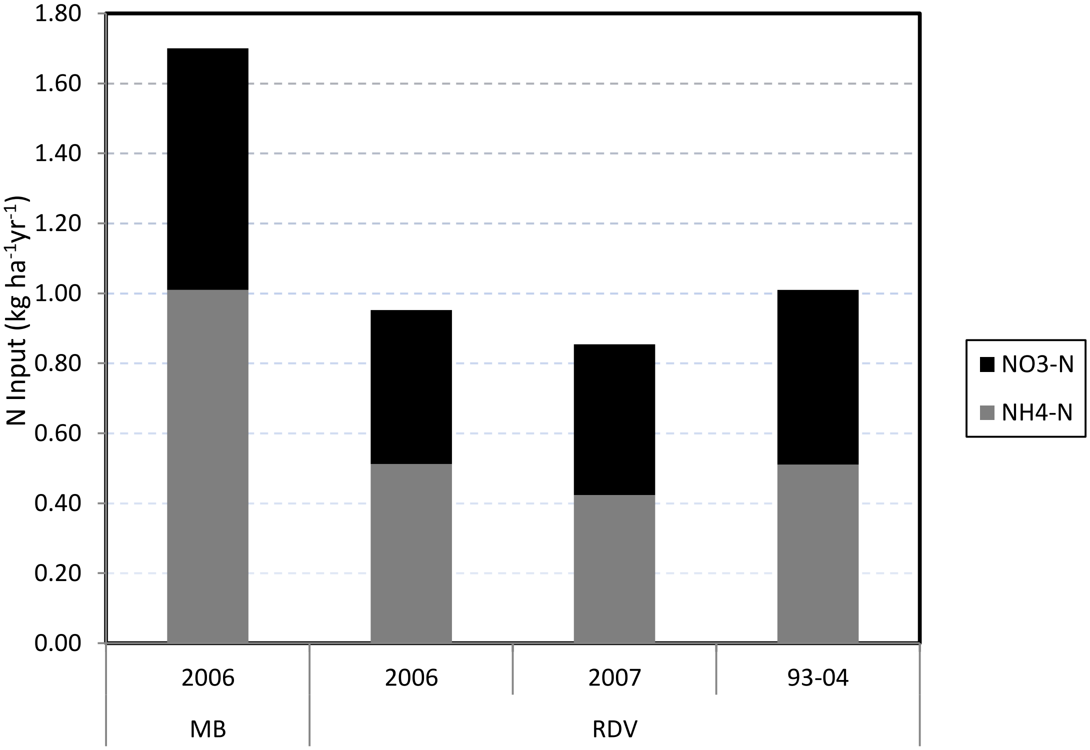


Fig. 3.1 Comparison of snow pack surveys (N input kg ha^{-1}) for 2006, 2007, and long-term average of RDV from 1993-2004 (from Ingersoll *et al.*, 2008 table 2).

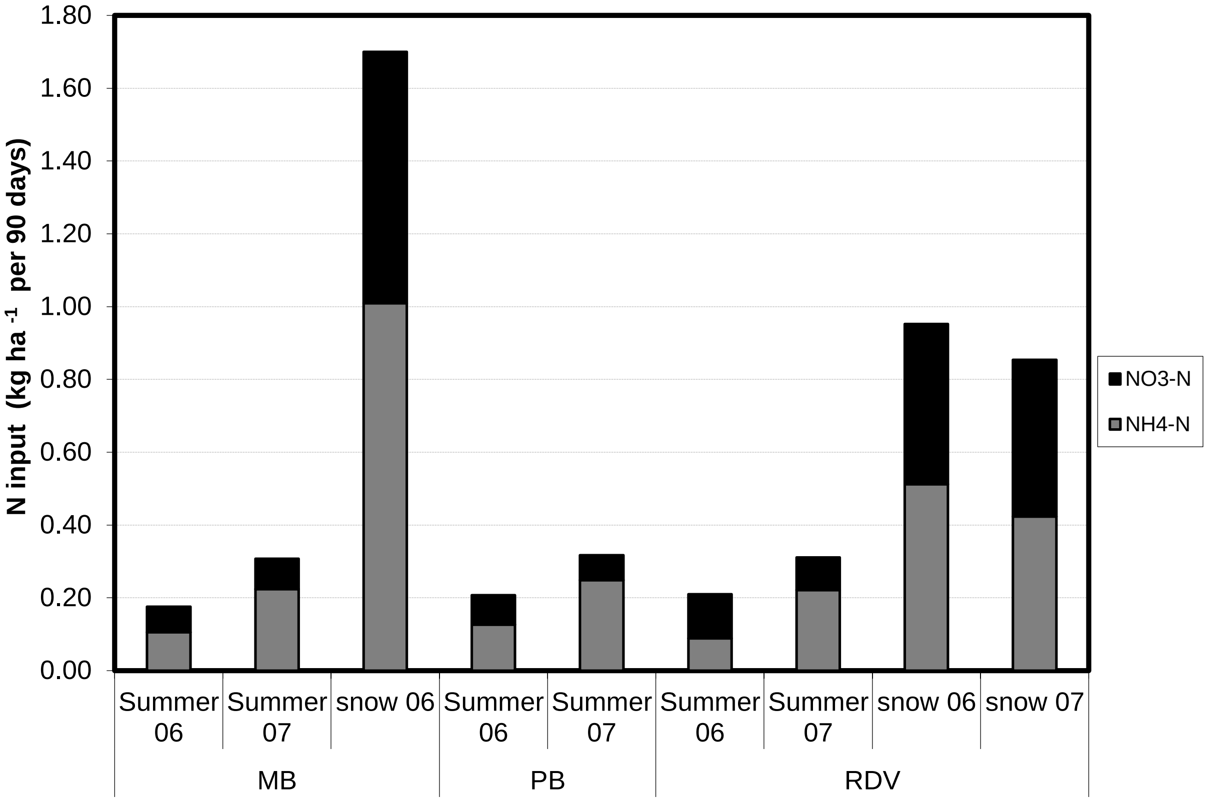


Fig 3.2 Amount and form of inorganic N input (kg N ha^{-1}) and during 90-day period in summer 2006 and 2007. All sites compared to winter inputs at MB (2006) and RDV (2006, 2007).

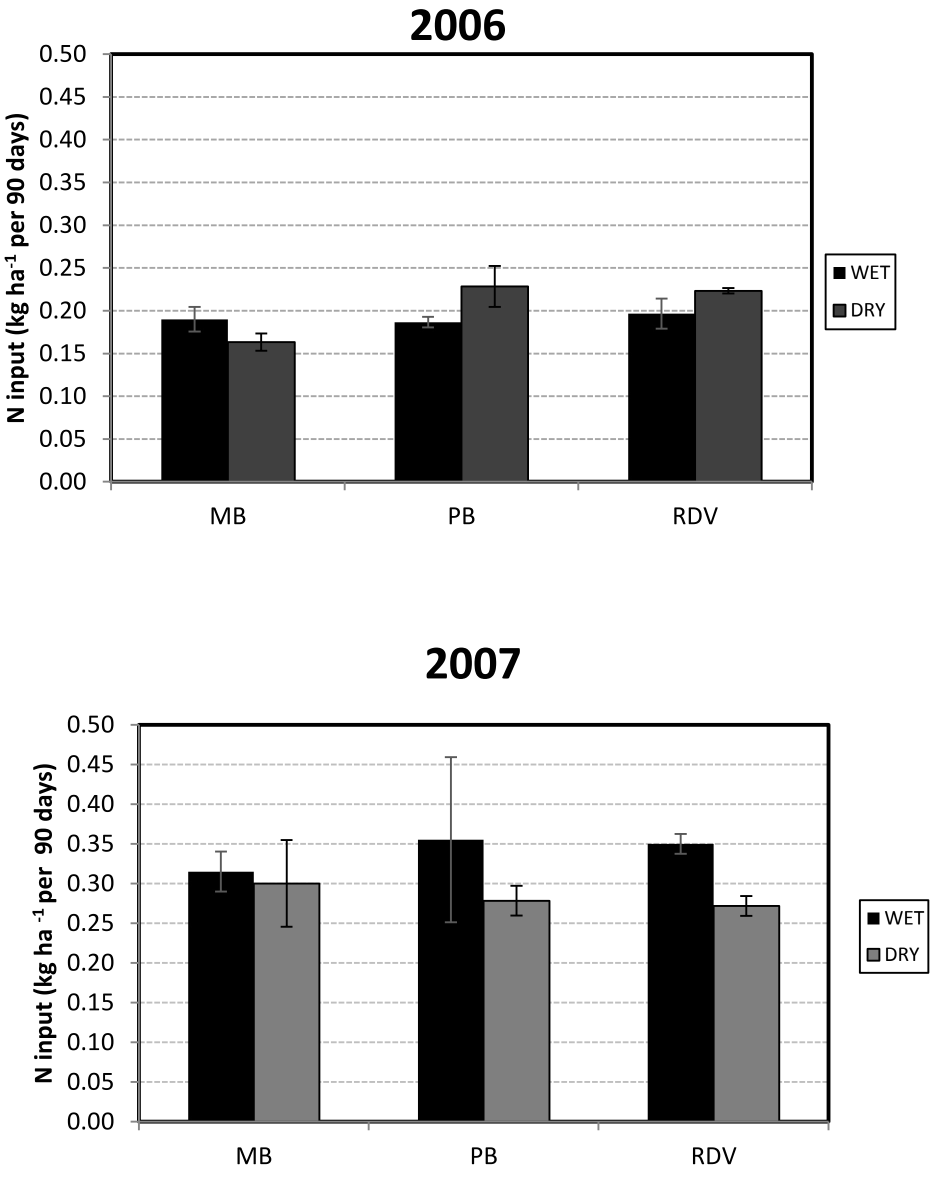


Fig 3.3. Comparison of inorganic N input (kg N ha^{-1} per 90 days) during summer 2006 and 2007 in dry and wet sites along a north-south gradient.

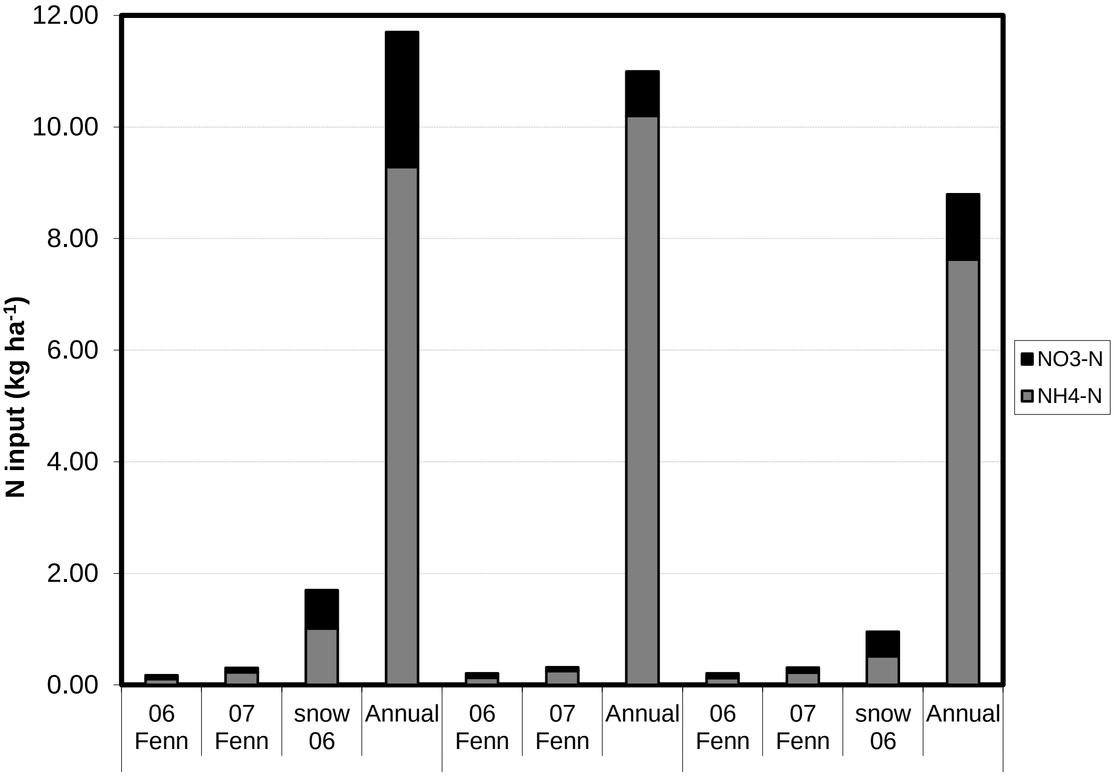


Figure 3.4 Amount and form of N input during summer 2006 and 2007 (Fenn collectors), in winter 2006, and annual input estimates (Johnson collectors) for Moose Basin, Paint Brush, and Rendezvous Mountain (kg N ha^{-1}).

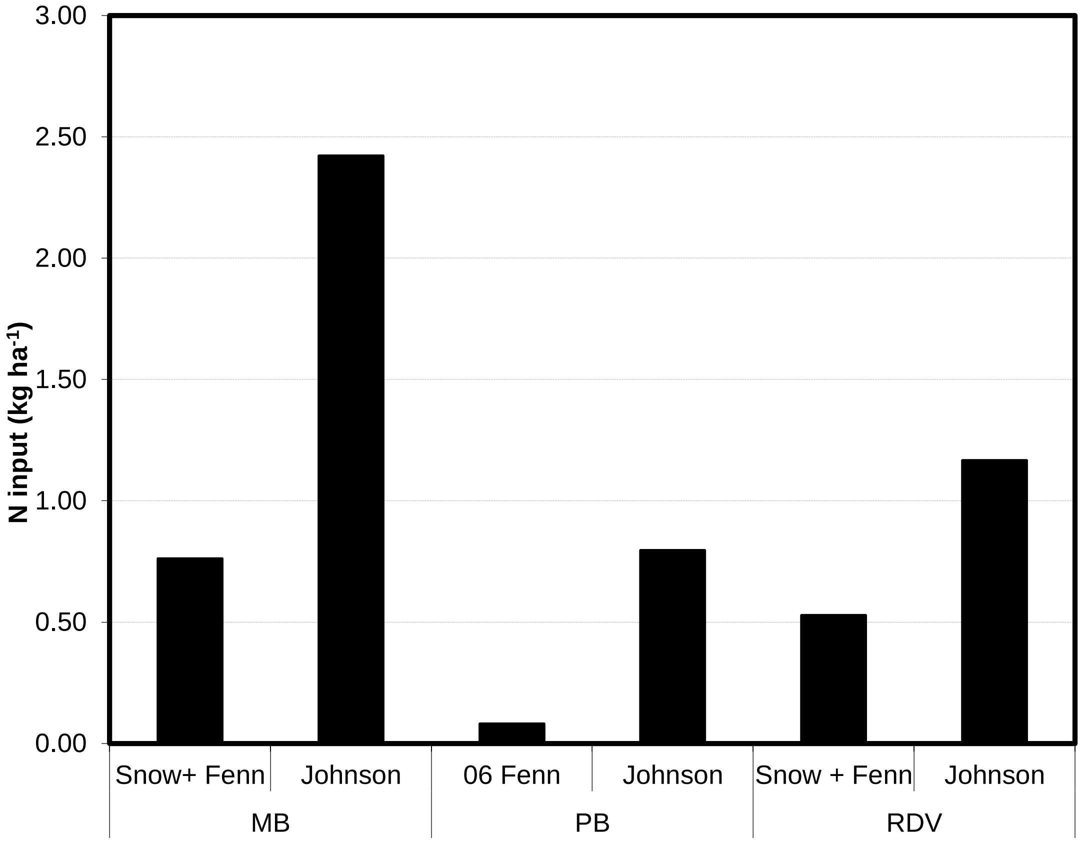


Fig. 3.5. Comparison of annual NO_3^- N input for the sum of the 2006 Fenn + Snow pack and Johnson collectors.

CHAPTER 4

EFFECTS ON NITROGEN STATUS IN THE SOILS AND PLANTS

The objectives of this study were to examine whether the observed differences in N deposition had any influence on N status at the three sites, and whether edaphic conditions at each site affected N status. Ultimately, the goal was to identify system indicators to detect changes in ecosystem structure/function that could be used in the future as early warning signs. The soil objective was to assess soil N status both statically and dynamically and evaluate what system characteristics, if any, were strong or weak indicators for differences in soil N status. The plant response objective was to assess effects of N deposition on plant characteristics, such as above and belowground biomass, N content, species composition and species richness.

Soil Microclimate

The microclimate of the soil needs to be considered when examining soil nutrient cycling. The major factors of soil microclimate, such as soil moisture and temperature, influence the rate at which N cycles and becomes available in the soil. Soil moisture and temperature affect nutrient availability by influencing distribution and chemical reactions throughout the soil matrix. Our hypothesis was that wet sites would have more available N than the dry sites because 1) wet sites capture more snow and N deposition and 2) wet sites remain wetter in the summer than the dry sites allowing more turnover of organic matter.

To determine soil moisture regime, moisture probes (ECH2O probes) were placed in sites that were *a priori* designated as wet and dry based on snow retention patterns (see chapter 2). Six probes (3 for wet and 3 for dry sites) were placed between the plot pairs at the three locations for a total of 18 probes. These probes were factory calibrated to provide volumetric moisture content using a uniformly loamy soil that did not consider a coarse fraction for the soil. Due to data logger failure, continuous annual moisture measurements were not possible. When each plot was visited (twice per summer), soil moisture was measured with a hand-held device (ECH20 Check soil moisture monitor, Decagon Devices© Washington) and expressed as millivolts converted into volumetric water content.

In addition, soil samples were taken from the plots on 8/4/2006 and 7/11/2007 at MB, 8/14/2006 and 7/20/2007 at PB, and 9/4/2006 and 7/14/2007 at RDV for KCl extractions. Soil moisture content was determined on a subsample for each plot and converted into volumetric moisture content based on measured gravimetric soil moisture content and soil bulk density from soil cores. This allowed a comparison between the ECH20 probe readings and volumetric soil moisture content, and a means to further compare edaphic conditions. The ECH20 probes recorded a range between 0.03-2.27% soil moisture, while the soil samples had a range of 3.15-38.29% volumetric soil moisture content. A simple regression analysis to evaluate if there was any correlation between the ECH20 probe readings and the volumetric moisture content of the soil samples revealed that there was little, if any, correlation between the two methods ($R^2= 0.026$). The ECH20 probes may therefore not be good indicators to monitor soil moisture unless the probes are calibrated to the specific soils under investigation rather than using the factory calibrations (see Cobos and Chambers, 2010 to learn how to calibrate ECH20 probes for specific soils).

A two-way ANOVA was performed with volumetric soil moisture as the variable and compared to edaphic conditions across all locations and within sites to see if there were differences in soil moisture content. For summer of 2006, there was a significant difference in soil moisture between wet and dry sites (P = 0.004) with a significant interaction (P = 0.03) between location and edaphic condition that shows that differences between wet and dry plots were most pronounced at PB, whereas MB and RDV showed no significant difference between wet and dry plots (Fig. 4.1). There was a significant difference among the locations (P = 0.006) with RDV soils (4.4% soil moisture content) significantly drier than MB and PB sites (13.8% and 14.2% soil moisture content, respectively) which were not different from each other. For summer of 2007, there was a significant difference at the site level between wet and dry (P = 0.002), again mainly due to the PB sites where the wet sites had 15.3% and the dry sites had 3.1% soil moisture content. Neither location nor the interaction were significant.

Comparison of the 2006 and 2007 volumetric soil moisture suggests that 2006 was a wetter year than 2007. Climate data from the PRISM (Parameter-elevation regression independent slopes model, 2006 and 2007) data set suggests that in summer 2006 (i.e. July-September) MB received 5.8-7.5 cm, PB received 5.0-6.5 cm, and RDV received 5.6-7.0 cm of summer precipitation. In 2007, MB received 4.0-6.8 cm, PB received 5.6-6.6 cm, and RDV received 4.8-5.8 cm summer precipitation. Based on these data, MB and RDV indeed received slightly less precipitation in 2007 suggesting it was a drier year for those sites.

In summary, all the dry sites seem to have similar soil moisture over the snow free period for both 2006 and 2007. However, sites designated as “wet” differed in soil moisture content among sites and sampling dates. The PB wet sites had significantly more soil moisture in 2006 compared to the RDV sites but not in 2007. The PB wet sites could of had significantly more moisture in 2007, but the soils in the RDV wet sites were wetter in 2007 than in 2006, thus reducing differences to insignificant levels between PB and RDV wet sites in 2007. The PRISM data suggests that 2006 had slightly more summer precipitation than 2007.

We also wanted to see how different the edaphic sites were in terms of temperature. To determine this, temperature data loggers were installed in each plot-pair (18 loggers total) to measure soil temperature. There was an abrupt increase in soil temperature during the winter to spring transition, where temperatures went from zero to above zero, indicating the date of the beginning of the snow free period (Table 4.1). We anticipated that differences in edaphic conditions were reflected in the dates of the snow free periods, namely that wet sites would have a snow pack longer than dry sites. However, the data from wet and dry sites at RDV and the three wet sites and one dry site at PB suggest that the snow free period for these sites came on similar dates (late June). Data from the one wet site and the three dry sites at MB suggest that the snow free period came earlier (late May-early June) at MB than RDV and PB, and that MB dry sites might have lost snow pack slightly earlier than the MB wet sites.

An ANOVA was performed on the temperature on the date of the snow free period to see if there were differences in temperature among the three sites. There was a significant difference in location (P = 0.005) with RDV sites being overall warmer than

PB and MB (i.e., RDV 3.96 °C, PB 1.84 °C, and MB 2.26 °C on the date of the snow free period). A comparison among wet versus dry sites was not done because there were not enough data.

Annual degree days were calculated from 7/10/2006 to 7/10/2007 for MB and RD, and from 8/14/2006 to 8/14/2007 for PB (Fig. 4.2) and wet and dry sites were compared with a two-way ANOVA. The results showed that there is a significant difference between locations ($P = 0.02$) with RDV being slightly warmer than PB, while the temperature regime at MB was not different from either RDV or PB. There were no significant differences in edaphic conditions and no significant interactions among location and edaphic conditions.

In summary, the moisture and temperature data show that RDV had lower overall soil moisture during the growing season, and warmer soil temperatures on the date when soils became snow free compared to PB and MB. The PB wet sites had the most soil moisture of all three sites, but the temperature on the date of the snow free period was not significantly different from MB and RDV. The observed higher precipitation in the north of the park at MB from the Nanus *et al.*, (2003) maps and the PRISM data does not seem to have an effect on the soil moisture or temperature along a north to south gradient for the summer months, but it may perhaps have an effect on total annual and winter precipitation.

Ecosystem Response: above and belowground N pools

This study considered the N status in terms of N content in both aboveground biomass with live and dead foliage, and belowground with plant roots and soil.

Combining all N pools gives an estimate of the total amount of N contained in the entire ecosystem. Summer 2007 was not considered in this analysis because there is no aboveground biomass data.

The two-way ANOVA of 2006 ecosystem N pools show that there was a significant difference in location ($P = 0.004$) with MB (average N is 186.60 g N m^{-2}) having more total N than the PB (average N is 93.79 g N m^{-2}) and RDV (average N is 86.82 g N m^{-2}). There were no significant differences among wet and dry sites or a significant interaction between edaphic conditions and site (Fig. 4.3).

Soil Response: N Status

This study considered both static and dynamic indicators of soil N status. Static indicators include total N pool, extractable inorganic N (NH_4^+ and NO_3^-) in the root zone, and C:N ratios. Dynamic indicators include net N mineralization and nitrification potential, and availability of inorganic N in the soil root zone measured by PRS™ probes over standardized 100-day intervals.

It was hypothesized that soil N status in the modeled high N deposition sites (i.e. Moose Basin) would be higher than the soil N status in the modeled low N deposition sites (i.e. Rendezvous Mountain). The hypothesis stated that MB was already showing signs of higher N status possibly expressed by higher soil N, more extractable N, lower C:N, more available N, and higher nitrification potential, compared to the RDV and PB sites. To test this hypothesis, soil cores were taken in 2006 and 2007, and a sample from the fine fraction portion (with roots removed) was analyzed for C and N content (g m^{-2} in top 10 cm of soil), and then C:N ratios calculated from this. Inorganic N was measured in the soil with KCl extractions. Net mineralization and nitrification potential was measured as a dynamic laboratory indicator under a controlled environment. PRS™ probes were placed in the field plots for periods of times representing the changes in seasonal flux and then the data were standardized to 100 days during summer and winter to provide dynamic and seasonal field rates of resin-extractable N.

Soil N pools in the upper 10 cm determined from the soil cores were used in this study to compare N status across all three sites (Fig. 4.4). For the 2006 N pools in the top 10 cm of the soil, the two-way ANOVA shows that there was a significant difference (P = 0.001) in total soil N among locations, with MB (site average is 168.34 g N m^{-2}) having a larger soil N pool than RDV (site average is 76.68 g N m^{-2}) and PB (site average is 75.08 g N m^{-2}). There were no significant differences between wet and dry sites overall, but there was a significant interaction (P = 0.03) with dry sites having higher or lower N pools than wet sites depending on location (Fig. 4.4).

For 2007, the two-way ANOVA showed a significant difference among locations (P = 0.03) with MB (site average is 185.9 g N m^{-2}) again having a significantly larger N pool than PB (site average 126.73 g N m^{-2}) and RDV (site average 122. g N m^{-2}) which did not differ from each other. There were no significant differences between the wet and dry sites overall, but there was a significant interaction (P = 0.03) where the MB dry plots (238.49 g N m^{-2}) had a larger N pool than the MB wet plots (133.31 g N m^{-2} , Fig. 4.4) but no significant differences between wet and dry plots for RDV and PB.

It was hypothesized that the observed high N deposition (MB) sites would have lower C:N ratios than the low N deposition sites (RDV and PB) due to higher N in the soil. The two-way ANOVA for the 2006 C:N ratio showed a significant difference in location (P = 0.05) with the highest values at RDV, that were significantly different from those in PB, with no significant difference between MB and either PB or RDV. There was no significant difference between wet and dry sites, and no significant interactions between edaphic conditions and location (Fig. 4.5).

The ANOVA for 2007 C:N showed a significant effect of location ($P = 0.01$) where RDV had a significantly higher C:N than MB and PB, while MB and PB were not significantly different from each other. There was a significant difference between wet and dry sites ($P = 0.02$) and there was a significant interaction between the location and edaphic conditions ($P = 0.02$) with significantly higher C:N in the RDV wet sites compared to the RDV dry sites. At MB and PB there were no differences between wet and dry sites (Fig. 4.5).

The RDV wet sites consistently had the highest C:N ratio for both years (27.70). There were few differences in C:N ratio among sites (MB = 10.92 PB = 10.12, RDV = 20.49), between edaphic conditions, and between years. Therefore C:N ratio may not be a strong indicator for assessing differences soil N status but soil N pools alone, may be more sensitive and reveal more information.

Extractable soil NH_4^+-N and NO_3^- N is a static indicator of the amount of inorganic N in the soil at any one given time. For both 2006 and 2007, extractable soil NO_3^- was consistently below detection limits (detection limit was 0.25 mg L^{-1}). However, NH_4^+ was detectable for 2006 (Fig. 4.6), but not in 2007 and statistical analysis was done for NH_4^+-N for 2006, but not 2007 for this reason. Extractable soil NO_3^- N in the field may therefore not be a strong indicator of N status in the soil, but extractable soil NH_4^+-N may be considered if it is detectable.

The 2006 two-way ANOVA for extractable soil NH_4^+-N showed no significant differences between location ($P = 0.09$), likely due to high variability in NH_4^+ levels among sites and edaphic conditions. There were no significant differences between wet and dry plots across all sites, and no significant interaction among edaphic condition and location. Since soil NO_3^- N was rarely detectable and NH_4^+-N was detectable for 2006, but not in 2007, KCl-extractable inorganic N in the field may not be a very good indicator for soil N status. It should be noted that we only did one KCl extraction per site per summer. Alpine ecosystems have high seasonal variability in N cycling where soil N availability differs during the seasons (Bardgett *et al.*, 2007; Miller *et al.*, 2007 a and b). Soil N may not be as available during this period of time as it is during the spring or fall. Perhaps a series of KCl extractions to coincide with the seasons would be more appropriate when trying to chartacterize soil N status with field KCl extractions.

Inorganic soil N (NH_4^+ and NO_3^-) flux was determined from a dynamic field measurement using PRS™ probes (exchange membranes) placed in each plot for three consecutive seasons (i.e. summer 2006, winter 2006, and summer 2007). The N release rates in µg NH_4^+ and NO_3^- per 10 cm^2 over a standardized 100-day period were analyzed (Fig. 4.7).

The two-way ANOVA for summer 2006 PRS™-probe N values indicated significant differences in total inorganic N mostly due to differences in NO_3^- N with no significant differences in NH_4^+-N. The two-way ANOVA for NO_3^- -N showed a significant difference among locations ($P = <0.0001$) where MB had the most probe-extractable NO_3^- -N (site average 71.28 ± 11.31 µg), RDV was intermediate (site average 41.88 ± 5.70 µg) and PB had the least amount of soil NO_3^--N (site average 5.40 ± 0.92

µg) during the summer growing season. There was a significant difference between wet and dry sites (P = 0.02) but also a significant interaction between location and edaphic conditions (P = 0.0004). The wet plots in MB (95.65 ± 5.40 µg) had more soil NO_3^--N than the dry plots (46.9 ± 4.11 µg), whereas the RDV dry plots (48.17 ± 3.81 µg) had more soil NO_3^--N than the wet plots (35.60 ± 10.42 µg). There were no significant differences in NO_3^--N between the PB wet and dry plots.

Comparing wet sites among locations, they followed the overall location trend with highest NO_3^- -N levels at MB, intermediate NO_3^- -N at RDV, and the lowest at PB. For the dry sites, PB dry plots had significantly lower soil NO_3^- -N than the RDV and MB dry sites which were not significantly different from each other.

For winter 2006/2007, there were no significant differences in PRS™-extractable N among locations or any significant differences between wet and dry sites. A statistical analysis was not done on the summer 2007 PRS™ data because data from all MB sites, all PB dry sites and one PB wet site, and two RDV wet sites are missing. There were not enough data to perform an ANOVA. Fig. 4.7 illustrates the available data.

At both PB and RDV sites, the PRS™ extractable N levels were higher in 2007 than 2006, even though summer 2007 was considered drier.

In this study, PRS™-extractable NH_4^+-N was less sensitive while PRS™-extractable NO_3^- N showed good responses to changes in edaphic conditions and locations, and may be a better indicator of differences in N status. Based on our results, the PRS™ probes seem to be good dynamic indicators for soil N. Putting them out for each season is a practical and inexpensive method for monitoring the changes in N cycling in the soil.

To check whether variability in N status among the wet and dry plots at the three locations were related to variation in soil moisture, simple regressions were performed on inorganic N in soil KCl extractions (2006 only), soil N content, and PRS™-extractable N (per 100 days) against volumetric soil moisture for 2006, 2007, and 2006 + 2007 combined. None of the above regressions showed any significant correlations except the PRS™-extractable N for 2006 ($P = 0.01$ $R^2 = 0.30$) which showed a weak positive correlation with soil moisture. These results suggest that there is no direct relationship between the status of N and the soil moisture regime.

Mineralization and nitrification potential derived from laboratory incubations were used as dynamic indicators of NH_4^+ and NO_3^- release in the soil in a controlled environment. It was hypothesized that the MB sites would exhibit higher nitrification and net N mineralization rates than the RDV sites based on the supposedly higher availability of N in the soils. Incubations were done for 60 days, and soil samples were extracted and analyzed for NH_4^+ and NO_3^- at 0, 30, and 60 days. The soil samples at time zero were extracted in the field and are the KCl soil extractions discussed above.

The 2006 two-way ANOVA for potential net N mineralization rate (Fig. 4.8) showed a significant difference among locations ($P = 0.02$) where PB had the highest mineralization potential, RDV had the lowest, and MB was not significantly different from PB and RDV. There was a significant difference between wet and dry sites ($P = 0.02$) where wet sites generally had higher net N mineralization rates. However, there was also a significant interaction between edaphic conditions and location ($P = 0.001$), where MB and RDV showed no differences between their wet and dry plots, while there was a significant difference between the PB wet and dry plots most likely driving this significant wet-dry effect.

The ANOVA for the 2007 net mineralization potential (Fig. 4.8) indicated a significant difference between locations (P = 0.05), where the PB sites again exhibited greater mineralization potential than RDV, and rates in MB were not significantly different from either PB or RDV. There were no significant differences between wet and dry sites, and no significant interaction between location and edaphic conditions.

There were no differences in net N mineralization potential among the dry sites at the three locations, whereas differences among wet sites were observed in 2006, but not in 2007. The PB sites had the highest net N mineralization potential for 2006 and 2007 driven mostly by the high rates in the wet sites. The mineralization potential notably decreased across all sites from 2006 to 2007 concurrent with a decline in soil moisture.

If net mineralization potential was broken down into net release of NH_4^+-N and NO_3^--N, significant differences were observed for NO_3^--N only. We therefore specifically analyzed nitrification potential for 2006 and 2007 in this study (Fig. 4.9). Since T0 (initial) NO_3^- levels were derived from field KCl soil extractions, and were frequently below the detection limit, it was assumed that the T0 values were equal to zero for both 2006 and 2007. A two-way ANOVA was done on the T1 and T2 NO_3^--N concentrations to see if differences among sites and/or edaphic conditions emerged.

The two-way ANOVA for the 2006 nitrification potential (Fig. 4.9) showed no significant differences after 30 days incubation, but a significant difference among locations was observed after 60 days incubation (P = 0.02), with RDV having the lowest nitrification potential while PB and MB had higher nitrification potential. There were no significant differences in net nitrification potential among wet and dry sites after 30 and 60 days incubations, nor was there a significant interaction between location and edaphic conditions.

The 2007 ANOVA for nitrification potential (Fig. 4.9) again showed a significant difference among locations for the 60 days incubation ($P = 0.04$), but not the 30 days incubation. RDV had a significantly lower nitrification potential than MB, with MB and PB showing higher values that were not different from each other. There was a significant difference between wet and dry sites for the 60 days incubation ($P = 0.01$), but not for the 30 days incubation, where wetter sites generally exhibited higher nitrification potential than dry sites. There was no significant interaction between location and edaphic conditions after either 30 or 60 days of incubation.

The nitrification and mineralization potentials were much lower in 2007 than in 2006. Soil sampling protocols were the same for 2006 and 2007, and all field and lab procedures followed were the same for both seasons. Since soil samples were incubated at field moisture, these differences could be due to annual variability in moisture content. Despite this year-to-year variability in soil moisture content and N mineralization and nitrification rates, the MB sites showed consistently higher nitrification potential than RDV in both 2006 and 2007.

The various indicators of N status collectively are able to distinguish some, but not all, sites in terms of differences in N cycling. Overall, our data showed that MB, characterized by higher atmospheric N inputs, showed signs of relative N enrichment: lower soil C:N ratio, higher overall soil N content, higher extractable NH_4^+-N, more PRS™ extractable inorganic soil N in the summer, but not in the winter, and higher nitrification potential compared to the RDV sites. The RDV location showed characteristics of being a N-poor site with the highest soil C:N ratio, lower KCl and PRS™-extractable inorganic N, and the lowest net N mineralization potential. The PB sites were harder to distinguish because they seem to have high variability in the indicators of N status that are intermediate and not always different from those at the MB and RDV locations. In general, the PB sites showed signs of higher N availability. Thus, with this analysis several indicators of N status were able to distinguish N-rich from N-poor soil characteristics, but not intermediary soil N conditions. In broad terms, data are supportive of our hypothesis that MB is N enriched compared to RDV.

Plant Response: N Status

It was hypothesized that the plant communities in the high N deposition sites (MB) would have more live biomass, more N in roots and foliage, different allocation of above and belowground biomass, and decreased species richness than the sites with lower modeled N deposition (RDV). To test this hypothesis, in summer 2006 all aboveground biomass within a 50 cm^2 frame was removed and separated into live and dead portions, then analyzed for C and N content. Roots were removed from the same soil cores as the soil C and N cores in 2006 and 2007 (to ensure volume and bulk density were the same) and analyzed for C and N, and a full floral survey was done for each plot in 2006 (See appendix 2 and 3 for full floral data and percent cover procedures).

Plant biomass plays an important role in the quantity and quality of SOM and nutrient turnover in an ecosystem (Booth et al., 2005). Alpine ecosystems typically have poor SOM quality leading to the slow breakdown of litter because of harsh climatic conditions such as short growing seasons and cold temperatures (Baron, 1992; Fisk et al., 1998; Burns, 2003). It is important to look at both above and belowground biomass and N content in an alpine system during the peak season of growth. This is because the aboveground biomass is only prevalent for the short growing season and a significant amount of biomass and N is found in the roots (Körner, 2003). Evaluation of the mass and N content of plant biomass can give an indication of litter production and potential soil N input since alpine plants senesce annually (May et al., 1982). This may aid in explaining how much N is being cycled into the soil and where it is being allocated. For this project, the hypothesis was that in the modeled high N deposition sites there would be more aboveground biomass as a reflection of higher N availability to plants, and less N stored in the roots of the plants.

For 2006, two-way ANOVAs were performed on total biomass (i.e. aboveground live+dead+root), aboveground (i.e. live+dead) biomass, and the root mass separately to see if there were any differences between locations and edaphic conditions. There were no significant differences in total plant biomass when above and belowground biomass were combined (MB 401 ± 224 g m^{-2}, PB 453 ± 207 g m^{-2}, RDV 188 ± 178 g m^{-2}) and there were no significant differences in the mass of the aboveground biomass. However, when total biomass was broken down into individual components (i.e. live, dead, and roots), there were significant differences for each component of aboveground and belowground biomass.

The two-way ANOVA for the live aboveground plant biomass (i.e. weight in g m^{-2} Fig. 4.10) showed a significant difference between locations ($P = 0.03$) where MB had more live biomass than RDV, and PB had intermediate amounts that showed no differences with either MB or RDV. There were no significant differences among edaphic conditions or interaction between location and edaphic conditions for the live aboveground biomass. The mass of the dead aboveground biomass were not different among locations, but there was a significant difference between edaphic conditions (P = 0.01) with more dead litter at wet versus dry plots (Fig. 4.10). There were no significant interactions between location and edaphic conditions for the dead biomass.

The two-way ANOVA for the 2006 mass of the roots contained in the top 10 cm of the mineral soil (Fig 4.11) showed significant differences among locations (P = 0.006) with PB (210 g m^{-2} per top 10 cm of soil) having more root biomass than MB (78 g m^{-2} per top 10 cm of soil) and RDV (64 g m^{-2} per top 10 cm of soil), which were not different from one another. There were no significant differences between wet and dry sites, but there was a significant interaction between location and edaphic conditions (P = 0.02) where PB had significantly higher root biomass in the wet plots than the dry plots.

For 2007, the National Park Service did not allow destructive sampling for aboveground biomass, but soil cores were taken and root biomass was estimated. Trends in root biomass were generally similar to those in 2006. However, the two-way ANOVA showed that there were no significant differences in root mass for 2007 (P = 0.3, Fig. 4.11).

Simple regressions were performed on gravimetric soil moisture content versus aboveground and belowground biomass to see if there were any correlations between moisture and biomass. The gravimetric soil moisture content was averaged for both years, then compared to plant biomass. For both the aboveground and belowground biomass, there were no significant correlations between soil moisture and biomass (R^2 = 0.05 for both aboveground and belowground biomass).

The N content in the foliar and root biomass (Fig. 4.12 and 4.13) were considered in this study as potential indicators of N status. The hypothesis was that the modeled high N deposition sites (MB) would have higher above and belowground N content than the low N deposition sites (RDV), but that less N would be allocated to the belowground biomass.

The two-way ANOVA for the 2006 live aboveground N content showed significant differences between locations ($P = 0.01$) with MB having the highest live foliar N content (MB 4.93 ± 3.0 g m^{-2}) and no differences between PB (2.34 ± 1.4 g m^{-2}) and RDV (1.71 ± 1.0 g m^{-2}; Fig. 4.12). There were no significant differences between edaphic conditions and no interaction between location and edaphic conditions. For all locations, the aboveground N content followed the biomass with the PB location intermediate for both biomass and live N content.

The two-way ANOVA for the 2006 dead aboveground N content showed no significant differences between locations. There were no trends observed between biomass and dead N content for the three locations. There was a significant difference between edaphic conditions ($P = 0.03$) with the wet sites having more overall N content in the dead foliage than the dry sites. There were no significant interactions between edaphic condition and location. The two-way ANOVA for the root N content in 2006 and 2007 showed no significant differences between sites or edaphic conditions (Fig. 4.13).

In addition, a simple regression was done between volumetric soil moisture and N content of aboveground foliage, belowground roots, and both above and belowground plant N content for 2006 to see if soil moisture affected content and/or distribution of N in the plants. Furthermore, simple regression was done for volumetric soil moisture content, root N content for 2006, 2007, and 2006+2007 to see if variability in soil moisture could account for the observed variability in the N content of the roots among sites, edaphic conditions, or sampling years. None of the variables significantly correlated with soil moisture content for either or both years.

The C:N ratios for aboveground live, dead, and roots were a measure to assess translocation of N in plants, and whether or not more N was stored in foliage, litter, or roots. The two-way ANOVA for 2006 data showed no significant differences in C:N for aboveground live and dead foliage, and roots. However, in 2007, there was a significant difference in the root C:N ratio ($P = 0.02$) with RDV (C:N ratio 23) having a higher root C:N ratio than PB (C:N ratio 12) or MB (C:N ratio 11, Fig. 4.14). The C:N ratios for MB and PB were consistently around 10-12 for both 2006 and 2007, but the RDV C:N ratio for roots was higher in 2007 than 2006 with no significant differences between edaphic conditions.

Allocation of plant N in 2006 was used as an indicator to see if different N availability caused differences in how much N was allocated in belowground versus aboveground biomass. If the ratio was low, this would suggest preferential N allocation to foliar biomass, and if the ratio was high, it would suggest preferential N allocation to root biomass. A simple regression of root N content versus foliar N content suggested that there was no significant correlation between above and belowground N content. The ratios for root:foliar N content were 574 for MB, 1594 for PB, and 4635 for RDV. A two-way ANOVA was performed on the root:foliar N content to see if there were statistical differences across locations and edaphic conditions since differences in root:foliar N content were empirically observed. There were no significant differences among location or edaphic conditions, and no significant interactions between location and edaphic conditions.

Live:dead N content ratios were examined because a high ratio might suggest that the plant, due to low soil N, removes N from its foliage before it senesces, therefore litter will be low in N. A low live:dead N content ratio may suggest that the plant does not remove N from its foliage before senescence due to enough soil and root N storage. The ratios for live:dead N content of plants were 42 for MB (NOTE: MBD1B was an outlier with a ratio of 367. When the outlier was removed, the live:dead N content had a mean of 13), a live:dead N content of 4 for PB, and RDV had 7.5 live:dead N content. A simple regression was done on alive vs. dead N content which had a significant correlation ($P = 0.001$, $R^2 = 0.35$). This suggests that the N content of the dead litter simply followed the N content of the live foliage. A two-way ANOVA was performed on the live:dead N content to see if there were differences across locations and edaphic conditions since there appeared to be differences, and there was a correlation between live:dead N content. There were no significant differences across location or within edaphic conditions.

The MB sites had the highest foliar N, total, and available soil N, but litter N content for all three locations were not significantly different. This indicates that MB may have sufficient supplies of N and the plants at that location may be exhibiting “luxury N uptake” described in a study that Bowman (1994) did in alpine communities where he fertilized plots and removed N as a limiting factor in the ecosystem. (Bowman, 1994). It is possible that MB is increasing aboveground biomass because N is no longer a limiting factor in the soil. This may be a reflection of MB receiving more N into its location in the form of precipitation. Considering the above markers for N status, MB consistently had higher values than RDV. Thus both soil N and plant biomass N content values, in this case, could detect changes of N status. However, C:N ratios in soil and plants were less informative.

The 2006 plant survey assessed percent cover in the 50 cm^2 areas prior to the destructive sampling of the aboveground biomass (Table 4.2). All vegetation was then clipped to the bare ground and separated into live and dead portions, then weighed after being dried. The two-way ANOVA results showed a significant difference among locations ($P = 0.001$) with plots at PB (44.58 ± 13.5%) having more plant cover than at MB (21.96 ± 4.0%) and RDV (13.58 ± 3.6%) which were not different in plant cover. There was a significant difference between wet and dry conditions ($P = 0.001$) where wet edaphic conditions generally had more plant cover than dry edaphic conditions, but there was also a significant interaction between location and edaphic conditions ($P = 0.002$). These differences were mainly driven by the PB site: the PB wet plots had more cover (72.08 ± 12.4%) than the PB dry plots (17.08 ± 1.8%). There were no significant differences in cover among the wet and dry plots in MB and RDV locations (Table 4.2).

The MB and RDV locations had similar vegetation communities consisting of a mix of forbs and woody plants giving similar cover characteristics. MB consisted of a mix of sparse vegetation and heath for the dry sites, and a mix of forbs and grasslands for the wet sites. RDV had a mix of sparse vegetation and forbs for the dry sites, and a mix of forbs and dwarf shrubs for the wet sites. The MB sites had significantly more aboveground biomass than PB or RDV, but not significantly more cover. The PB location was different in vegetation in that it had bare ground with sparse vegetation in the dry sites, and had grassland/graminoid communities for the wet sites. The graminoid community in the PB wet plots could account for the significant difference in cover.

A floral survey was done in summer of 2006 and 2007 within a 1m x 1m area at the center of each plot (see appendix 3 for full floral survey data) to see if species composition and richness differed between the modeled high N deposition sites (MB) and the low N deposition sites (RDV). Since the majority of alpine plants are perennial, there were little, if any differences in plant composition from 2006 to 2007. To best represent the diversity in the wet and dry communities for each location, the six wet plots and six dry plots were combined to create a 6m x 6m macroplot to represent one wet and one dry community at each location.

The floral survey counted a total of 85 species (in 2006 and 2007) at all three locations across the wet and dry communities. Of the 85 species counted, 59 species were found at MB, 41 species at PB, and 68 species at RDV. The most abundantly occurring species were *Poa cusiskii epilis* and *Sibaldia procumbens* which was observed in 5 of the 6 wet and/or dry community plot locations (Fig. 4.15). *Poa cusickii epilis* was found in all but the RDV wet sites, and *Sibaldia procumbens* was found in all but the MB dry sites. These species had a high frequency, but they were not the dominant species in all the sites.

Percent cover by plant species was used to calculate species evenness and identify the dominant species at each site. Table 4.3 summarizes the species richness, dominant species in each location and edaphic condition, and estimated mean (based on cover category ranges) for cover in each macroplot. The high estimated cover for *Poa cusickii epilis* (79.17%) found at the PB wet sites confirms that the PB plant communities are primarily grass communities affecting plant cover. Aside from MB and RDV dry sites (being dominated by shrub and forbs), all the sites appear to be dominated by graminoids (i.e. rush, sedge, grass). Although it is not uncommon for alpine ecosystems to be graminoid-dominant, there are several cases in which excessive N is predicted to decrease plant diversity with an eventual increase in graminoid species (Suding et al., 2006; Bowman, 2000; Näsholm, 1997; Seastedt and Vaccaro, 2001; Fenn *et al.,* 1998; Blett, 2004). Monitoring dominant cover for species may be a good indicator for changes in N status. However, this is a long term indicator and goes beyond the scope of this project.

The Shannon-Weiner index was calculated for each macroplot from the floral survey data and the values were used in a two-way ANOVA to test species richness differences among location and edaphic condition (Fig. 4.16). Since there were no differences in plant composition between 2006 and 2007, the 2006 data was further analyzed. There were no significant differences in Shannon-Weiner index numbers among locations or edaphic conditions. Although there were no statistically significant differences in species diversity, RDV wet sites had the highest diversity (Shannon-Weiner = 1.3), while MB wet sites had the lowest (Shannon-Weiner = 1.06). For evenness (table 4.3), the percent cover for each species was compared to the total cover for the 6 m^2 macroplot. All locations had an even distribution of species (evenness number = 0.02), except for MBW (eveness number = 0.03) and PBW (evenness number = 0.04) where both had a dominant species of plant that had a much higher cover than any other species in the plot. Based in our data, in this system, species richness and diversity may not be a good short-term marker to detect the effect of small differences in N deposition.

Discussion

The total ecosystem N pool for the Tetons across locations had an average of 122.4 g N m^{-2}. Most N (85-95%) is typically found in the organic layer in alpine soils (Fisk et al., 1998). Of the total N pool, 8.81% was found in aboveground biomass (6.62% live 2.19% dead), 18.12% in root biomass, and 73.07% was found in soil N. The soil N concentration in our Teton sites was 0.26-0.31% (Table 4.5). Other sites in the Rocky Mountains have 0.3-0.6 % soil N in the Medicine Bow Mountains of Wyoming (Welker et al., 2004), and 0.97-1.36% at Niwot Ridge on the Colorado Front Range (Steltzer and Bowman, 1998; Baron et al., 2000). Root N content for the Tetons was 0.4-1.9% and for Niwot Ridge, it was 0.05-1.5% (Steltzer and Bowman, 1998) which compares well to the Teton data.

The C:N ratios of alpine soils differ among plant community type. Soils in alpine communities with sparse vegetation exhibit C:N ratios between 5-7, alpine grasslands show ratios between 8-28, and dwarf shrub and heath soils have 12-40 C:N ratios (Körner, 2003). The 2006 C:N ratios for this study show MB wet 11.09 MB dry 10.98, PB wet 6.52 PB dry 9.26, and RDV wet 21.90 RDV dry 15.98. All the locations had soil C:N ratios consistent with those reported for the appropriate alpine plant communities except RDV site, especially dry sites, which seemed to be a little higher than the expected 5-7 for sparse vegetation. High C:N ratios typically suggest soils low in N content (Conley et al., 2000) and soil organic matter. Perhaps microclimate constraints are limiting soil N more than normal at the Tetons.

The soil C:N ratios for Niwot ridge were between 25-32 (Table 4.5). The Tetons had C:N ranging between 7-20. The Colorado Front range receives more N deposition (9 kg N $ha^{-1}yr^{-1}$) than the Tetons (0.1-2.5 kg N $ha^{-1}yr^{-1}$ Bowman and Steltzer, 1998; Nanus et al., 2003). Therefore, the C:N ratios should be smaller than the Tetons. Niwot could be experiencing higher N immobilization and increases in NPP as described by Aber and others (1989, 1998) in the initial stages of N-saturation.

Low, but measurable concentrations of extractable NH_4^+ -N are typical in alpine soils during the season of highest plant biomass (Stark and Hart, 1997; Miller and Bowman, 2003; Makarov et al., 2010). Various studies of extractable NH_4^+-N soil concentrations in alpine systems during the growing season world-wide show values of 12-40 µg g^{-1} NH_4^+ -N in Russia, (Makarov et al., 2010), 3.6 µg g^{-1} in the Alps (Haselwandter et al., 1983), 10-20 µg g^{-1} NH_4^+-N in a wet sedge community in the arctic of Canada (Edwards et al., 2006), 0.9-9.6 µg g^{-1} in Alaska (Kielland, 1995), and 0-1.5 µg g^{-1} NH_4^+-N at Niwot Ridge in the Rocky Mountains (Lipson et al., 1999; Miller and Bowman, 2003; Schmidt et al., 2004). The Tetons showed 0.06-0.39 µg g^{-1} NH_4^+ -N which is in the low range of values overall for alpine and arctic systems (Table 4.5).

Based on exchange resins, the Tetons had 4.9-5.4 µg g^{-1}(top 10 cm soil per 100 days) of available N. Since methods for analyzing in situ availability rates of soil N differ so much, it is difficult to compare numerical values to other studies for this assessment. However, net mineralization and nitrification potential through laboratory incubations is a more appropriate and suitable way to make comparisons of a wide range of data. The Tetons had a net mineralization rate of 6-55 µg N g^{-1} and a net nitrification rate of 2-37 µg NO_3^- N g^{-1} after 60 days incubation. Niwot Ridge showed a net mineralization of 40-

200 µg N g^{-1} and 4-34 µg NO_3^- N g^{-1} per 60 days of incubation (Fisk et al., 1998; Baron et al., 2000; Bowman et al., 2006). While mineralization is higher at Niwot Ridge, both sites have similar nitrification potential (Table 4.5).

Plant biomass in alpine ecosystems around the world range from 170-1070 g m^{-2} for the Austrian Alps (Brzoska, 1969), 1740 g m^{-2} for the New Zealand Alps (Meurk, 1978), 100-400 g m^{-2} for the South-central Himalayas (Rikhari et al., 1992), 110-350 g m^{-2} for the Medicine Bow Mountains in southwestern Wyoming (Bliss, 1966), 100-250 g m^{-2} for the Central Rocky Mountains (Walker et al., 1994), and 50-150 g m^{-2} for Niwot Ridge along the Colorado Front Range (Theodose et al., 1996; Fisk, et al., 1998; Bowman et al., 2006). If broken down into wet and dry edaphic conditions, Niwot Ridge has an average of 155 g m^{-2} in the dry communities and 291 g m^{-2} live biomass in the wet communities (Fisk et al., 1998). The Tetons have 80-240 g m^{-2} in the wet and 80-255 g m^{-2} in the dry communities (Fig. 4.10). Overall, the Teton aboveground biomass compares well to the Central Rockies and typical alpine ecosystems throughout the world (Table 4.5).

Table 4.6 makes a detailed comparison of aboveground and belowground biomass and N content in wet and dry alpine communities at the Tetons compared to Niwot Ridge in the Colorado Front Range (Fisk et al., 1998). The wet and dry sites for both locations had similar site characteristics in that both Teton and Niwot Ridge dry communities where on exposed upper topographic areas with sparse vegetation (Forb and lichen dominant for Tetons and tussock forming sedges for Niwot Ridge). The wet communities were both low topographic areas with wet soils throughout the growing season due to snowmelt. Both wet communities at the Tetons and Niwot Ridge were dominated by sedges and rushes.

Niwot Ridge has much more dead biomass than the Tetons. The wet sites at Niwot Ridge seem to have twice as much overall biomass (mostly in the roots) than the wet sites at the Tetons, but the dry sites seem comparable. The N content of roots and aboveground biomass are similar for both wet and dry sites at both locations. Plant litter seems to accumulate more at Niwot Ridge than in the Tetons. This could possibly explain the higher mineralization rates at Niwot Ridge.

For species richness, the Tetons have an average of 6-20 species m^{-2} while Niwot Ridge has 12-29 species m^{-2} (Bowman et al., 2006). The input of N influences plant productivity and composition (Blair et al., 1998), and in alpine ecosystems, this overall N input is highly correlated to snowpack and topography (Fisk et al., 1998; Williams et al., 2003). This could explain why the wet sites in both the Tetons and Niwot Ridge have higher overall biomass than the dry sites.

Overall, when assessing N status in alpine ecosystems, wet sites have much more variability in the ecosystem markers considered in this study than the dry sites (which showed little, if any, differences among locations). This seems to be a trend observed in other alpine ecosystems in the Rocky Mountain Range (Fisk et al., 1998; Steltzer and Bowman, 1998; Bardgett et al., 2007). Therefore, when assessing N in alpine ecosystems, greater variation in wet sites implies that these wet sites may be more responsive to changes in N input, while dry site responses are more controlled by limitations in water availability.

Table 4.1 Dates and temperatures (C°) of the soil (at 10cm depth) based on the date when the temperature increased from 0° C indicating the onset of the snow free period.

Location	WET		DRY	
	Date	Temperature C°	Date	Temperature C°
Moose Basin (MB) 1	NO DATA	NO DATA	5/28/2007	2.02
Moose Basin (MB) 2	NO DATA	NO DATA	5/12/2007	1.59
Moose Basin (MB) 3	6/4/2007	3.06	5/13/2007	2.38
Paint Brush Canyon (PB) 1	6/29/2007	2.85	NO DATA	NO DATA
Paint Brush Canyon (PB) 2	6/22/2007	1.07	NO DATA	NO DATA
Paint Brush Canyon (PB) 3	6/16/2007	1.31	6/16/2007	2.14
Rendezvous Mountain (RDV) 1	6/19/2007	2.14	6/16/2007	5.01
Rendezvous Mountain (RDV) 2	6/14/2007	3.44	6/27/2007	4.57
Rendezvous Mountain (RDV) 3	6/13/2007	3.65	6/20/2007	4.97

Table 4.2. Mean percent cover with standard deviation for 50 cm^2 frames within each edaphic condition. Capital letters represent differences among locations. * Represents the interaction driving the significant differences among sites and edaphic conditions.

Location	Combined Plots		Wet % Cover	Dry % Cover
Moose Basin	21.6 ± 4.01	**B**	28.33 ± 4.91	15.58 ± 3.94
Paint Brush	44.58 ± 13.52	**A**	72.08 ± 12.42 *	17.08 ±1.82
Rendezvous Mtn.	13.58 ± 3.62	**B**	11.58 ± 2.81	15.58 ±7.31

Table 4.3 The number of species, quantitative distribution of species, dominant species, and the estimated median percent cover for dominant species occurring in each community. NOTE: PBD had a type of lichens as a dominant cover (20.83%). Since lichens is not considered a vascular plant species, it was omitted and the next dominant plant species cover was used.

Location	Number of Species (out of 85 total)	Mean Evenness (based on sp. Cover & total cover)	Dominant Species	Median Cover of Dominant Species (%)
MBW	23	0.03	Juncus parryi	15.82
MBD	36	0.02	Potentilla fruticosa	25.83
PBW	22	0.04	Poa cusiskii epilis	79.17
PBD	19	0.02	Juncus Parryi	3.06
RDVW	35	0.02	Carex breweri paddoensis	11.67
RDVD	33	0.02	Rumex paucifolius	15.83

Table 4.4 Comparison of soil N, C:N, extractable NH_4-N, net mineralization, and plant biomass of the Tetons with other alpine ecosystems in the Rocky Mountains and around the world.

Location	Soil N Conc. (%)	Soil C:N	Extractable NH_4-N ($\mu g\ g^{-1}$)	Net Mineralization Rate (μg N g^{-1})	Plant Biomass (g m^{-2})	Source
Tetons	0.2-0.3	7-20	0.06-0.39	6-55	80-240	This Study
Medicine Bow Mtns. (Wyoming)	0.3-0.6				110-350	Welker *et al.*, 2004, Bliss1966
Niwot Ridge (Colorado)	0.9-1.36	25-32	0-1.5	40-200	50-150	Steltzer and Bowman 1998, Baron *et al.*, 2000, Lipson *et al.*, 1999, Miller and Bowman 2003, Schmidt *et al.*, 2004, Fisk *et al.*, 1998, Bowman *et al.*, 2006, Theodose *et al.*, 1996
Central Rocky Mtns.					100-250	Walker *et al.*, 1994
Russia			12-40			Makarov *et al.*, 2010
Austrian Alps			3.6		170-1070	Haselwandter *et al.*, 1983, Brzoska 1969
Canada			10-20			Edwards *et al.*, 2006
Alaska			0.9-9.6			Kielland 1995
New Zealand					1740	Meurk 1978
Himalayas (South Central)					100-400	Rikhari *et al.*, 1992

Table 4.5 Comparison of aboveground and belowground biomass and N content between the Tetons and Niwot Ridge along the Colorado Front Range. *NOTE: Dead plant biomass for Niwot Ridge accounted for 3-4 times more biomass and N than the live portions of plants and roots. Dead plant matter was estimated from this observation and omitted from total biomass and N as actual numbers were not reported (Fisk *et al.,* 1998)

Biomass and N Content		Tetons		Niwot Ridge	
	g m^{-2} and g N m^{-2}	Wet	Dry	Wet	Dry
Aboveground	Live biomass	171	152	291	155
	Dead Biomass	110	27	~1595*	~840*
	Live N content	3.52	2.5	6.4	2.4
	Dead N content	1.43	0.34	~25*	~10*
Belowground	Root biomass	153	89	364	125
	Root N content	3.96	2.93	4.7	2
Total	Above Live + Root Biomass	324	241	655	280
	Above Live + Root N Content	7.48	5.43	11.1	4.4

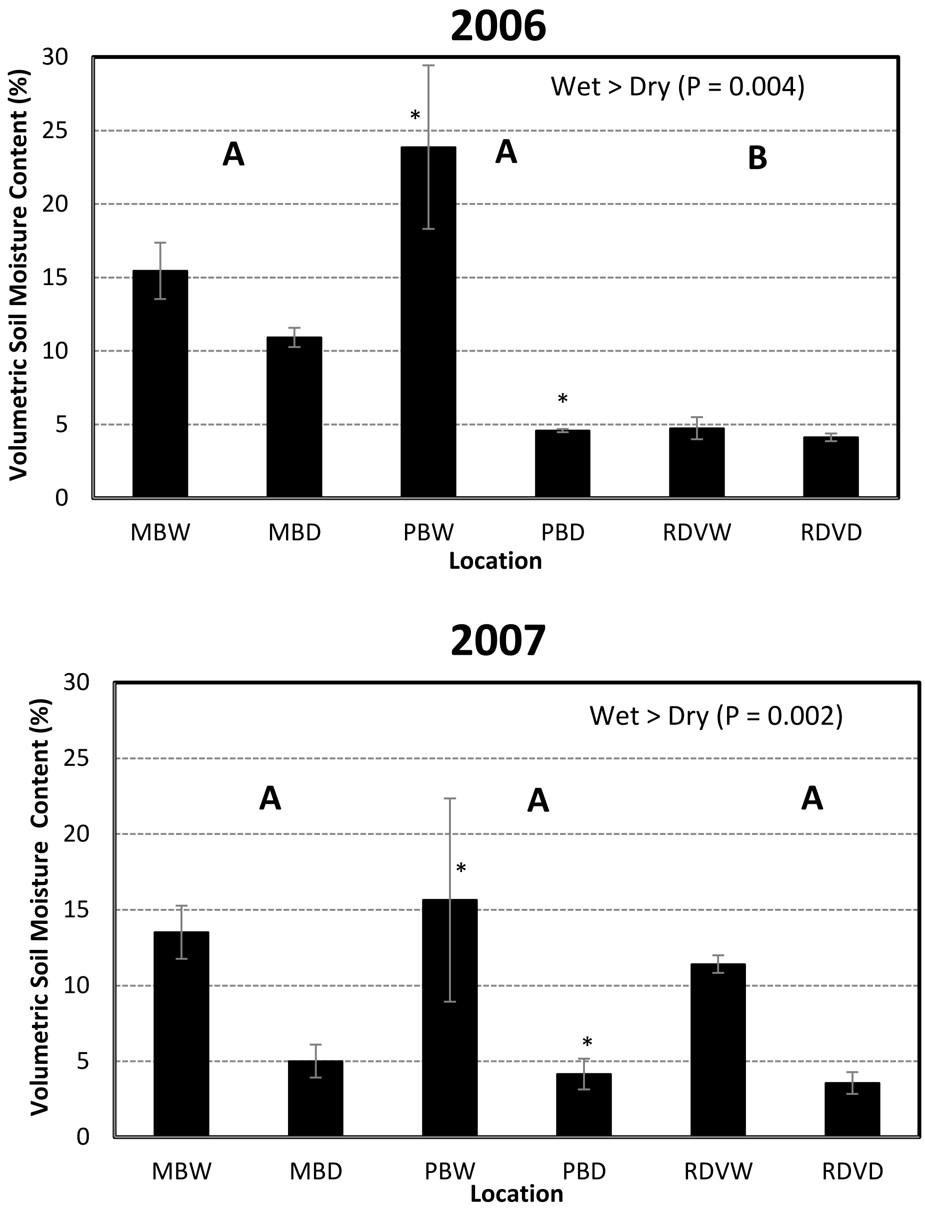


Fig. 4.1. Volumetric soil moisture for summer 2006 and 2007. Capital letters represent Tukey post-hoc test for differences between locations. * Represents significant differences within locations.

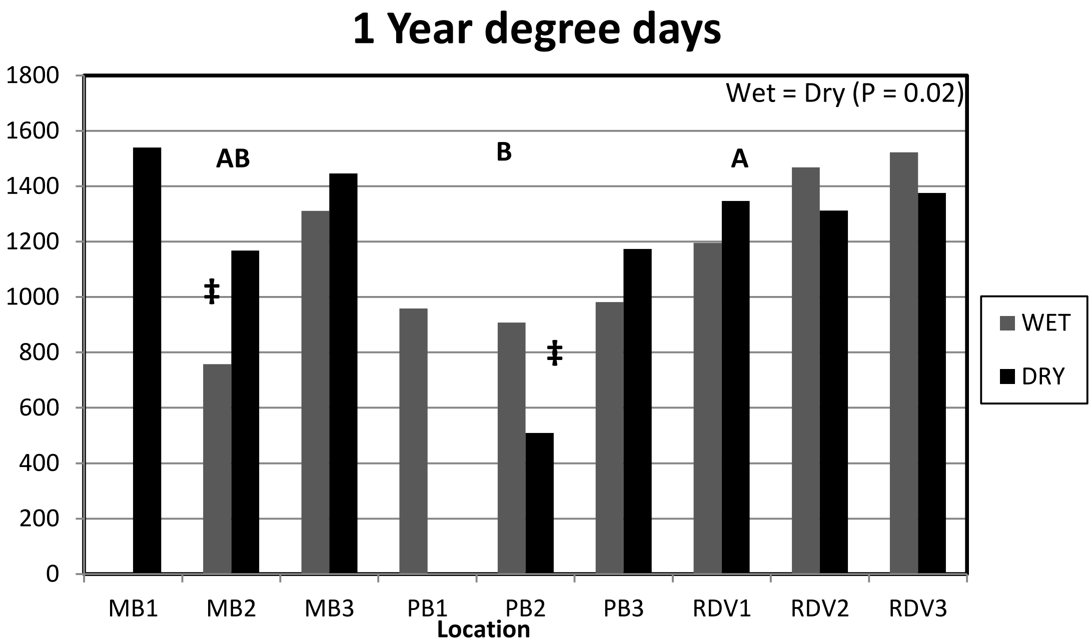


Fig. 4.2. Annual degree days from 7/10/2006-7/10/2007 for MB and RDV, and 8/14/2006-8/14/2007 for PB for each site. Capital letters represent differences between locations. (MBD1 and PBW2 have no data). ‡ MBW2 has data from 7/10/2006-10/30/2007 and ‡ PBD2 has missing data from 1/24/2007-5/31/2007.

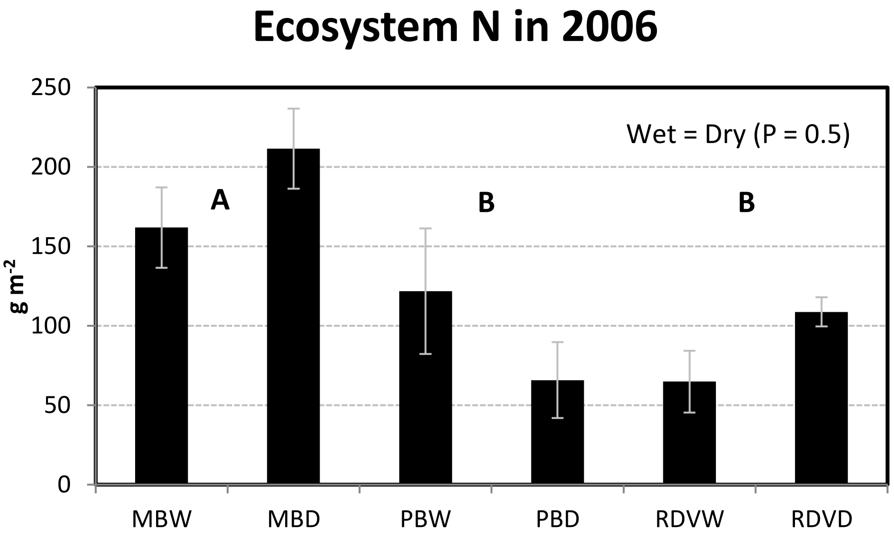


Fig. 4.3. Total N content consisting of live and dead foliage, root and soil N in each site (0-10 cm depth). Capital letters represent significant differences between locations.

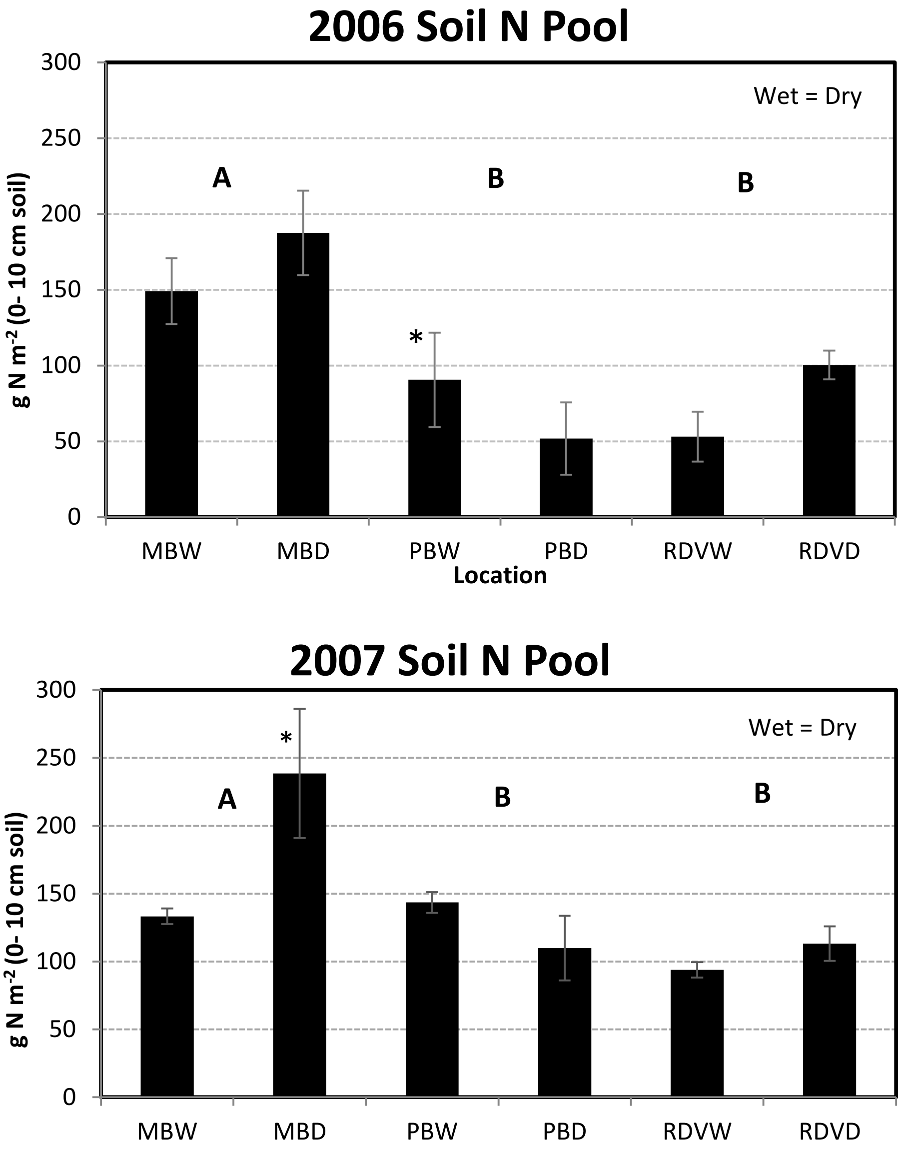


Fig. 4.4. Total soil N from 2006 and 2007. Capital letters represent differences between location. * Represents significant differences between wet and dry plots within location.

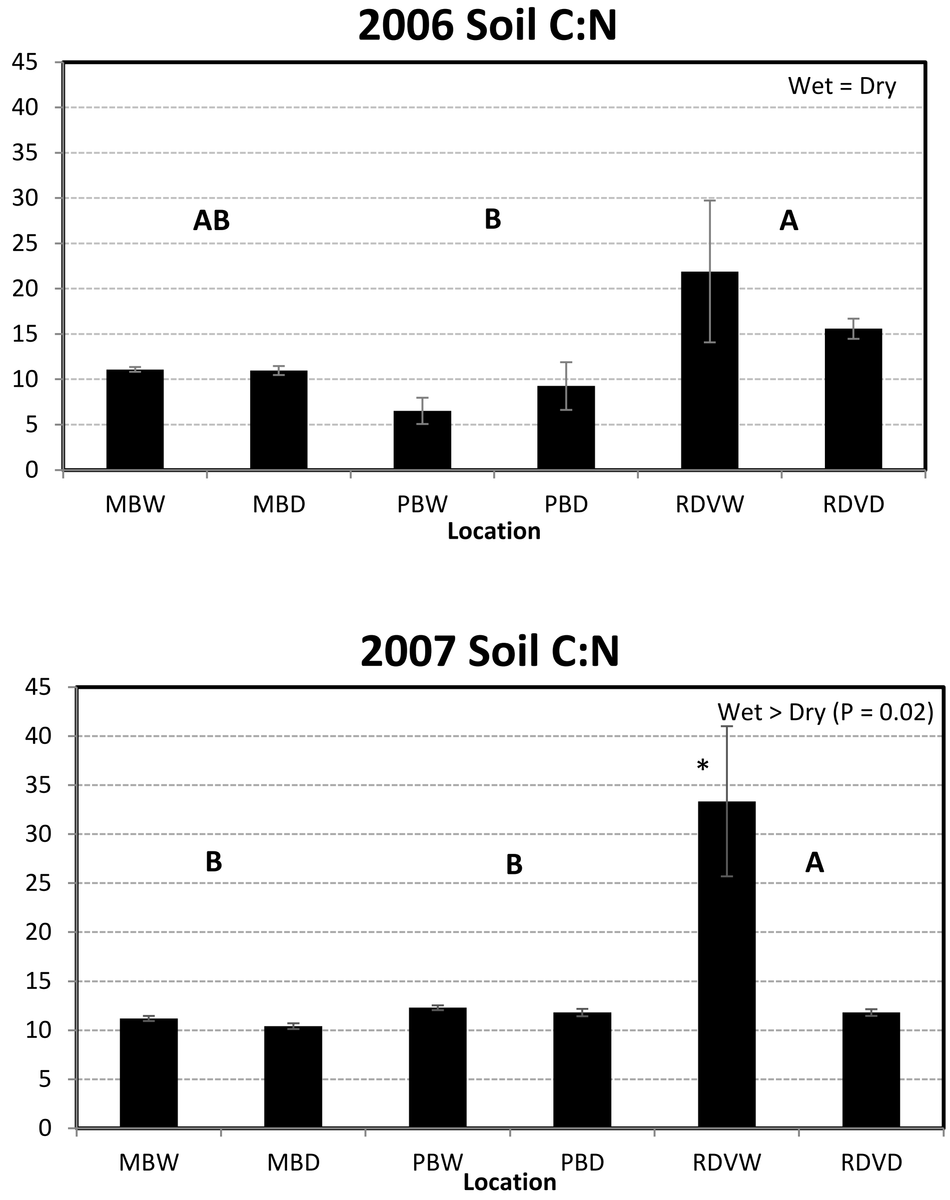


Fig. 4.5. 2006 and 2007 C:N ratios in g m^2 (0-10cm of soil). Capital letters represent differences among location. * Represents significant differences between wet and dry plots within location.

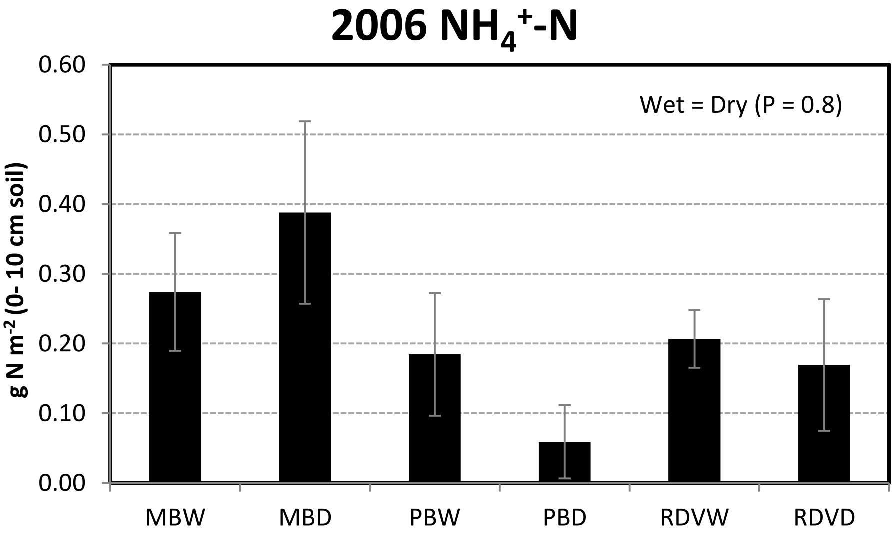


Fig. 4.6. Summer 2006 NH_4^+-N from field KCl extractions.

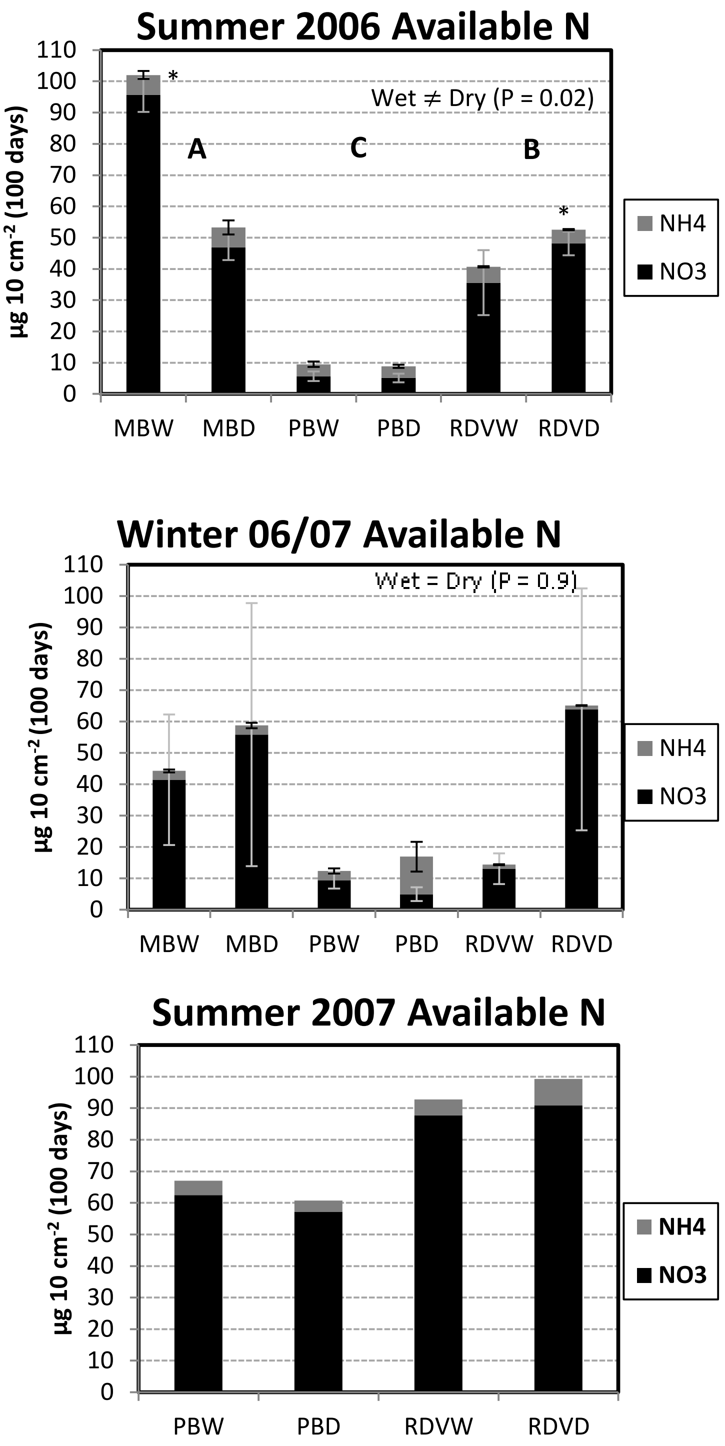


Fig. 4.7. Summer 2006, winter 2006/2007, and summer 2007 PRS-extractable inorganic N over a standardized 100 day period. Capital letters represent differences in NO_3^--N among location. *Represents differences in wet and dry NO_3^--N within sites. There is no data for MB and PBD for summer 2007.

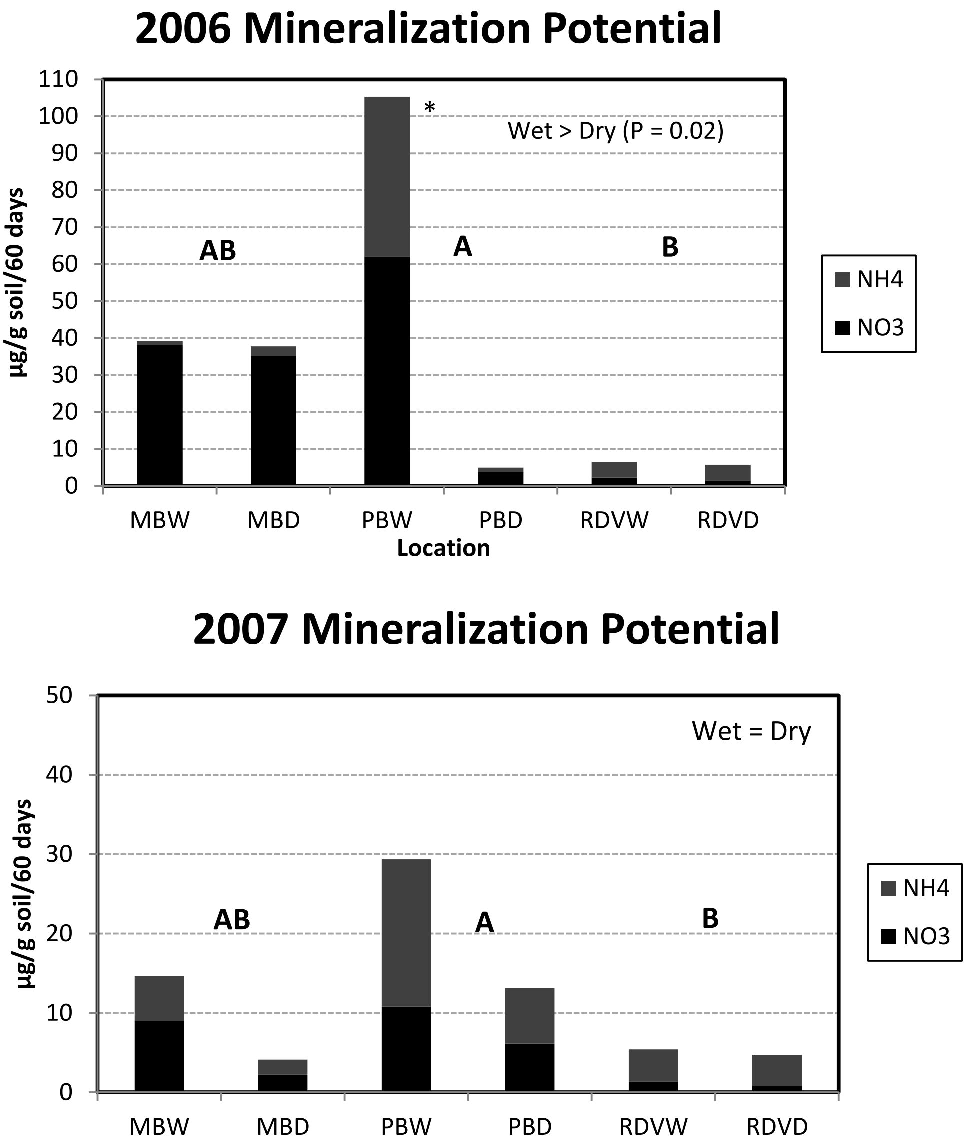


Fig. 4.8. 2006 and 2007 net N mineralization potential after 60 days of laboratory incubation. Capital letters represent differences among location. NOTE: for 2006, differences are for NO^-_3-N only and for 2007, differences are for net N Mineralization. * Represents differences within wet and dry plots.

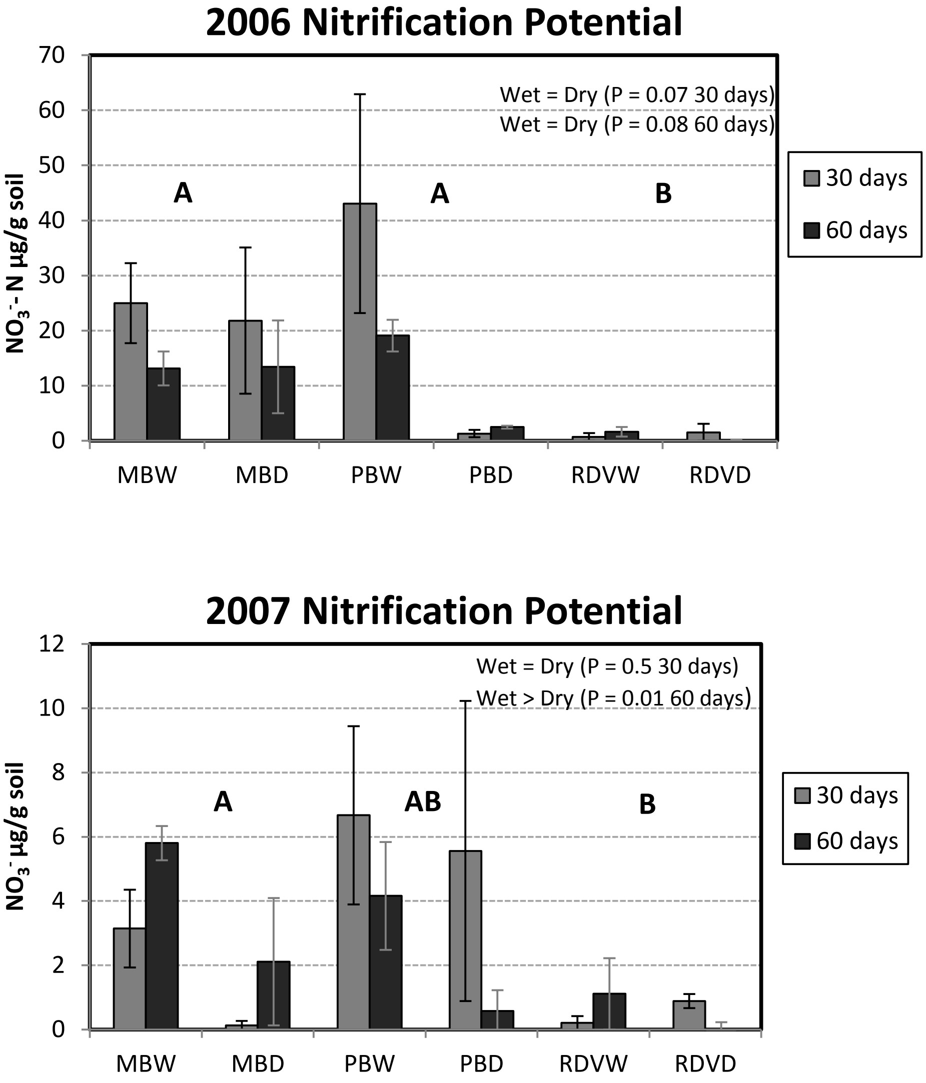


Fig. 4.9. 2006 and 2007 net nitrification potential after 30 and 60 days of laboratory incubation. Captial letters represent differences among location after 60 days incubation. For 2007, all wet plots were higher than all dry plots.

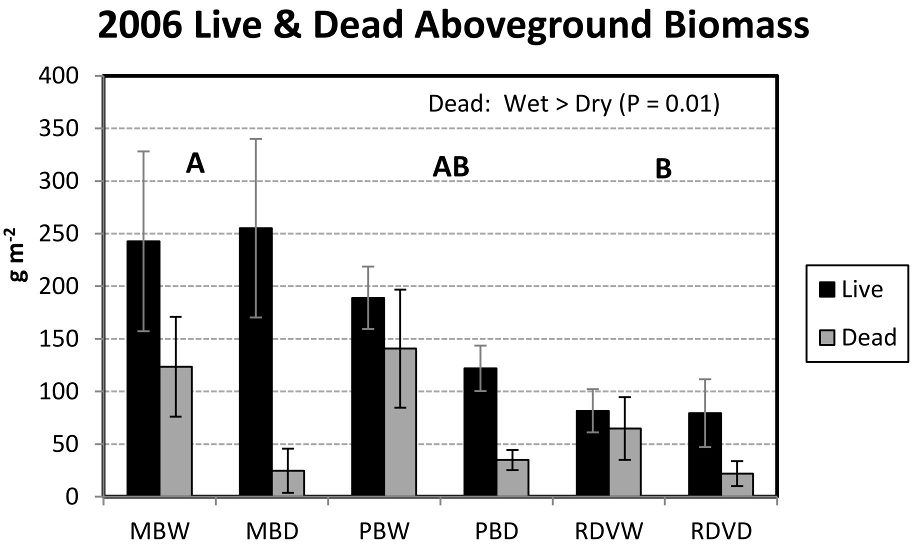
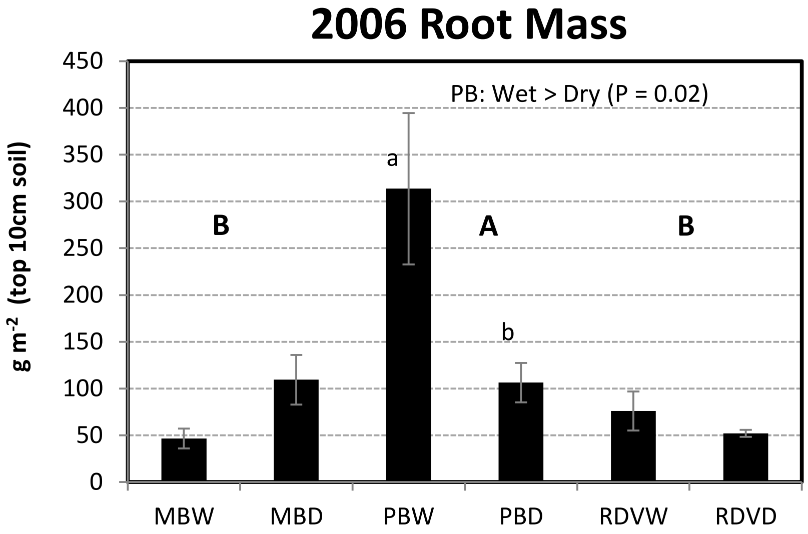


Fig. 4.10. Summer 2006 live and dead vegetation biomass (g m^{-2}). Capital letters represent significant differences in location for live biomass ($P = 0.03$).



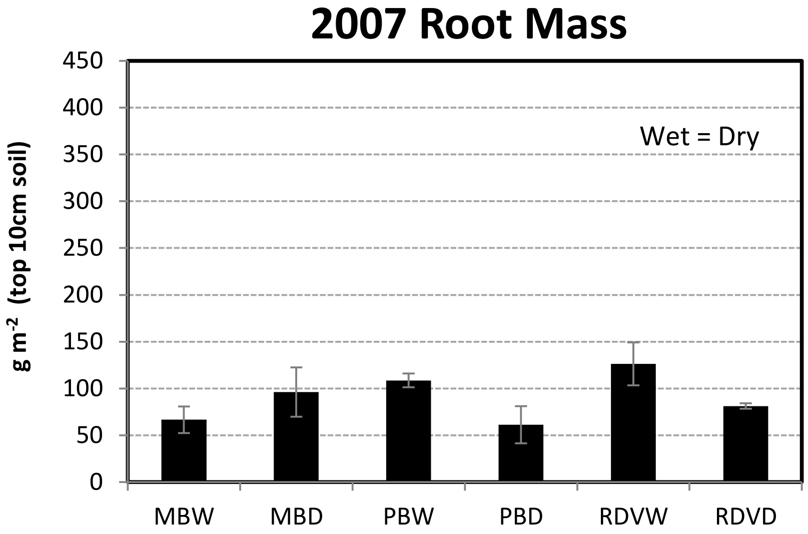


Fig. 4.11. 2006 and 2007 root mass. Capital letters represent differences among location.

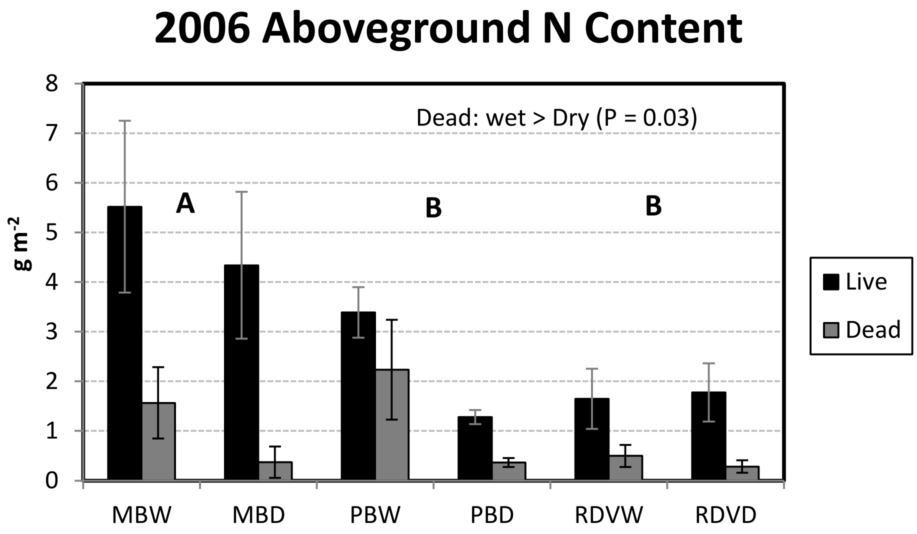
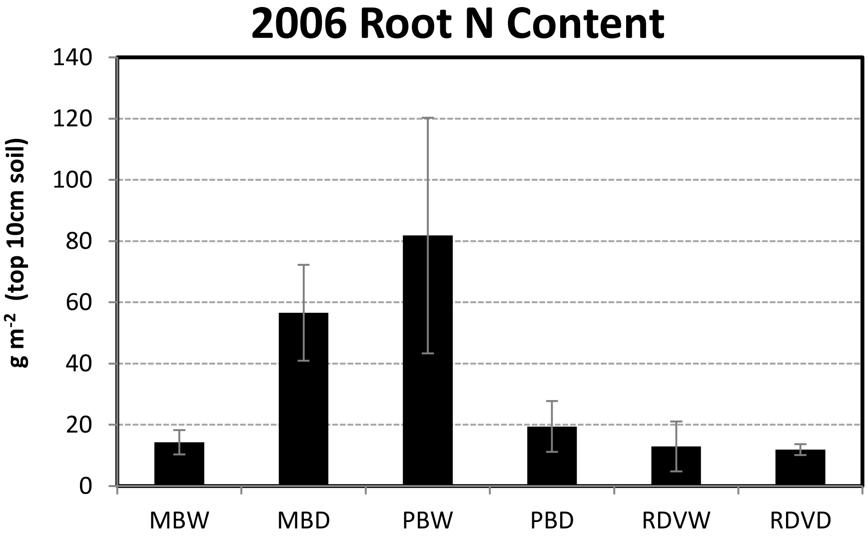


Fig. 4.12. 2006 nitrogen content with live and dead foliage considered. Capital letters represent differences among location for Live foliage.



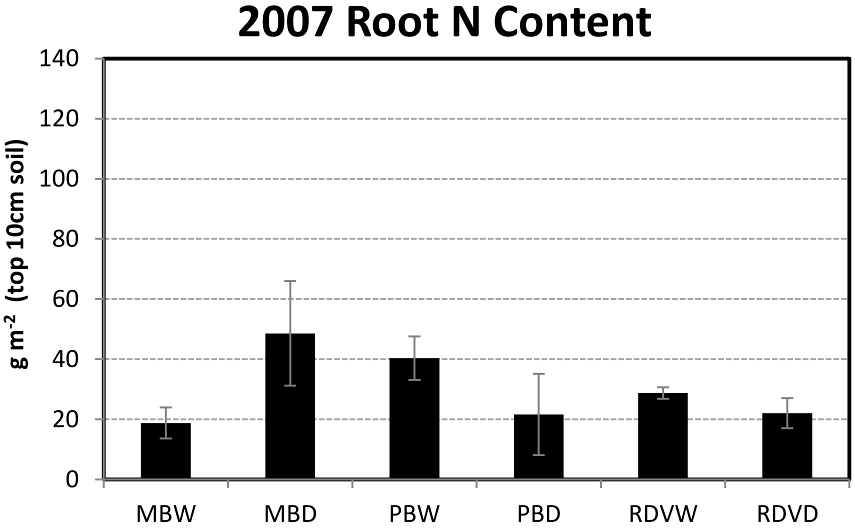


Fig. 4.13. 2006 and 2007 N content of roots.

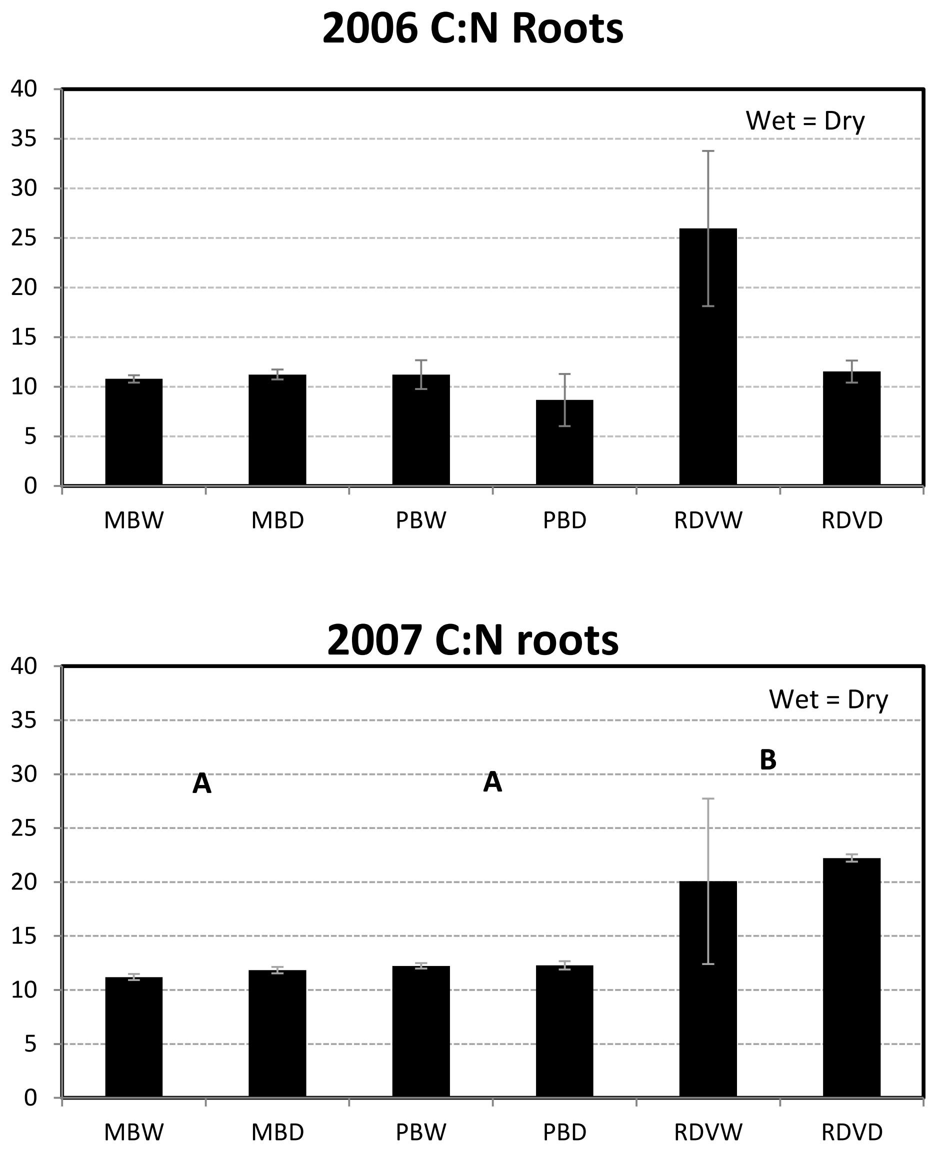


Fig. 4.14. 2006 and 2007 C:N ratios for the root biomass. Capital letters represent differences among location.

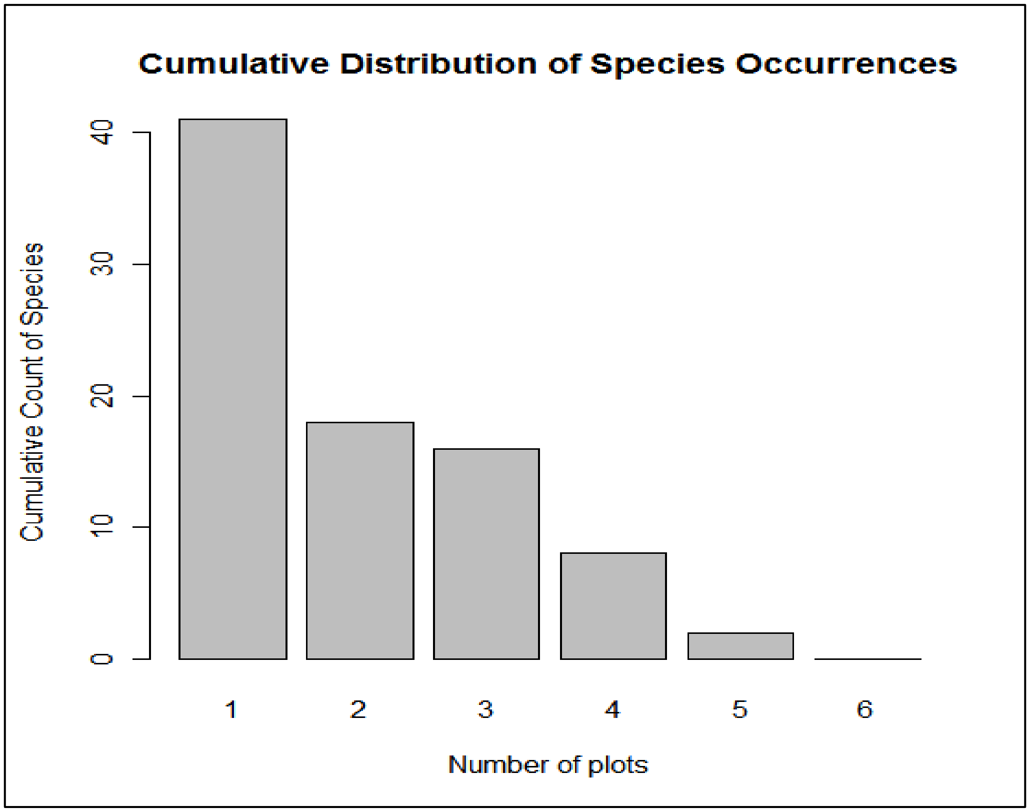


Fig. 4.15. Cumulative frequency for species occurrence and distribution observed across the six macroplots. Out of a total of 85 species observed, 41 species occurred in one plot only, 18 species occurred in two plots, 16 species occurred in three plots, 8 species occurred in four plots, and 2 species, (*Sibaldia procumbens* and *Poa cusiskii epilis*) occurred in five plots. There were no species observed that occurred in all six of the plots.

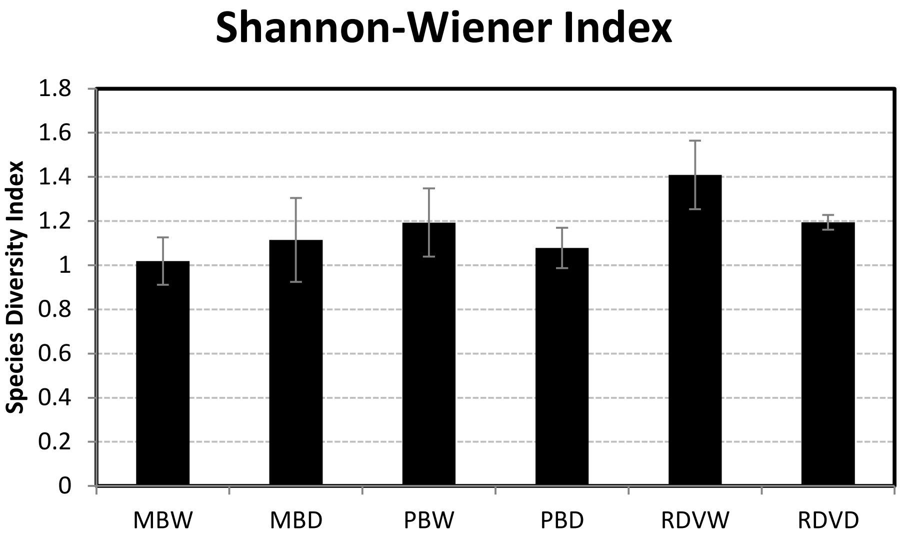


Fig. 4.16. Shannon-Wiener diversity index for species richness.

CHAPTER 5

CONCLUSIONS

The main objective of this study was to assess the “hot spot” of N deposition that were assumed to exist in the Grand Teton National Park (consisting of 0.1-2.5 kg N ha^{-1} yr^{-1} of N deposition) from the modeled Nanus deposition maps for the Rocky Mountains (Nanus et al., 2003), and how this affected soil and plant characteristics in alpine ecosystems.

The hypothesis that a N deposition gradient existed from north to south was inconclusive. Based on our data, the Tetons receive an average of 1.42 kg N ha^{-1}yr^{-1} with 0.85-1.17 kg N ha^{-1}yr^{-1} in the winter, and 0.25 kg N ha^{-1}yr^{-1} in the summer. This average value compares well with the Nanus (et al., 2003) model that estimate that the Tetons receive 0.1-2.5 kg N ha^{-1}yr^{-1}. The MB sites receive more N deposition (1.7 kg N ha^{-1}yr^{-1}) than RDV (0.85 kg N ha^{-1}yr^{-1}) mostly in the form of snow. There was no winter data for the PB sites, and only one snow pack survey for the MB sites because winter access to the sites was too dangerous so gradients could not be thoroughly tested. There were no differences in summer N input which generally was in the order of 0.18-0.32 kg N ha^{-1}yr^{-1} in 2006 and 2007. We encountered some methodological issues with the mixed-bed resin collectors (i.e. the Johnson collectors) relative to NH_4^+-N overestimation. The NO_3^- -N from the mixed-bed resin collectors might be a good candidate for further investigation of ecosystem NO_3^- -N deposition. Therefore, mixed-bed resins may not be suitable to estimate NH_4^+-N deposition when collectors are placed in harsh field conditions for prolonged periods of time.

The hypothesis that the effect of N deposition would be influenced by edaphic conditions was not supported. Our *a priori* wet and dry sites were different based on gravimetric soil moisture content (P = 0.004) and temperature (P = 0.005). During the snow free period, all dry sites had similar moisture content, while the wet sites exhibited greater variability. Based on annual degree days, the RDV location was overall warmer and drier compared to the MB sites (P = 0.02). These differences were found to have no effect on N deposition or N status in both plants and soils. The PRISM data suggested that 2006 had slightly more precipitation than 2007. The idea that N deposition follows precipitation may have relevance for MB receiving more N deposition than RDV. However, this is not reflected during the summer months.

The hypothesis that sites with higher N deposition (i.e., MB) had higher soil N content, more available N in the summer, and higher nitrification potential than sites with lower N deposition (i.e., RDV) was supported. These parameters showed MB to be an N-rich site, and RDV to be an N-poor site. The PB sites were intermediate in value, had high variability, and shared several aspects between both MB and RDV. Thus, in this study, the various soil indicators of N status were able to distinguish N-rich from N-poor soils, but not intermediate conditions.

The hypothesis of more soil N, more available N, higher net mineralization, and nitrification potential in wet vs. dry sites was not supported. There were three cases where the N differed between wet and dry (2007 soil N pool, Fig. 4.4, 2006 summer PRS extractable N, Fig. 4.7, and 2006 mineralization potential, Fig. 4.8), but that was mostly due to high variability and statistical interactions. Overall, there were little, if any differences between wet and dry sites for N status in the soils.

The hypothesis of the plant responses to N status was that high N deposition sites would have more above and belowground biomass, more N in foliage and roots, and would differ in species richness and composition. This hypothesis was only partially supported. The MB sites indeed had more aboveground (but not belowground) biomass, more N in foliage (but not roots), and greater percent cover than the RDV sites. The MB sites exhibited slightly less (although not statistically significant) plant diversity than RDV, which does not support our hypothesis. The hypothesis that wet sites would have more overall biomass than dry sites was generally supported, but were mainly driven by the large differences at the PB location.

In summary, our findings show that alpine ecosystems do respond to small changes in N deposition and N status as shown by both plant and soil indicators. Extreme differences can be established (i.e. N-rich vs. N-poor), but sites with intermediary conditions are difficult to distinguish with the indicators used in this study.

LITERATURE CITED

Aber, J.D., Nadelhoffer K.J., Steudler P., Melillo, J.M. 1989. NitrogeN-saturation in northern forest ecosystems. Bioscience 39, 378-386.

Aber , J.D. 1992. Nitrogen cycling and nitrogeN-saturation in temperate forest ecosystems. Trends in Ecological Evolution 7, 220-224.

Aber, J.D., McDowell, W., Nadelhoffer, K., Magill, A., Bernston, G., Kamakea, M., McNulty, S., Currie, W., Rustad, L. Fernandez, I. 1998. NitrogeN-saturation in temperate forest ecosystems. Bioscience 48(11), 921-934.

Adams, B.A., Holmes, E.L. 1935. Adsorptive properties of synthetic resins. Journal Society of Chemical Industry 54,1-6.

Anderson, J.M. 1977. The organization of soil animal communities. Soil organisms as components of ecosystems: International soils zoology colloquium, Lohm, U. and Persson, T. Stockholm, Sweden Natural Resource Council. Ecological Bulletins 25, 175-181.

Ashton, L.W., Miller, A.E., Bowman, W.D., Suding, K.N. 2008. Nitrogen preferences and plant-soil feedbacks as influenced by neighbors in the alpine tundra. Oecologia 156,625-636.

Asman, W.A.H., Sutton, M.A., Schjorring, J.K. 1998. Ammonia: emission, atmospheric transport and deposition. New Phytologist 139(1), 27-48. Disturbance of the N cycle.

Asner, G.P., Townsend, A.R., Riley, W.J., Matson, P.A., Neff, J.C., Cleveland, C.C. 2001. Physical and biogeochemical controls over terrestrial ecosystem responses to nitrogen deposition. Biogeochemistry 54(1),1-39.

Bardgett, R.D., Van Der Wal, R., Jónsdóttir, I.S., Quirk, H., Dutton, S. 2007 Temporal variability in plant and soil nitrogen in a high-arctic ecosystem. Soil Biology and Biochemistry. 39, 2129-2137.

Baron, J.S., McKnight, D., Denning, A.S. 1991. Sources of dissolved and particulate organic material in Loch Vale Watershed, Rocky Mountain National Park, USA. Biogeochemistry 15(2), 89-110.

Baron, J.S. 1992. Biogeochemical fluxes. J. Baron (ed.) Biogeochemistry of a Subalpine Ecosystem. Springer-Verlag, New York pp.218-231.

Baron, J.S., Ojima, D.S., Holland, E.A., Parton, W.J. 1994. Analysis of nitrogeN-saturation potential in rocky mountain tundra and forest: Implications for aquatic systems. Biogechemistry 27(1), 61-82.

Baron, J.S., Rueth, H.M., Wolfe, A.M., Nydick, K.R., Allstott, E.J., Minear, T., Moraska, B. 2000. Ecosystem responses to nitrogen deposition in the Colorado Front Range. Ecosystems 3 (4), 352-368.

Begon, M., Harper, J.L, Townsend, C.R. 1990. Ecology. Blackwell Scientific, Massachusetts.

Belnap, J., and Eldridge, D.J. 2001. Disturbance and recovery of biological soil crusts. In: J. Belnap and O.L. Lange, Editors, Biological Soil Crusts: Structure, Function, and Management, Springer, Berlin pp. 363–384.

Belnap, J., Caldwell, M.M., Ehleringer, J.R., Sanford, R.J. 2003. Nitrogen deposition and UV stressor impacts in Canyonlands National Park as affected by climatic pulse events. NPS final report. EPA reference: DW14938083-01-0. NPS reference:IA-1341-8-9002.

Benedict, J.B. 1968. Recent glacial history of an alpine area in the Colorado Front Range, U.S.A. II. Dating the glacial deposits. Journal of Glaciology, 7(49),77-87.

Bever, J.D., Westover, K.M., Antonovics, J. 1997. Incorporating the soil community into plant population dynamics: the utility of the feedback approach. The Journal of Ecology 85(5), 561-573.

Bilbrough, C.J, Welker, J.M., Bowman, W.D. 2000. Early spring nitrogen uptake by snow-covered plants: A comparison of arctic and alpine plant function under snowpack. Arctic, Antarctic, and Alpine Research 32(4), 404-411.

Binkley, D, Matson, P. 1983. Ion exchange resin bag method for assessing forest soil nitrogen availability. American Journal of Soil Science Society 47, 1050-1052.

Binkley, D., Vitousek, P.M, 1989. Soil nutrient availability in Plant Physiological Ecology: Field Methods and Instrumentation. Edited by Pearcy, R.W., Ehleringer, J.R., Mooney, H.A., Rundel, P.W. Chapman and Hall, London.

Blair, J.M., Seastedt, T.R., Rice, C.W., Ramundo, R.A., 1998. Terrestrial nutrient cycling in tallgrass prairie. Grassland dynamics. Long-term ecological research in tallgrass prairie. Editors: Knapp, A.K., Briggs, J.M., Hartnett, D.C., Collins, S.L. Oxford University Press, New York, New York, USA. PP 222-243.

Blett, T., Morris, K., Baron, J., Campbell, D., Cordova, K., Ely, D., Latimer, D., Mitchell, B., Shaver, C., Silverstein, M., Vimont, J. 2004. Nitrogen Deposition: issues and effects in Rocky Mountain National Park Technical background document. Denver, Colorado. Air Division of the National Park Service.

Bliss, L.C. 1966. Plant productivity in alpine microenvironments on Mt. Washington, New Hampshire. Ecological Monographs 36, 125-155.

Booth, M.S., Stark, J.M., Rastetter, E. 2005. Controls on nitrogen cycling in terrestrial ecosystems: A synthetic analysis of literature data. Ecological Monographs 75(2), 139-157.

Bowman, W.D. 1992. Inputs and storage of nitrogen in winter snowpack in an alpine ecosystem. Arctic and Alpine Research 24(3), 211-215.

Bowman, W.D., Theodose, T.A., Schardt, J.C. 1993. Constraints of nutrient availability on primary production in two alpine tundra communities. Ecology 74(7), 2085-2097.

Bowman. W.D. 1994. Accumulation and use of nitrogen and phosphorous following fertilization in two alpine tundra communities. Oikos, 70(2), 261-270

Bowman, W.D., Conant, T.R. 1994. Shoot growth dynamics and photosynthetic response to increased nitrogen availability in the alpine willow *Salix glauca.* Oecologia 97, 93-99.

Bowman, W.D., Theodose, T.A., Fisk, M.C. 1995. Physiological and production responses of plant growth forms to increases in limiting resources in alpine tundra: Implications for differential community response to environmental change. Oecologia 101, 217-227.

Bowman, W.D., Steltzer, H. 1998. Positive feedbacks to anthropogenic nitrogen deposition in Rocky Mountain alpine tundra. Ambio 27, 514-517.

Bowman, W.D. 2000. Biotic controls over ecosystem response to environmental change in alpine tundra of the Rocky Mountains. Ambio 29, 396-400.

Bowman, W.D., Bahn, L., Damm, M. 2003. Alpine landscape variation in foliar nitrogen and phosphorus concentrations and the relation to soil nitrogen and phosphorus availability. Arctic, Antarctic, and Alpine Research. 35(2), 114-149.

Bowman, W.D., Gartner, J.R., Holland, K., Wiedermann, M. 2006. Nitrogen critical loads for alpine vegetation and terrestrial ecosystem response: Are we there yet? Ecological Applications 16(3), 1183-1193.

Bouwman, A.F., Lee, D.S., Asman, W.A.H., Dentener, F.J., van der Hoek, K.W., Olivier, J.G.J., 1997. A global high-resolution emission inventory for ammonia. Global Biogeochemical Cycles 11, 561-587.

Brooks, P.D., Williams, M.W., Schmidt, S.K. 1996. Microbial Activity Under Alpine Snowpacks, Niwot Ridge, Colorado. Biogeochemsitry 32, 93-113.

Brooks, P.D., Williams, M.W., Schmidt, S.K. 1998. Inorganic nitrogen and microbial biomass dynamics before and during spring snowmelt. Biogeochemistry 43, 1-15.

Brooks, M.L. 2003. Effects of soil nitrogen on the dominance of alien annual plants in the Mojave Desert. Journal of Applied Ecology 40(2), 344-353.

Brzoska, W. 1969. Stoffproduktion und energiehaushalt der vegetation auf hochalpinem standort unter besonderer Berücksichtigung von *Ranunculus glacialis.* L. PhD thesis, University of Innsbruck.

Buol, S.W., Southard, R.J., Graham, R.C., McDaniel, P.A. 2003. Soil genesis and classification. Iowa state press, 5th edition, Blackwell publishing.

Burns, D.A. 2003. The effects of atmospheric nitrogen deposition in the Rocky Mountains of Colorado and southern Wyoming, USA-A critical review. Environmental Pollution. 127, 257-269.

Carey, C. 1993. Hypothesis concerning the causes of the disappearance of boreal toads from the mountains of Colorado. Conservation Biology. 7, 355-362.

Castro, M. S., Peterjohn, W.T., Melillo, J.M., Steudler, P.A. 1994. Effects of nitrogen fertilization on the fluxes of N_2O, CH_4, and CO_2 from soils in a Florida slash pine lantation. Canadian Journal of Forest Research 24, 9-13.

Chapin, F.S. 1980. The mineral nutrition of wild plants. Annual Review of Ecological Systems. 11, 233-260.

Chapin, F.S., Schulze, E.D., Mooney, H.A. 1990. The ecology and economics of storage in plants. Annual Review of Ecology and Systematics. 21,423-447.

Cobos, C.R. Chambers, C. 2010. Calibrating ECH20 moisture sensors. Decagon Devices Inc.© http://www.decagon.com/assets/Uploads/13393-04 CalibratingECH2OSoilMoistureProbes.pdf

Conley, A.H., Holland, E.A., Seastedt, T.R., Parton, W.J. 2000. Simulation of carbon and nitrogen cycling in an alpine tundra. Arctic, Antarctic, and Alpine Research, 32(2), 147-154.

Curtis, J., Grimes, K. 2004. Wyoming climate atlas. Water Resources Data System Dept 3943. Laramie, Wyoming.

Dail, D.B., Davidson, E.A., Chorover, J. 2001. Rapid abiotic transformation of nitrate in an acid forest soil. Biogeochemistry 54, 131-146.

Dawson, G.A., 1977. Atmospheric ammonia from undisturbed land. Journal of Geophysical Research 82, 3125-3133.

Edwards, A.C., Scalenghe, R., Freppaz, M. 2007. Changes in seasonal snow cover of alpine regions and its effect on soil processes: A review. Quaternary International 162-163,172-173.

Edwards, K.A., McCulloch, J., Kershaw, G.P., Jefferies, R.L., 2006. Soil microbial and nutrient dynamics in a wet arctic sedge meadow in late winter and early spring. Soil Biological Biochemistry. 38, 2843-2851.

Egerton-Warburton, L.M., Allen, E.B. 2000. Shifts in arbuscular mycorrhizal communities along an anthropogenic nitrogen deposition gradient. Ecological Applications 10, 484-496.

Fenn, M.E., Poth, M.E., Aber, J.D., Baron, J.S., Bormann, B.T., Johnson, D.W., Lemly, A.D., McNulty, S.G., Ryan, D.F., Stottlemyer, R.R. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management stratagies. Ecological Applications 8(3), 706-733.

Fenn, M. E., Haeuber, R., Tonnesen, G. S., Baron, J.S., Grossman-Clarke, S., Hope, D., Jaffe, D. A, Copeland, S., Geiser, L., Rueth, H. M., Sickman, J. O. 2003(a). Nitrogen emissions, deposition, and monitoring in the Western United States. Bioscience 53(4), 391-403.

Fenn, M.A., Baron, J.S., Allen, E.B., Rueth, H.M., Nydick, K.R.,Geiser, L., Bowman, W.D., Sickman, J.O., Meixner,T., Johnson, D.W., Neitlich, P. 2003(b). Ecological effects of nitrogen deposition in the western United States. Bioscience 53(4), 404-420.

Fenn, M.A., Poth, M.E., 1998. Indicators of nitrogen status in California forests. USDA Forest Service General Technical Report. PSW-GTR-166.

Fenn, M.A., Poth, M.E. 2004. Atmospehric pollutants and trace gases: Monitoring nitrogen deposition in throughfall using ion exchange resin columns: A field test in the San Bernadino Mountains. Journal of Environmental Quality 33, 2007-2014.

Fenneman, N.M. 1931. Physiography of the Western United States. New York, McGraw Hill, pp534.

Fisk, M.C., Schmidt, S.K. 1995. Nitrogen mineralization and microbial biomass N dynamics in three alpine tundra communities. American Journal of Soil Science Society 59, 1036-1043.

Fisk, M.C., Schmidt, S.K. 1996. Microbial responses to nitrogen additions in alpine tundra soil. Soil Biology and Biochemsitry 28(6), 751-755.

Fisk, M.C., Schmidt, S.K., Seastedt, T.R. 1998. Topographic patterens of above and belowground production and nitrogen cycling in alpine tundra. Ecology 79(7), 2253-2266.

Friedland, A.J., Miller, E.K. 1999. Major-element cycling in a high-elevation Adirondack forest: patterns and changes, 1986-1996. Ecological Applications 9(3), 958-967.

Frey, S.D., Knorr, M., Parrent, J.L., Simpson, R.T. 2004. Chronic nitrogen enrichment affects the structure and function of the soil microbial community in temperate hardwood and pine forests. Forest Ecology and Management. 196, 159-171.

Geiser, L.H. Neitlich, P.N. 2006. Air pollution and climate gradients in western Oregon and Washington indicated by epiphytic macrolichens. Environmental Pollution 145, 203-218.

Goldberg, D.E., Miller, T.E. 1990. Effects of different resource additions of species diversity in an annual plant community. Ecology 71, 213-225.

Good, J.M, Pierce, K.L. 1996. Interpreting the landscape: Recent and ongoing geology of Grand Teton and Yellowstone National Parks. Grand Teton Natural History Association and USGS, Wyoming.

Gough, L., Osenberg, C.W., Gross, K.L., Collins, S.L. 2000. Fertilization effects on species density and primary productivity in herbaceous plant communities. Oikos 89(3), 428-439.

Goulding, K.W.T., Bailey, N.J., Bradbury, N.J., Hargreaves, P., Howe, M., Murphy, D.V., Poulton, P.R., Willson, T.W. 1998. Nitrogen deposition and its contribution to nitrogen cycling and associated soil processes. New Phytologist. 139(1), 49-58.

Grossman, D. H., D. Faber-Langendoen, A. S. Weakley, M. Anderson, P. Bourgeron, R. Crawford, K. Goodin, S. Landaal, K. Metzler, K. D. Patterson, M. Pyne, M. Reid, and L. Sneddon. 1998. International classification of ecological communities: terrestrial vegetation of the United States. Volume I. The National Vegetation Classification System: development, status, and applications. The Nature Conservancy, Arlington, Virginia, USA.

Güsewell, S., Koerselman, W., Verhoeven, J.T.A. 2002. Time-dependent effects of fertilization on plant biomass in floating fens. Journal of Vegetation Science 13(5), 705-718.

Habeck, J.R., 1987. Present-day vegetation in the Northern Rocky Mountains. Annals of the Missouri Botanical Garden. 74(4), 804-840.

Hadley, K. 1984. A Biogeographic interpretation of vascular alpine plant distribution within the Rocky Mountain cordillera: preliminary investigations M.A. Thesis, Department of Geography. University of Wyoming.

Hangs, R.D., Greer, K.J., Sulewski, C.A. 2004. The effect of interspecific competition on conifer seedling growth and nitrogen availability measured using ion-exchange membranes. Canadian Journal of Forest Research 34, 754-761.

Hanselman, T.A., Graetz, D.A., Obreza, T.A., 2004. A comparison of in-situ methods for measuring net N mineralization rates of organic soil amendments. Journal of Environmental Quality. 33, 1098-1105.

Harris, A.G, Tuttle, E., Tuttle, S.D. 1997. Geology of national parks: 5th edition. Kendall/Hunt Publishing, Iowa.

Harte, J. Hoffman, E. 1989. Possible effects of acidic deposition on a Rocky Mountain population of the Tiger salamander *Ambystoma tigrinum.* Conservation Bilogy. 3(2), 149-158.

Haselwandter, K., Hofmann, A., Holzmann, H.P., Read, D.G. 1983. Availability of nitrogen and phosphorous in the nival zone of the Alps. Oecologia 57, 266-269.

Howard, D.M., Howard, P.J.A. 1980. Effect of species, source of litter, type of soil and climate on litter decomposition: microbial decomposition of tree and shrub leaf litter. Oikos 34(3), 115-124.

Heidorn, K.C. 1979. A chronology of important events in the history of air pollution meteorology to 1970. Bulletin of American Meteorological Society. 78, 1589-1597.

Hungate, B.A., Jordan, T.E., Jackson, R.B., Drake, B.G., Henebry, G.M., Wedin, D.A., Tilman, D. 1997. Atmospheric nitrogen deposition. Science News Series 275(5301), 739-741.

Ingersoll, G.P., Mast, M.A., Campbell, D.H., Clow, D.W., Nanus,L., Turk, J.T. 2008. Trends in snowpack chemistry and comparison to National Atmospheric Deposition Program results for the Rocky Mountains, US, 1993–2004. Atmospheric Environment. 42, 6098–6113.

Ingersoll, G.P., Mast, M.A., Campbell, D.H., Clow, D.W., Nanus, L., and Turk, J.T., 2009. Rocky Mountain snowpack physical and chemical data for selected sites, 1993–2008, U.S. Geological Survey Data Series 369.

Isard, S.A. 1986. Factors influencing soil moisture and plant community distribution on Niwot Ridge, Front Range, Colorado, USA. Arctic and Alpine Research 18(1), 83-96.

Ives, J.D. 1980. The development of a Front Range Mountain Research Station. In: Ives, J.D. (ed.), Geoecology of the Colorado Front Range. Buolder: Westview Press, xv-xxv.

Jaeger, C.H., Monson, R.K., 1992. The adaptive significance of nitrogen storage in *Bistorta bistoroides*, an alpine herb. Oecologia 92, 578-585.

Jaeger, C.H., Monson. R.K., Fisk, M.C., Schmidt, S.K. 1999. Seasonal partitioning of nitrogen by plants and soil microorganisms in an alpine ecosystem. Ecology 80(6), 1883-1891.

Jaramillo, V.J., Kauffman, J.B., Renterìa-Rodrìguez, L., Cummungs, D.L., Ellingson, L.J. 2003. Biomass, carbon, and nitrogen pools in Mexican tropical dry forest landscapes. Ecosystems 6(7), 609-629.

Johnson, D.W., Verburg, P.S.J., Arnone, J.A. 2005. Soil extraction, ion exchange resin, and ion exchange membrane measures of soil mineral nitrogen during incubation of a tall grass prairie soil. Soil Science Society of America. 69, 260-265.

Kielland, K. 1995. Landscape patterns of free amino acids in arctic tundra soils. Biogeochemsitry. 31, 85-98.

Kiesecker, J. 1996. pH-mediated predator-prey interactions between *Ambystoma tigrinum* and *Pseudacris triseriata*. Ecological Applications 6, 111-117.

Kittel, T.G.F, Thornton, P.E., Royle, J.A., Chase, T.N 2002. Climates of the Rocky Mountains: historical and future patterns. Chapter 4. Rocky Mountain futures, an ecological perspective, edited by Baron, J.S. Island press Washington, DC.

Kjønaas, O.J, 1999. In situ efficiency of ion exchange resins in studies of nitrogen transformation. American Journal of Soil Science Society 63, 399-409.

Köchy, M., Wilson, S.D. 2001. Nitrogen deposition and forest expansion in the northern Great Plains. Journal of Ecology. 89, 807-817.

Körner, C.H. 1989. The nutritional status of plants from high altitudes. Oecologia. 81, 379-391.

Körner, C.H. 2003. Alpine plant life: functional plant ecology of high mountain ecosystems. Springer Publishing Verlag, Berlin, Heidelberg, New York.

Langan, S.J. 1999. Impact of Nitrogen Deposition on Natural and Semi-natural Ecosystems. Dordrecht; Boston, Kluwer academic.

Langlois, J.L., Johnson, D.W., Mehuys, G.R. 2003. Adsorption and recovery of dissolved organic phosphorus and nitrogen by mixed-bed ion-exchange resin. American Journal of Soil Science Society 67, 889-894.

Lesham, Y. 1996. Nitric oxide in biological systems. Plant Growth Regulation. 18, 155-159.

Ley, R.E., Williams, M.W., Schmidt, S.K. 2004. Microbial population dynamics in an extreme environment: controlling factors in talus soils at 3750 m in the Colorado Rocky Mountains. Biogeochemistry 68(3), 313-335.

Lipson, D.A., Bowman. W.D., Monson, R.K. 1996. Luxury uptake and storage of nitrogen in the rhizomatous alpine herb, *Bistorta bistoroides.* Ecology 77, 1277-1285.

Lipson, D.A., Monson, R.K. 1998. Plant-microbe competition for soil amino acids in the alpine tundra: effects of freeze-thaw and dry-rewet events. Oecologia 113, 406-414.

Lipson, D.A., Schmidt, S.K, Monson, R.K. 1999. Links between microbial population dynamics and nitrogen availability in an alpine ecosystem. Ecology 80(5), 1623-1631.

Lipson, D.A., Schadt, C.W., Schmidt, S.K. 2002. Changes in microbial community structure and function in an alpine dry meadow following spring snow melt. Microbial Ecology 43, 307-314.

Ljungholm, K., Norin, B., Wadso, I. 1979. Microcalorimetric observations of microbial activity in normal and acidified soils. Oikos 33, 24-30.

Love, D.J., Reed, J.C., Pierce, K.L. 1989. Creation of the Teton Landscape. Chapter Title: Precambian Rocks-The Core of the Tetons. (www.nps.gov/history/history/online_books/grte/grte_geology/sec5.htm)

Magill, A.H., Aber, J.D., 1998. Long-term effects of experimental nitrogen additions on foliar litter decay and humus formation in forest ecosystems. Plant and Soil 203, 301-311.

Magill, A.H., Aber, J.D., Berntson, G.M, McDowell, W.H., Nadelhoffer, K.J., Melillo, J.M., Steudler, P. 2000. Long-term nitrogen additions and nitrogeN-saturation in two temperate forests. Ecosystems 3(3), 238-253.

Magill, A.H., Aber, J.D., Currie, W.S., Nadelhoffer, K.J., Martin, M.E., McDowell, W.H., Melillo, J.M., Steudler, P. 2004. Ecosystem response to 15 years of chronic nitrogen additions at the Harvard Forest LTER, Massachusetts, USA. Forest Ecology and Management 196, 7-28.

Mahaney, W.C. 1975. Soils of post-audubon age, Teton Glacier area, Wyoming. Arctic and Alpine Research. 7(2), 141-153.

Makarov, M.I., Leoshkina, N.A., Ermak, A.A., Malysheva, T.I. 2010. Seasonal dynamics of the mineral nitrogen forms in mountain-meadow alpine soils. Eurasian Soil Science. 43(8), 905-913.

Mamo, M., Ginting, D., Renken, R., Eghball, B. 2004. Stability of ion exchange resin under freeze-thaw or dry-wet environment. Soil Science Society of America Journal. 68, 677-681.

Mancinelli, R.L. 1984. Population dynamics of alpine tundra soils bacteria, Niwot Ridge, Colorado Front Range, USA. Arctic and Alpine Research 16(2), 185-192.

Mancinelli, R.L. 1986. Alpine tundra soil bacterial responses to increased soil loading rates of acid precipitation, nitrate, and sulfate, Front Range, Colorado, USA. Arctic and Alpine Research 18(3), 269-275.

Mancinelli, R.L.; Keigley, R.B. 1983. Effects of increases in soil loading rates of nitrate, nitrate/sulfate and acid rain on the alpine tundra soil bacterial community. Abstracts of the Annual Meeting of the American Society of Microbiology, New Orleans, Louisiana, 232. Abstract.

May, D.E., Webber, P.J., May, T.A. 1982. Success of transplanted alpine tundra plants on Niwot Ridge, Colorado. J. of Applied Ecology 19, 965-976.

McCune, B., Rogers, P., Ruchty, A. Ryan, B. 1998. Lichen communities for forest health monitoring in Colorado, USA. A report to the USDA Forest Service. 32 pages. Department of Botany and Plant Pathology, Oregon State University, Corvallis, OR. USDA Forest Service, Intermountain Research Station. Department of Botany, Arizona State University, Tempe, AZ (PDF:http://fia.fs.fed.us/lichen/pubs/).

McKnight, D.M., Smith, R.L., Bradbury, J.P., Baron, J.S., Spaulding, S. 1990. Phytoplankton dynamics in three Rocky Mountain lakes, Colorado, U.S.A. Arctic and Alpine Research. 22, 264-274.

Mears, B., 1981. Periglacial wedges and the late Pleistocene environment of Wyoming's intermontane basins. Quaternary Research 15, 171-198.

Meixner, T., Allen, E.B., Tonnessen, K., Fenn, M.A., Poth, M. 2002. Atmospheric nitrogen deposition: implications for managers of western U.S. parks. Park Science 21(2), 30-33.

Meurk, C.D. 1978. Alpine phytomass and primary productivity in Central Otago, New Zealand. New Zealand Journal of Ecology 1, 27-50.

Miller, A.E., Bowman, W.D.. 2003. Alpine plants show species-level differences in the uptake of organic and inorganic nitrogen. Plant and Soil. 250, 283-292.

Miller, A.E., Bowman, W.D., Suding, K.S. 2007a. Plant uptake of inorganic and organic nitrogen: neighbor identity matters. Ecology. 88(7), 1832-1840.

Miller, A.E., Schimel, J.P., Sickman, J.O., Meixner, T., Doyle, A.P., Melack, J.M. 2007b. Mineralization responses at near-zero temperatures in three alpine soils. Biogeochemistry 84, 233-245.

Mishustin, E.N. 1975. Microbial associations of soils types. Microbial Ecology 2, 97-118.

Mohn, J., Schürmann, A., Hagedorn, F., Schleppi, P., Bachofen, R. 2000. Increased rates if denitrification in nitrogen-treated forest soils. Forest Ecology and Management. 137, 113-119.

Mullen, R.B., Schmidt, S.K, Jaeger, C.H. 1998. Nitrogen uptake during snowmelt by the snow buttercup, *Ranunculus adoneus*. Arctic and Alpine Research 30(2), 121-125.

Nagy, M.L., Johansen, J.R., St. Clair, L.L., Webb, B.L. 2005. Recovery patterns of microbiotic soil crusts 70 years after arsenic contamination. Journal of Arid Environments. 63(1), 304-323.

Nagy, L. and Grabherr, G. 2009. The Biology of alpine habitats. Oxford University Press, New York.

Nanus, L., Campbell, D.H., Ingersoll, G.P., Clow, D.W., Mast, M.A. 2003. Atmospheric deposition maps for the rocky mountains. Atmospheric Environment 37, 4881-4892.

Näsholm, T. 1997. Qualitative and quantitative changes in plant nitrogen acquisition induced by anthropogenic nitrogen deposition. New Phytologist. 139(1), 87-90. Disturbance of the N cycle.

National Atmospheric Deposition Program (NRSP-3). 2005. NADP Program Office, Illinois State Water Survey, 2204 Griffith Dr., Champaign, IL 61820.

Nemergut, D.R., Townsend, A.R., Sattin, S.R., Freeman, K.R., Flerer, N., Neff, J.C., Bowman, W.D., Schadt, C.W., Weintraub, M.N., Schmidt, S.T. 2008. The effects of chronic nitrogen fertilization on alpine tundra soil microbial communities: implications for carbon and nitrogen cycling. Environmental Microbiology 10(11), 3093-3105.

Norby, R.J., Weerasuriya, Y., Hanson, P. J. 1989. Induction of nitrate reductase activity in red spruce needles by NO_2 and HNO_3 vapor. Canadian Journal Forest Research 19, 889-896.

Ohlson, M., Nordin, A., Nasholm, T. 1995. Accumulation of amino acids in forest plants in relation to ecological amplitude and nitrogen supply. Functional Ecology 9 (4), 596-605.

Olivier J.G.J., Bouwman A.F., van der Maas C.W.M., Berdowski J.J.M., Veldt C., Bloos J.P.J., Visschedijk A.J.H., Zandveld P.Y.J., Havelag J.L. 1995. Description of EDGAR Version 2.0: A set of global emission inventories of greenhouse gases and ozone-depleting substances for all anthropogenic and most natural sources on a per country basis on 1 x grid. In: RIWM, 771060 002

Peterjohn, T.W., Adams, M.B., Gillian, F.S. 1996. Symptoms of nitrogeN-saturation in two Appalachian hardwood forest ecosystems. Biogeochemsitry 35(3), 507-522.

Parameter-elevation regression independent slopes model. 2006 and 2007. Oregon climate center, Oregon State University, Oregon. http://prismmap.nacse.org/nn/index.phtml?vartype=ppt&month=01&year0=2006&year1=2006

Philippi, T.E., Dixon, P.M., Taylor, B.E. 1998. Detecting trends in species composition. Ecological Applications 8(2), 300-308.

Reddy, M.M., Caine, N. 1990. Dissolved solutes budget of a small alpine basin, Colorado. In Poppoff, I.G., Goldman, C.R., Loeb, S.L., Leopold, L.B. (eds.), International Mountain Watershed Symposium. Tahoe Conservation District, S. Lake Tahoe, California, 370-385.

Rehder, H. 1976. Nutrient turnover studies in alpine ecosystems. I. Phytomass and nutrient relations in four mat communities of the northern calcareous Alps. Oecologia 22(10), 411-423.

Reno, K. 2006. Using resin traps for assessment of N leaching in agricultural production systems. Journal of Undergraduate Research. 8, 1.

Riggan, P. J., Lockwood, R. N., Lopez, E. N. 1985. Deposition and processing of airborne nitrogen pollutants in Mediterranean-type ecosystems of southern California. Environmental Science and Technology 19, 781-789.

Rikhari, H.C., Negi, G.C.S., Pant, G.B., Rana, B.S., Singh, S.P. 1992. Phytomass and primary productivity in several communities of a central Himalayan alpine meadow, India. Arctic and Alpine Research 24, 334-351.

Rueth, H.M., Baron, J.S., Allstott, E.J. 2003. Responses of engleman spruce forests to nitrogen fertilization in the Colorado Rocky Mountains. Ecological Applications 13(3), 664-673.

Rusch, D., Sievering, H. 1995. Variation in ambient air nitrogen concentration and total annual atmospheric deposition at Niwot Ridge, Colorado. Pp. 23-32 In Tonnessen, K. A., M. W. Williams, and M. Tranter (eds.). Biogeochemistry of Seasonally Snow Covered Basins. International Association of Hydrological Sciences, Wallingford, UK, IAHS-AIHS Publication no. 228. pp 465

Rusek, J. 1993. Air-pollution mediated changes in alpine ecosystems and ecotones. Ecological Applications 3(3), 409-416.

Sala, O.E., Jackson, R.B., Mooney, H.A., Howarth, R.W. 2000. Methods in ecosystem science. Springer-Verlag, New York Inc. Chapter 14, Nutrient transformations by John M. Stark.

Schmidt, S.K., Lipson, D.A., Ley, R.E., Fisk, M.C., West, A.E. 2004. Impacts of chronic nitrogen additions vary seasonally and by microbial functional group in tundra soils. Biogeochemistry 69(1), 1-17.

Scott, M.G., Hutchinson, T.C., Feth M.J. 1989. A comparison of the effects on Canadian boreal forest lichens of nitric and sulphuric acids as sources of rain acidity. New Phytologist 11(4), 663-671

Seastedt, T.R., Vaccaro, L. 2001. Plant species richness, productivity, and nitrogen and phosphorus limitations across a snowpack gradient in alpine tundra, Colorado, U.S.A. Arctic, Antarctic, and Alpine Research. 33(1), 100-106.

Simkin, S.M., Lewis, D.N., Weathers, K.C., Lovett, G.M., Shwarz, K. 2004. Determination of sulfate, nitrate, and chloride in throughfall using ion-exchange resin. Water, Air, and Soil Pollution, 153, 343-354.

Sitaula, B. K., Bakken, L. R., Abrahamsen, G. 1995. N-fertilization and soil acidification effects on N_2O and CO_2 emission from temperate pine forest soil. Soil Biology and Biochemistry 27, 1401-1408.

Skogley, E.O., Dobermann, A. 1996. Synthetic ion-exchange resins: soils and environmental studies. Journal of Environmental Quality 25(1), 13-24.

Smith, R.B., Siegel, L.J. 2000. Windows into the earth: The geologic story of Yellowstone and Grand Teton National Parks. Oxford University Press, New York.

Soil Survey of Teton County, Wyoming, Grand Teton National Park Area, 1975. U.S. Department of Agriculture, Soil Conservation Service, U.S. Department of the Interior, National Parks Service, and Wyoming Agricultural Experiment Station.

Stams, A.J.M., Schipholt, I.J. Lutke. 1990. Nitrate accumulation in leaves of vegetation of a forested ecosystem receiving high amounts of atmospheric ammonium sulfate. Plant and Soil 125, 143-145.

Stark, J.M, Firestone, M.K. 1995. Mechanisms for soil moisture effects on activity of nitrifying bacteria. Applied and Environmental Microbiology. 61, 218-221.

Stark, J.M, Hart, S.C. 1997. High rates of nitrification and nitrate turnover in undisturbed coniferous forests. Nature. 385, 61-64.

Steltzer, H., Bowman, W.D. 1998. Differential influence of plant species on soil nitrogen transformations within moist meadow alpine tundra. Ecosystems 1, 464-474.

Stoddard, J.L. 1994. Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. in: Baker LA (ed) Environmental Chemistry of Lakes and Reservoirs . pp. 223-284, American Chemical Society, Washington DC, USA.

Suding, K.N., Miller, A.E., Bechtold, H., Bowman, W.D. 2006. The consequences of species loss on ecosystem nitrogen cycling depends on community compensation. Oecologia. 149, 141-149.

Susfalk, R.B., Johnson, D.W. 2002. Ion exchange resin based soil solution lysimeters and snowmelt solution collectors. Community Soil Science and Plant Analysis. 33(7&8), 1261-1275.

Theodose, T.A., Jaeger, C.H., Bowman, W.D., Schardt, J.C., 1996. Uptake and allocation of 15 N in alpine plants: implications of the importance of competitive ability in predicting community structure in a stressful environment. Oikos 75(1), 59-66.

Theodose, T.A., Bowman, W.A. 1997. Nutrient availability, plant abundance, and species diversity in two alpine tundra communities. Ecology 78(6), 1861-1872.

Tilman, D. 1982. Resource competition and community structure. Princeton University Press, New Jersey.

USDA Staff, 1999. Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys. United States Department of Agriculture, Agriculture Handbook Natural Resources Conservation Service Number 436 second edition.

Van Der Eerden, L. 1998. Nitrogen on microbial and global scales. New Phytologist 139(1), 201-204. Disturbance of the N cycle.

Vitousek, P.M, Aber, J.A., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H., Tilman, G.D. 1997. Human alterations of the global nitrogen cycle: causes and consequences. Issue in Ecology 1, 1-15.

Walker, M.D., Webber, P.J., Arnold, E.H., Ebert-May, D. 1994. Effects of interannual climate variation on aboveground phytomass in alpline vegetation. Ecology 75, 393-408.

Weiss, S.B. 1999. Cars, cows, and checkerspot butterflies: Nitrogen deposition and management of nutrient-poor grasslands for a threatened species. Conservation Biology 13, 1476-1486.

Welker, J.M., Fahnestock, J.T., Povirk, K.L., Bilbrough, C.J., Piper, R.E. 2004. Alpine grassland CO2 ecxhange and nitrogen cycling: grazing history effects, Medicine Bow Range, Wyoming, U.S.A. Arctic, Antarctic, and Alpine Research. 36(1), 11-20.

Williams, M.W., Brooks, D.T., 1997. Organic and inorganic nitrogen pools in talus fields and subtalus water, Green Lakes Valley, Colorado Front Range. Hydrological processes 11(13), 1747-1760.

Williams, M.W., Tonnessen, K.A., 2000. Critical loads for inorganic nitrogen deposition in the Colorado Front Range, USA. Ecological Applications 10(6), 1648-1665.

Williams, M.W., Hood, E., Caine, N. 2001. Role of organic nitrogen in the nitrogen cycle of a high-elevation catchment, Colorado Front Range. Water Resources Reserarch. 37(10), 2569-2581.

Williams, M.W. Losleben, M.V., Hamann, H.B. 2003. Alpine areas in the Colorado Front Range as monitors of climate change and ecosystem response. The Geographical Review 92(2), 180-191.

Witkamp, M., 1966. Decomposition of leaf litter in relation to environment microflora and microbial respiration. Ecology 47, 194-200.

Wolfe, A.P., Baron, J.S., Cornett, R.J., 2001. Anthropogenic nitrogen deposition induces rapid changes in alpine lakes of the Colorado Front Range (USA). Journal of Paleolimnology. 25, 1-7.

Young, J.F., Lewis, D.J., Fowkes, C.J., Glenn, W.R., 1982. Soil survey of Teton county, Wyoming, Grand Teton National Park area. United States Department of Agriculture Soil Conservation Service, United States Department of the Interior, National Park Service, Wyoming Agriculture Experiment Station.

APPENDICES

Appendix 1 Summary of plot pair location, edaphic condition, replicate, UTMs, and specific elevation for each plot.

	edaphic condition	plot site	plot designation	UTM X	UTM Y	elevation (Meters)
MB	WET	1	A	512656	4866537	2810
MB	WET	2	A	512590	4866486	2921
MB	WET	3	A	512928	4866659	2872
MB	WET	1	B	512656	4866537	2810
MB	WET	2	B	512590	4866486	2921
MB	WET	3	B	512928	4866659	2872
ML	WET	1	A	515579	4849331	2975
ML	WET	2	A	515574	4849430	2980
ML	WET	3	A	515721	4849485	2978
ML	WET	1	B	515579	4849331	2975
ML	WET	2	B	515574	4849430	2980
ML	WET	3	B	515721	4849485	2978
RDV	WET	1	A	509432	4827157	2839
RDV	WET	2	A	509464	4827283	2862
RDV	WET	3	A	509492	4827467	2851
RDV	WET	1	B	509432	4827157	2839
RDV	WET	2	B	509464	4827283	2862
RDV	WET	3	B	509492	4827467	2851
MB	DRY	1	A	512622	4866552	2912
MB	DRY	2	A	512660	4866476	2927
MB	DRY	3	A	512940	4866626	2881
MB	DRY	1	B	512622	4866552	2912
MB	DRY	2	B	512660	4866476	2927
MB	DRY	3	B	512940	4866626	2881
ML	DRY	1	A	515142	4849303	3081
ML	DRY	2	A	515214	4849371	3070
ML	DRY	3	A	515159	4849385	3067
ML	DRY	1	B	515142	4849303	3081
ML	DRY	2	B	515214	4849371	3070
ML	DRY	3	B	515159	4849385	3067
RDV	DRY	1	A	509321	4827587	2925
RDV	DRY	2	A	509332	4827647	2915
RDV	DRY	3	A	509358	4827748	2926
RDV	DRY	1	B	509321	4827587	2925
RDV	DRY	2	B	509332	4827647	2915
RDV	DRY	3	B	509358	4827748	2926

Appendix 2. percent cover estimate scale based on Grossman *et al.*, (1998) national vegetation classification system. Increments of 5% were used instead of 15%.

Index number	**Percent Cover**	**Index number**	**Percent Cover**
1	0-5%	7	30-35%
2	5-10%	8	35-40%
3	10-15%	9	40-50%
4	15-20%	10	50-55%
5	20-25%	11	55-60%
6	25-30%	12	60-65%

Appendix 3. Floral Survey for Each Plot

MBW1A

Date: 8/4/2006 and 7/11/2007	% Slope: 0
UTM X: 512656	Aspect: 0
UTMY: 4866537	
Elevation (Meters): 2810	

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	25-30	0	40	60	grass, sedge, forbs	6
28	Ranunculus eschscholtzii	10-15	0	40	60	grass, sedge, forbs	6
46	Hieracium fendleri	0-5	0	40	60	grass, sedge, forbs	6
53	Symphyotrichum foliaceus	10-15	0	40	60	grass, sedge, forbs	6
55	Juncus drummondii	15-20	0	40	60	grass, sedge, forbs	6
57	Poa cusiskii epilis	5-10	0	40	60	grass, sedge, forbs	6

MBW2A

Date: 8/4/2006 and 7/11/2007 %Slope: 2.5
UTM X: 512590 Aspect: N/A
UTMY: 4866486
Elevation (Meters): 2921

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	0-5	0	70	30	grass, sedge, forbs	10
7	Arnica mollis	0-5	0	70	30	grass, sedge, forbs	10
8	Erigeron speciosus	0-5	0	70	30	grass, sedge, forbs	10
20	Castilleja miniata	0-5	0	70	30	grass, sedge, forbs	10
23	Sibaldia procumbens	0-5	0	70	30	grass, sedge, forbs	10
30	Senecio serratta	5-10	0	70	30	grass, sedge, forbs	10
41	Erigeron formosissimus	5-10					10
55	Juncus drummondii	5-10	0	70	30	grass, sedge, forbs	10
62	Senecio hydrophilus	0-5	0	70	30	grass, sedge, forbs	10
71	Ligusticum filicinum	5-10	0	70	30	grass, sedge, forbs	10

MBW3A

Date: 8/4/2006 and 7/11/2007 %Slope: 1
UTM X: 512928 Aspect: slightly west
UTMY: 4866659
Elevation (Meters): 2872

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
4	Elymus trachycaulus	0-5	0	70	30	forb	8
8	Erigeron speciosus	5-10	0	70	30	forb	8
11	Mertensia ciliata	0-5	0	70	30	forb	8
19	Phleum alpinum	0-5	0	70	30	forb	8
30	Senecio serratta	5-10					8
39	Arnica longifolia	25-30	0	70	30	forb	8
52	Boechera microphylla	0-5	0	70	30	forb	8
71	Ligusticum filicinum	0-5	0	70	30	forb	8

MLW1A

Date: 8/14/2006 and 7/20/2007 %Slope: 5
UTM X: 515579 Aspect: East
UTMY: 4849331
Elevation (Meters): 2839

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	50-55	10	5	85	grass, sedge, forb	13
12	Salix arctica petraea	5-10	10	5	85	grass, sedge, forb	13
13	Phyllodoce empetriformis	10-15	10	5	85	grass, sedge, forb	13
22	Antennaria umbrinella	0-5	10	5	85	grass, sedge, forb	13
38	Carex paysonis	5-10	10	5	85	grass, sedge, forb	13
41	Erigeron formosissimus	0-5	10	5	85	grass, sedge, forb	13
42	Salix wolfii	15-20	10	5	85	grass, sedge, forb	13
45	Deschampsia cespitosa	5-10	10	5	85	grass, sedge, forb	13
47	Carex breweri paddoensis	35-40	10	5	85	grass, sedge, forb	13
57	Poa cusiskii epilis	65-70	10	5	85	grass, sedge, forb	13
57	Poa cusiskii epilis	15-20	10	5	85	grass, sedge, forb	13
65	Boechera lyallii	5-10	10	5	85	grass, sedge, forb	13
78	UK	0-5	10	5	85	grass, sedge, forb	13

MLW2A

Date: 8/14/2006 and 7/20/2007 %Slope: 5
UTM X: 515574 Aspect: East
UTMY: 4849430
Elevation (Meters): 2862

Species #	Species/genus Name	Species % cover	1m2 % rock	1m2 % bare ground	1m2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	75-80	0	0	100	grass, sedge, forb, short shrub	6
38	Carex paysonis	25-30	0	0	100	grass, sedge, forb, short shrub	6
45	Deschampsia cespitosa	10-15	0	0	100	grass, sedge, forb, short shrub	6
57	Poa cusiskii epilis	85-90	0	0	100	grass, sedge, forb, short shrub	6
65	Boechera lyallii	0-5	0	0	100	grass, sedge, forb, short shrub	6
70	Potentilla diversifolia perdissecta	0-5	0	0	100	grass, sedge, forb, short shrub	6

MLW3A

Date: 8/14/2006 and 7/20/2007 %Slope: 3
UTM X: 515721 Aspect: West
UTMY: 4849485
Elevation (Meters): 2851

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	85-90	0	10	90	grass, sedge, forb, short shrub	13
6	Luzula parviflora	5-10	0	10	90	grass, sedge, forb, short shrub	13
8	Erigeron speciosus	0-5	0	10	90	grass, sedge, forb, short shrub	13
13	Phyllodoce empetriformis	0-5	0	10	90	grass, sedge, forb, short shrub	13
19	Phleum alpinum	15-20	0	10	90	grass, sedge, forb, short shrub	13
22	Antennaria umbrinella	5-10	0	10	90	grass, sedge, forb, short shrub	13
23	Sibaldia procumbens	0-5	0	10	90	grass, sedge, forb, short shrub	13
38	Carex paysonis	25-30	0	10	90	grass, sedge, forb, short shrub	13
40	Epliobium glaberrinum fastigiatum	0-5	0	10	90	grass, sedge, forb, short shrub	13
41	Erigeron formosissimus	25-30	0	10	90	grass, sedge, forb, short shrub	13
44	Carex leporinella	0-5	0	10	90	grass, sedge, forb, short shrub	13
45	Deschampsia cespitosa	5-10	0	10	90	grass, sedge, forb, short shrub	13
57	Poa cusiskii epilis	80-85	0	10	90	grass, sedge, forb, short shrub	13

RDVW1A

Date: 7/24/2006 and 7/14/2007 %Slope: 5
UTM X: 509432 Aspect: East
UTMY: 4827157
Elevation (Meters): 2839

Species #	**Species/genus Name**	**Species % cover**	**$1m^2$ % rock**	**$1m^2$ % bare ground**	**$1m^2$ % veg cover**	**community type**	**Number of Species Per Plot**
2	Carex haydeniana	0-5	0	30	70	grass, sedge, forb	20
3	Poa leptocoma	0-5	0	30	70	grass, sedge, forb	20
9	Erigeron rydbergii cronq.	0-5	0	30	70	grass, sedge, forb	20
17	Anemone multifida tetonensis	0-5	0	30	70	grass, sedge, forb	20
19	Phleum alpinum	0-5	0	30	70	grass, sedge, forb	20
21	Achilliea millefolium lanulosa	0-5	0	30	70	grass, sedge, forb	20
24	Trisetum spicatum	0-5	0	30	70	grass, sedge, forb	20
27	Castilleja sulphurea	0-5	0	30	70	grass, sedge, forb	20
28	Ranunculus eschscholtzii	0-5	0	30	70	grass, sedge, forb	20
29	Anemone parviflora	0-5	0	30	70	grass, sedge, forb	20
31	Senecio dimorphophyllus	5-10					20
32	Epilobium brachyocarpum	0-5	0	30	70	grass, sedge, forb	20
37	Poa alpina	0-5					20
46	Hieracium fendleri	0-5	0	30	70	grass, sedge, forb	20
47	Carex breweri paddoensis	15-20	0	30	70	grass, sedge, forb	20
52	Boechera microphylla	0-5	0	30	70	grass, sedge, forb	20
53	Symphyotrichum foliaceus	0-5	0	30	70	grass, sedge, forb	20
62	Senecio hydrophilus	15-20	0	30	70	grass, sedge, forb	20
64	Erigeron simplex	0-5	0	30	70	grass, sedge, forb	20
81	UK	10-15	0	30	70	grass, sedge, forb	20

RDVW2A

Date: 7/24/2006 and 7/14/2007 %Slope: 5
UTM X: 509464 Aspect: East
UTMY: 4827283
Elevation (Meters): 2862

Species #	**Species/genus Name**	**Species % cover**	**1m^2 % rock**	**1m^2 % bare ground**	**1m^2 % veg cover**	**community type**	**Number of Species Per Plot**
3	Poa leptocoma	0-5	10	70	20	grass, sedge, forb, short shrub	19
9	Erigeron rydbergii cronq.	0-5	10	70	20	grass, sedge, forb, short shrub	19
17	Anemone multifida tetonensis	0-5	10	70	20	grass, sedge, forb, short shrub	19
18	Antennaria microphylla	5-10	10	70	20	grass, sedge, forb, short shrub	19
19	Phleum alpinum	0-5	10	70	20	grass, sedge, forb, short shrub	19
21	Achilliea millefolium lanulosa	0-5	10	70	20	grass, sedge, forb, short shrub	19
24	Trisetum spicatum	5-10	10	70	20	grass, sedge, forb, short shrub	19
25	Sedum lanceolatum	0-5	10	70	20	grass, sedge, forb, short shrub	19
26	Sedum debile	0-5	10	70	20	grass, sedge, forb, short shrub	19
27	Castilleja sulphurea	5-10	10	70	20	grass, sedge, forb, short shrub	19
29	Anemone parviflora	0-5	10	70	20	grass, sedge, forb, short shrub	19
37	Poa alpina	0-5	10	70	20	grass, sedge, forb, short shrub	19
47	Carex breweri paddoensis	15-20	10	70	20	grass, sedge, forb, short shrub	19
48	Linanthus Pungens	15-20	10	70	20	grass, sedge, forb, short shrub	19
50	Salix drummondii	5-10	10	70	20	grass, sedge, forb, short shrub	19
53	Symphyotrichum foliaceus	10-15	10	70	20	grass, sedge, forb, short shrub	19
64	Erigeron simplex	0-5	10	70	20	grass, sedge, forb, short shrub	19
65	Boechera lyallii	0-5	10	70	20	grass, sedge, forb, short shrub	19
75	UK	0-5	10	70	20	grass, sedge, forb, short shrub	19

RDVW3A

Date: 7/24/2006 and 7/14/2007
UTM X: 509492
UTMY: 4827467
Elevation (Meters): 2851

%Slope: 3
Aspect: West

NOTES: 1m^2 Veg cover is 65% moss

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
17	Anemone multifida tetonensis	0-5	15	5	80	grass, sedge, forb, short shrub	12
18	Antennaria microphylla	10-15	15	5	80	grass, sedge, forb, short shrub	12
24	Trisetum spicatum	5-10	15	5	80	grass, sedge, forb, short shrub	12
25	Sedum lanceolatum	0-5	15	5	80	grass, sedge, forb, short shrub	12
26	Sedum debile	0-5	15	5	80	grass, sedge, forb, short shrub	12
29	Anemone parviflora	0-5	15	5	80	grass, sedge, forb, short shrub	12
31	Senecio dimorphophyllus	5-10					12
32	Epilobium brachyocarpum	0-5	15	5	80	grass, sedge, forb, short shrub	12
37	Poa alpina	0-5	15	5	80	grass, sedge, forb, short shrub	12
50	Salix drummondii	15-20	15	5	80	grass, sedge, forb, short shrub	12
54	Carex elynoides	0-5					12
61	Hedysarum occidentale	0-5					12

MBW1B

Date: 8/4/2006 and 7/11/2007
UTM X: 512656
UTMY: 4866537
Elevation (Meters): 2810

%Slope: 0
Aspect: N/A

NOTES: Signs of Pocket gopher activity in plot

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	45-50	0	60	40	grass, sedge, forbs	7
2	Carex haydeniana	5-10	0	60	40	grass, sedge, forbs	7
28	Ranunculus eschscholtzii	5-10	0	60	40	grass, sedge, forbs	7
53	Symphyotrichum foliaceus	10-15	0	60	40	grass, sedge, forbs	7
55	Juncus drummondii	5-10	0	60	40	grass, sedge, forbs	7
57	Poa cusiskii epilis	0-5	0	60	40	grass, sedge, forbs	7
71	Ligusticum filicinum	0-5	0	60	40	grass, sedge, forbs	7

MBW2B

Date: 8/4/2006 and 7/11/2007 %Slope: 2.5
UTM X: 512590 Aspect: N/A
UTMY: 4866486
Elevation (Meters): 2921

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	0-5	0	20	80	grass, sedge, forbs	9
8	Erigeron speciosus	0-5	0	20	80	grass, sedge, forbs	9
19	Phleum alpinum	0-5	0	20	80	grass, sedge, forbs	9
30	Senecio serratta	10-15	0	20	80	grass, sedge, forbs	9
55	Juncus drummondii	35-40	0	20	80	grass, sedge, forbs	9
57	Poa cusiskii epilis	0-5	0	20	80	grass, sedge, forbs	9
62	Senecio hydrophilus	5-10	0	20	80	grass, sedge, forbs	9
71	Ligusticum filicinum	10-15	0	20	80	grass, sedge, forbs	9
77	UK	5-10	0	20	80	grass, sedge, forbs	9

MBW3B

Date: 8/4/2006 and 7/11/2007 %Slope: 1
UTM X: 512928 Aspect: West
UTMY: 4866659
Elevation (Meters): 2872

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
8	Erigeron speciosus	0-5	0	20	80	forb	7
19	Phleum alpinum	15-20	0	20	80	forb	7
30	Senecio serratta	5-10	0	20	80	forb	7
32	Epilobium brachycarpum	10-15	0	20	80	forb	7
39	Arnica longifolia	30-35	0	20	80	forb	7
60	Rumex paucifolius	5-10	0	20	80	forb	7
71	Ligusticum filicinum	0-5	0	20	80	forb	7

MLW1B

Date: 8/14/2006 and 7/20/2007
UTM X: 515579
UTMY: 4849331
Elevation (Meters): 2839

%Slope: 1
Aspect: East

NOTES: $1m^2$ Bare ground is cryptobiotic crust

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	25-30	10	20	70	grass, sedge, forb	14
12	Salix arctica petraea	5-10	10	20	70	grass, sedge, forb	14
13	Phyllodoce empetriformis	5-10	10	20	70	grass, sedge, forb	14
22	Antennaria umbrinella	15-20	10	20	70	grass, sedge, forb	14
23	Sibaldia procumbens	5-10	10	20	70	grass, sedge, forb	14
38	Carex paysonis	35-40	10	20	70	grass, sedge, forb	14
40	Epliobium glaberrinum fastigiatum	0-5	10	20	70	grass, sedge, forb	14
41	Erigeron formosissimus	0-5	10	20	70	grass, sedge, forb	14
47	Carex breweri paddoensis	25-30	10	20	70	grass, sedge, forb	14
57	Poa cusiskii epilis	55-60	10	20	70	grass, sedge, forb	14
65	Boechera lyallii	0-5	10	20	70	grass, sedge, forb	14
78	UK	5-10	10	20	70	grass, sedge, forb	14
80	UK	0-5	10	20	70	grass, sedge, forb	14
83	moss 2	0-5	10	20	70	grass, sedge, forb	14

MLW2B

Date: 8/14/2006 and 7/20/2007	%Slope: 5
UTM X: 515574	Aspect: East
UTMY: 4849430	
Elevation (Meters): 2862	

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	70-75	0	5	95	grass, sedge, forb, short shrub	8
22	Antennaria umbrinella	0-5	0	5	95	grass, sedge, forb, short shrub	8
23	Sibaldia procumbens	0-5	0	5	95	grass, sedge, forb, short shrub	8
38	Carex paysonis	0-5	0	5	95	grass, sedge, forb, short shrub	8
45	Deschampsia cespitosa	5-10	0	5	95	grass, sedge, forb, short shrub	8
47	Carex breweri paddoensis	0-5	0	5	95	grass, sedge, forb, short shrub	8
57	Poa cusiskii epilis	90-95	0	5	95	grass, sedge, forb, short shrub	8
84	moss 3	0-5	0	5	95	grass, sedge, forb, short shrub	8

MLW3B

Date: 8/14/2006 and 7/20/2007 %Slope: 3
UTM X: 515721 Aspect: west
UTMY: 4849485
Elevation (Meters): 2851

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	40-45	0	10	90	grass, sedge, forb, short shrub	8
8	Erigeron speciosus	10-15	0	10	90	grass, sedge, forb, short shrub	8
19	Phleum alpinum	5-10	0	10	90	grass, sedge, forb, short shrub	8
22	Antennaria umbrinella	0-5	0	10	90	grass, sedge, forb, short shrub	8
38	Carex paysonis	0-5	0	10	90	grass, sedge, forb, short shrub	8
41	Erigeron formosissimus	25-30	0	10	90	grass, sedge, forb, short shrub	8
45	Deschampsia cespitosa	0-5	0	10	90	grass, sedge, forb, short shrub	8
57	Poa cusiskii epilis	85-90	0	10	90	grass, sedge, forb, short shrub	8

RDVW1B

Date: 7/24/2006 and 7/14/2007
UTM X: 509432
UTMY: 4827157
Elevation (Meters): 2839

%Slope: 5
Aspect: East

NOTES: The 1m^2 veg cover is 65% moss

Species #	**Species/genus Name**	**Species % cover**	**1m^2 % rock**	**1m^2 % bare ground**	**1m^2 % veg cover**	**community type**	**Number of Species Per Plot**
2	Carex haydeniana	0-5	15	5	80	grass, sedge, forb	13
2	Carex haydeniana	10-15	15	5	80	grass, sedge, forb	13
3	Poa leptocoma	20-25	15	5	80	grass, sedge, forb	13
21	Achilliea millefolium lanulosa	0-5	5	80	15	grass, sedge, forb	13
24	Trisetum spicatum	0-5	5	80	15	grass, sedge, forb	13
27	Castilleja sulphurea	5-10	5	80	15	grass, sedge, forb	13
29	Anemone parviflora	0-5	5	80	15	grass, sedge, forb	13
30	Senecio serratta	5-10					13
31	Senecio dimorphophyllus	10-15					13
37	Poa alpina	0-5					13
60	Rumex paucifolius	0-5	5	80	15	grass, sedge, forb	13
62	Senecio hydrophilus	10-15	5	80	15	grass, sedge, forb	13
67	Solidago spathulata nana	0-5	5	80	15	grass, sedge, forb	13

RDVW2B

Date: 7/24/2006 and 7/14/2007 %Slope: 5

UTM X: 509464 Aspect: East

UTMY: 4827283

Elevation (Meters): 2862

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
3	Poa leptocoma	0-5	20	65	15	grass, sedge, forb, short shrub	15
9	Erigeron rydbergii cronq.	0-5	20	65	15	grass, sedge, forb, short shrub	15
17	Anemone multifida tetonensis	0-5	20	65	15	grass, sedge, forb, short shrub	15
18	Antennaria microphylla	5-10	20	65	15	grass, sedge, forb, short shrub	15
21	Achilliea millefolium lanulosa	5-10	20	65	15	grass, sedge, forb, short shrub	15
25	Sedum lanceolatum	0-5	20	65	15	grass, sedge, forb, short shrub	15
26	Sedum debile	0-5	20	65	15	grass, sedge, forb, short shrub	15
30	Senecio serratta	5-10	20	65	15	grass, sedge, forb, short shrub	15
31	Senecio dimorphophyllus	5-10	20	65	15	grass, sedge, forb, short shrub	15
37	Poa alpina	15-20	20	65	15	grass, sedge, forb, short shrub	15
47	Carex breweri paddoensis	25-30	20	65	15	grass, sedge, forb, short shrub	15
48	Linanthus Pungens	0-5	20	65	15	grass, sedge, forb, short shrub	15
50	Salix drummondii	20-25	20	65	15	grass, sedge, forb, short shrub	15
64	Erigeron simplex	0-5	5	80	15	grass, sedge, forb, short shrub	15
67	Solidago spathulata nana	0-5	20	65	15	grass, sedge, forb, short shrub	15

RDVW3B

Date: 7/24/2006 and 7/14/2007 %Slope: 3 NOTES: 1m^2 %rock is all gravel

UTM X: 509492 Aspect: West

UTMY: 4827467

Elevation (Meters): 2851

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
17	Anemone multifida tetonensis	0-5	30	40	30	grass, sedge, forb, short shrub	15
18	Antennaria microphylla	5-10	30	40	30	grass, sedge, forb, short shrub	15
19	Phleum alpinum	0-5	30	40	30	grass, sedge, forb, short shrub	15
23	Sibaldia procumbens	0-5	30	40	30	grass, sedge, forb, short shrub	15
25	Sedum lanceolatum	0-5	30	40	30	grass, sedge, forb, short shrub	15
26	Sedum debile	0-5	30	40	30	grass, sedge, forb, short shrub	15
27	Castilleja sulphurea	0-5	30	40	30	grass, sedge, forb, short shrub	15
29	Anemone parviflora	0-5	30	40	30	grass, sedge, forb, short shrub	15
30	Senecio serratta	0-5					15
32	Epilobium brachyocarpum	0-5	30	40	30	grass, sedge, forb, short shrub	15
37	Poa alpina	0-5	30	40	30	grass, sedge, forb, short shrub	15
43	Pedicularis groenlandica	0-5	30	40	30	grass, sedge, forb, short shrub	15
50	Salix drummondii	30-35	30	40	30	grass, sedge, forb, short shrub	15
61	Hedysarum occitentale	0-5	30	40	30	grass, sedge, forb, short shrub	15
70	Potentilla diversifolia perdissecta	0-5	30	40	30	grass, sedge, forb, short shrub	15

MBD1A

Date: 8/4/2006 and 7/11/2007 %Slope: 5 NOTES: 1m^2 %rock is all gravel
UTM X: 512622 Aspect: South
UTMY: 4866552
Elevation (Meters): 2912

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
9	Erigeron rydbergii cronq.	0-5	10	70	20	grass, sedge, forbs	10
10	Ivesia gordonii	10-15	10	70	20	grass, sedge, forbs	10
21	Achilliea millefolium lanulosa	0-5	10	70	20	grass, sedge, forbs	10
33	Bistorta bistortoides	0-5	10	70	20	grass, sedge, forbs	10
37	Poa alpina	0-5	10	70	20	grass, sedge, forbs	10
63	Antennaria media	0-5	10	70	20	grass, sedge, forbs	10
72	UK	0-5	10	70	20	grass, sedge, forbs	10
73	moss 1		10	70	20	grass, sedge, forbs	10
82	UK	0-5	10	70	20	grass, sedge, forbs	10
83	moss 2	5-10	10	70	20	grass, sedge, forbs	10

MBD2A

Date: 8/4/2006 and 7/11/2007
UTM X: 512660
UTMY: 4866476
Elevation (Meters): 2927

%Slope: 3
Aspect: South

NOTES: 1m^2 rock is all gravel. 1m^2 veg cover has 35% moss and lichens

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	0-5	20	40	40	grass, sedge, forbs	16
5	Danthonia		20	40	40	grass, sedge, forbs	16
9	Erigeron rydbergii cronq.	0-5	20	40	40	grass, sedge, forbs	16
24	Trisetum spicatum	0-5	20	40	40	grass, sedge, forbs	16
25	Sedum lanceolatum	0-5	20	40	40	grass, sedge, forbs	16
27	Castilleja sulphurea	0-5	20	40	40	grass, sedge, forbs	16
34	Astragalus kentrophyta	15-20	20	40	40	grass, sedge, forbs	16
36	Petradoria pumila	0-5	20	40	40	grass, sedge, forbs	16
54	Carex elynoides	15-20	20	40	40	grass, sedge, forbs	16
57	Poa cusiskii epilis	0-5	20	40	40	grass, sedge, forbs	16
63	Antennaria media	0-5	20	40	40	grass, sedge, forbs	16
70	Potentilla diversifolia perdissecta	0-5	20	40	40	grass, sedge, forbs	16
72	UK	0-5	20	40	40	grass, sedge, forbs	16
74	UK	0-5	20	40	40	grass, sedge, forbs	16
83	moss 2	55-60	20	40	40	grass, sedge, forbs	16
85	lichens 1	35-40	20	40	40	grass, sedge, forbs	16

MBD3A

Date: 8/4/2006 and 7/11/2007 %Slope: 2
UTM X: 512940 Aspect: West
UTMY: 4866626
Elevation (Meters): 2881

Species #	**Species/genus Name**	**Species % cover**	**1m^2 % rock**	**1m^2 % bare ground**	**1m^2 % veg cover**	**community type**	**Number of Species Per Plot**
4	Elymus trachycaulus	0-5	0	5	95	grass, sedge, forbs, dwarf shrub	8
21	Achilliea millefolium lanulosa	0-5	0	5	95	grass, sedge, forbs, dwarf shrub	8
25	Sedum lanceolatum	0-5	0	5	95	grass, sedge, forbs, dwarf shrub	8
35	Potentilla fruticosa		0	5	95	grass, sedge, forbs, dwarf shrub	8
40	Epliobium glaberrinum fastigiatum	25-30	0	5	95	grass, sedge, forbs, dwarf shrub	8
66	UK	0-5	0	5	95	grass, sedge, forbs, dwarf shrub	8
68	UK	15-20	0	5	95	grass, sedge, forbs, dwarf shrub	8
76	UK	75-80	0	5	95	grass, sedge, forbs, dwarf shrub	8

MLD1A

Date: 8/14/2006 and 7/20/2007
UTM X: 515142
UTMY: 4849303
Elevation (Meters): 3031

%Slope: 5
Aspect: South

NOTES: $1m^2$ Rock is 50% gravel. $1m^2$ veg cover is 5% lichens

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	5-10	60	30	10	n/a	12
8	Erigeron speciosus	0-5					12
15	Androsace septentrionalis	0-5					12
22	Antennaria umbrinella	0-5	60	30	10	n/a	12
23	Sibaldia procumbens	5-10	60	30	10	n/a	12
38	Carex paysonis	0-5	60	30	10	n/a	12
46	Hieracium fendleri	0-5	60	30	10	n/a	12
57	Poa cusiskii epilis	5-10	60	30	10	n/a	12
64	Erigeron simplex	0-5	60	30	10	n/a	12
65	Boechera lyallii	0-5	60	30	10	n/a	12
82	UK	0-5	60	30	10	n/a	12
85	lichens 1	10-15	60	30	10	n/a	12

MLD2A

Date: 8/14/2006 and 7/20/2007
UTM X: 515214
UTMY: 4849371
Elevation (Meters): 3070

%Slope: 12
Aspect: North

NOTES: $1m^2$ veg cover is 50% cryptobiotic crust

Species #	**Species/genus Name**	**Species % cover**	**$1m^2$ % rock**	**$1m^2$ % bare ground**	**$1m^2$ % veg cover**	**community type**	**Number of Species Per Plot**
1	Juncus parryi	5-10	30	0	70	grass, forb	10
6	Luzula parviflora	0-5					10
15	Androsace spetentionalis	0-5					10
22	Antennaria umbrinella	5-10	30	0	70	grass, forb	10
23	Sibaldia procumbens	0-5	30	0	70	grass, forb	10
38	Carex paysonis	5-10					10
46	Hieracium fendleri	0-5	30	0	70	grass, forb	10
65	Boechera lyallii	0-5	30	0	70	grass, forb	10
85	lichens 1	25-30	30	0	70	grass, forb	10
86	lichens 2	5-10	30	0	70	grass, forb	10

MLD3A

Date: 8/14/2006 and 7/20/2007
UTM X: 515159
UTMY: 4849385
Elevation (Meters): 3067

%Slope: 6
Aspect: South/East

NOTES: 1m^2 rock is 65% gravel 5% cobble

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	10-15	70	20	10	grass, forb	12
6	Luzula parviflora	0-5	70	20	10	grass, forb	12
8	Erigeron speciosus	0-5	70	20	10	grass, forb	12
14	Minuartia austromontana	0-5	70	20	10	grass, forb	12
22	Antennaria umbrinella	0-5	70	20	10	grass, forb	12
23	Sibaldia procumbens	0-5	70	20	10	grass, forb	12
33	Bistorta bistortoides	0-5	70	20	10	grass, forb	12
37	Poa alpina	0-5	70	20	10	grass, forb	12
46	Hieracium fendleri	0-5	70	20	10	grass, forb	12
57	Poa cusiskii epilis	5-10	70	20	10	grass, forb	12
64	Erigeron simplex	0-5	70	20	10	grass, forb	12
86	lichens 2	35-40	70	20	10	grass, forb	12

RDVD1A

Date: 7/24/2006 and 7/14/2007 %Slope: 6
UTM X: 509321 Aspect: East
UTMY: 4827587
Elevation (Meters): 2925

Species #	**Species/genus Name**	**Species % cover**	**1m^2 % rock**	**1m^2 % bare ground**	**1m^2 % veg cover**	**community type**	**Number of Species Per Plot**
2	Carex haydeniana	10-15	0	60	40	grass, sedge, forb	8
3	Poa leptocoma	15-20	0	60	40	grass, sedge, forb	8
15	Androsace spetentionalis	15-20	0	60	40	grass, sedge, forb	8
19	Phleum alpinum	0-5	0	60	40	grass, sedge, forb	8
48	Linanthus Pungens	0-5	0	60	40	grass, sedge, forb	8
52	Boechera microphylla	5-10	0	60	40	grass, sedge, forb	8
60	Rumex paucifolius	20-25	0	60	40	grass, sedge, forb	8
82	UK	0-5	0	60	40	grass, sedge, forb	8

RDVD2A

Date: 7/24/2006 and 7/14/2007
UTM X: 509332
UTMY: 4827647
Elevation (Meters): 2915

%Slope: 3
Aspect: West

NOTES: 1m^2 rock is 25% cobble and 10% gravel

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	0-5	35	30	35	grass, sedge, forb	16
2	Carex haydeniana	0-5	35	30	35	grass, sedge, forb	16
15	Androsace spetentionalis	0-5	35	30	35	grass, sedge, forb	16
17	Anemone multifida tetonensis	0-5	35	30	35	grass, sedge, forb	16
20	Castilleja miniata	0-5	35	30	35	grass, sedge, forb	16
24	Trisetum spicatum	0-5	35	30	35	grass, sedge, forb	16
25	Sedum lanceolatum	0-5	35	30	35	grass, sedge, forb	16
29	Anemone parviflora	0-5	35	30	35	grass, sedge, forb	16
48	Linanthus Pungens	15-20	35	30	35	grass, sedge, forb	16
51	Ranunculus alismifolius	0-5	35	30	35	grass, sedge, forb	16
52	Boechera microphylla	0-5	35	30	35	grass, sedge, forb	16
53	Symphyotrichum foliaceus	5-10	35	30	35	grass, sedge, forb	16
57	Poa cusiskii epilis	0-5	35	30	35	grass, sedge, forb	16
59	Poa cusiskii epilis	0-5	35	30	35	grass, sedge, forb	16
60	Rumex paucifolius	15-20	35	30	35	grass, sedge, forb	16
65	Boechera lyallii	0-5	35	30	35	grass, sedge, forb	16

RDVD3A

Date: 7/24/2006 and 7/14/2007 %Slope: 1
UTM X: 509358 Aspect: North
UTMY: 4827748
Elevation (Meters): 2926

Species #	**Species/genus Name**	**Species % cover**	**1m^2 % rock**	**1m^2 % bare ground**	**1m^2 % veg cover**	**community type**	**Number of Species Per Plot**
2	Carex haydeniana	5-10	5	60	35	grass, sedge, forb	16
9	Erigeron rydbergii cronq.	5-10	5	60	35	grass, sedge, forb	16
15	Androsace spetentionalis	0-5	5	60	35	grass, sedge, forb	16
20	Castilleja miniata	0-5	5	60	35	grass, sedge, forb	16
21	Achilliea millefolium lanulosa	0-5	5	60	35	grass, sedge, forb	16
23	Sibaldia procumbens	0-5	5	60	35	grass, sedge, forb	16
24	Trisetum spicatum	15-20	5	60	35	grass, sedge, forb	16
32	Epilobium brachyocarpum	0-5	5	60	35	grass, sedge, forb	16
38	Carex paysonis	0-5	5	60	35	grass, sedge, forb	16
52	Boechera microphylla	5-10	5	60	35	grass, sedge, forb	16
53	Symphyotrichum foliaceus	0-5	5	60	35	grass, sedge, forb	16
56	Lithophragma glabrum	0-5	5	60	35	grass, sedge, forb	16
58	Geranium richardsonii	5-10	5	60	35	grass, sedge, forb	16
60	Rumex paucifolius	5-10	5	60	35	grass, sedge, forb	16
64	Erigeron simplex	0-5	5	60	35	grass, sedge, forb	16
85	lichens 1	0-5	5	60	35	grass, sedge, forb	16

MBD1B

Date: 8/4/2006 and 7/11/2007	%Slope: 5
UTM X: 512622	Aspect: South
UTMY: 4866552	
Elevation (Meters): 2912	

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	0-5				grass, sedge, forbs	15
5	Danthonia					grass, sedge, forbs	15
9	Erigeron rydbergii cronq.	0-5				grass, sedge, forbs	15
10	Ivesia gordonii	10-15				grass, sedge, forbs	15
14	Minuartia austromontana	0-5				grass, sedge, forbs	15
21	Achilliea millefolium lanulosa	0-5				grass, sedge, forbs	15
24	Trisetum spicatum	0-5				grass, sedge, forbs	15
25	Sedum lanceolatum	0-5				grass, sedge, forbs	15
34	Astragalus kentrophyta	5-10				grass, sedge, forbs	15
37	Poa alpina	0-5				grass, sedge, forbs	15
41	Erigeron formosissimus	0-5					15
47	Carex breweri paddoensis	0-5				grass, sedge, forbs	15
53	Symphyotrichum foliaceus	0-5				grass, sedge, forbs	15
54	Carex elynoides	0-5				grass, sedge, forbs	15
83	moss 2	15-20				grass, sedge, forbs	15

MBD2B

Date: 8/4/2006 and 7/11/2007 %Slope: 3
UTM X: 512660 Aspect: South
UTMY: 4866476
Elevation (Meters): 2927

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
5	Danthonia					grass, sedge, forbs	9
21	Achilliea millefolium lanulosa	0-5				grass, sedge, forbs	9
24	Trisetum spicatum	0-5				grass, sedge, forbs	9
34	Astragalus kentrophyta	5-10				grass, sedge, forbs	9
36	Petradoria pumila	5-10				grass, sedge, forbs	9
54	Carex elynoides	15-20				grass, sedge, forbs	9
69	Polygonum douglasii	0-5				grass, sedge, forbs	9
70	Potentilla diversifolia perdissecta	0-5				grass, sedge, forbs	9
83	moss 2	35-40				grass, sedge, forbs	9

MBD3B

Date: 8/4/2006 and 7/11/2007 %Slope: 2
UTM X: 512940 Aspect: West
UTMY: 4866626
Elevation (Meters): 2881

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
2	Carex haydeniana	10-15				grass, sedge, forbs, dwarf shrub	17
14	Minuartia austromontana	0-5				grass, sedge, forbs, dwarf shrub	17
21	Achilliea millefolium lanulosa	10-15				grass, sedge, forbs, dwarf shrub	17
27	Castilleja sulphurea	0-5				grass, sedge, forbs, dwarf shrub	17
34	Astragalus kentrophyta	0-5				grass, sedge, forbs, dwarf shrub	17
35	Potentilla fruticosa					grass, sedge, forbs, dwarf shrub	17
37	Poa alpina	0-5				grass, sedge, forbs, dwarf shrub	17
40	Epliobium glaberrinum fastigiatum	15-20				grass, sedge, forbs, dwarf shrub	17
53	Symphyotrichum foliaceus	5-10				grass, sedge, forbs, dwarf shrub	17
54	Carex elynoides	0-5				grass, sedge, forbs, dwarf shrub	17
63	Antennaria media	0-5				grass, sedge, forbs, dwarf shrub	17
68	UK	10-15				grass, sedge, forbs, dwarf shrub	17
70	Potentilla diversifolia perdissecta	0-5				grass, sedge, forbs, dwarf shrub	17
72	UK	5-10				grass, sedge, forbs, dwarf shrub	17
77	UK	0-5				grass, sedge, forbs, dwarf shrub	17
79	UK	0-5				grass, sedge, forbs, dwarf shrub	17
83	moss 2	5-10				grass, sedge, forbs, dwarf shrub	17

MLD1B

Date: 8/14/2006 and 7/20/2007

UTM X: 515142

UTMY: 4849303

Elevation (Meters): 3081

%Slope: 5

Aspect: South

NOTES: $1m^2$ veg cover is 20% cryptobiotic crust

Species #	**Species/genus Name**	**Species % cover**	**$1m^2$ % rock**	**$1m^2$ % bare ground**	**$1m^2$ % veg cover**	**community type**	**Number of Species Per Plot**
1	Juncus parryi	20-25	0	50	50	n/a	12
8	Erigeron speciosus	5-10					12
14	Minuartia austromontana	0-5					12
15	Androsace spetentionalis	0-5					12
22	Antennaria umbrinella	5-10	0	50	50	n/a	12
23	Sibaldia procumbens	0-5	0	50	50	n/a	12
33	Bistorta bistortoides	0-5	0	50	50	n/a	12
38	Carex paysonis	0-5	0	50	50	n/a	12
46	Hieracium fendleri	0-5	0	50	50	n/a	12
57	Poa cusiskii epilis	20-25	0	50	50	n/a	12
64	Erigeron simplex	5-10	0	50	50	n/a	12
85	lichens 1	25-30	0	50	50	n/a	12

MLD2B

Date: 8/14/2006 and 7/20/2007
UTM X: 515214
UTMY: 4849371
Elevation (Meters): 3070

%Slope: 12
Aspect: North

NOLTES: $1m^2$ rock is all cobble. $1m^2$ veg cover is 5% lichens

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
6	Luzula parviflora	0-5	90	0	10	grass, forb	10
14	Minuartia austromontana	0-5	90	0	10	grass, forb	10
15	Androsace spetentionalis	0-5					10
22	Antennaria umbrinella	5-10	90	0	10	grass, forb	10
23	Sibaldia procumbens	0-5	90	0	10	grass, forb	10
38	Carex paysonis	15-20	90	0	10	grass, forb	10
46	Hieracium fendleri	0-5	90	0	10	grass, forb	10
57	Poa cusiskii epilis	0-5	90	0	10	grass, forb	10
70	Potentilla diversifolia perdissecta	0-5	90	0	10	grass, forb	10
85	lichens 1	0-5	90	0	10	grass, forb	10

MLD3B

Date: 8/14/2006 and 7/20/2007 %Slope: 6 NOTES: 1m^2 rock is all gravel
UTM X: 515159 Aspect: South/Eas
UTMY: 4849385
Elevation (Meters): 3067

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	5-10	70	20	10	grass, forb	12
6	Luzula parviflora	0-5	70	20	10	grass, forb	12
14	Minuartia austromontana	0-5	70	20	10	grass, forb	12
22	Antennaria umbrinella	5-10	70	20	10	grass, forb	12
23	Sibaldia procumbens	5-10	70	20	10	grass, forb	12
33	Bistorta bistortoides	0-5	70	20	10	grass, forb	12
38	Carex paysonis	0-5	70	20	10	grass, forb	12
44	Carex leporinella	0-5	70	20	10	grass, forb	12
46	Hieracium fendleri	0-5	70	20	10	grass, forb	12
57	Poa cusiskii epilis	0-5	70	20	10	grass, forb	12
85	lichens 1	15-20	70	20	10	grass, forb	12
86	lichens 2	15-20	70	20	10	grass, forb	12

RDVD1B

Date: 7/24/2006 and 7/14/2007 %Slope: 6
UTM X: 509321 Aspect: East
UTMY: 4827587
Elevation (Meters): 2925

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	0-5	5	80	15	grass, sedge, forb	13
2	Carex haydeniana	0-5	5	80	15	grass, sedge, forb	13
4	Elymus trachycaulus	0-5	5	80	15	grass, sedge, forb	13
9	Erigeron rydbergii cronq.	10-15	5	80	15	grass, sedge, forb	13
15	Androsace spetentionalis	0-5	5	80	15	grass, sedge, forb	13
25	Sedum lanceolatum	0-5	5	80	15	grass, sedge, forb	13
48	Linanthus Pungens	5-10	5	80	15	grass, sedge, forb	13
52	Boechera microphylla	0-5	5	80	15	grass, sedge, forb	13
52	Boechera microphylla	0-5	5	80	15	grass, sedge, forb	13
53	Symphyotrichum foliaceus	0-5	5	80	15	grass, sedge, forb	13
58	Geranium richardsonii	5-10	5	80	15	grass, sedge, forb	13
60	Rumex paucifolius	0-5	5	80	15	grass, sedge, forb	13
65	Boechera lyallii	0-5	5	80	15	grass, sedge, forb	13

RDVD2B

Date: 7/24/2006 and 7/14/2007 %Slope: 3
UTM X: 509332 Aspect: West
UTMY: 4827647
Elevation (Meters): 2915

Species #	Species/genus Name	Species % cover	$1m^2$ % rock	$1m^2$ % bare ground	$1m^2$ % veg cover	community type	Number of Species Per Plot
9	Erigeron rydbergii cronq.	5-10	60	20	20	grass, sedge, forb	12
15	Androsace spetentionalis	0-5	60	20	20	grass, sedge, forb	12
16	Erigonum jamesii	5-10	60	20	20	grass, sedge, forb	12
24	Trisetum spicatum	0-5	60	20	20	grass, sedge, forb	12
25	Sedum lanceolatum	0-5	60	20	20	grass, sedge, forb	12
32	Epilobium brachyocarpum	0-5	60	20	20	grass, sedge, forb	12
47	Carex breweri paddoensis	5-10	60	20	20	grass, sedge, forb	12
49	Penstemon rydbergii	0-5	60	20	20	grass, sedge, forb	12
51	Ranunculus alismifolius	0-5	60	20	20	grass, sedge, forb	12
52	Boechera microphylla	0-5	60	20	20	grass, sedge, forb	12
57	Poa cusiskii epilis	0-5	60	20	20	grass, sedge, forb	12
60	Rumex paucifolius	0-5	60	20	20	grass, sedge, forb	12

RDVD3B

Date: 7/24/2006 and 7/14/2007 %Slope: 1
UTM X: 509358 Aspect: North
UTMY: 4827748
Elevation (Meters): 2926

Species #	Species/genus Name	Species % cover	1m^2 % rock	1m^2 % bare ground	1m^2 % veg cover	community type	Number of Species Per Plot
1	Juncus parryi	5-10	5	30	65	grass, sedge, forb	15
2	Carex haydeniana	15-20	5	30	65	grass, sedge, forb	15
3	Poa leptocoma	0-5	5	30	65	grass, sedge, forb	15
4	Elymus trachycaulus	0-5	5	30	65	grass, sedge, forb	15
9	Erigeron rydbergii cronq.	5-10	5	30	65	grass, sedge, forb	15
19	Phleum alpinum	0-5	5	30	65	grass, sedge, forb	15
30	Senecio serratta	0-5					15
48	Linanthus Pungens	5-10	5	30	65	grass, sedge, forb	15
52	Boechera microphylla	0-5	5	30	65	grass, sedge, forb	15
53	Symphyotrichum foliaceus	0-5	5	30	65	grass, sedge, forb	15
56	Lithophragma glabrum	0-5	5	30	65	grass, sedge, forb	15
58	Geranium richardsonii	5-10	5	30	65	grass, sedge, forb	15
60	Rumex paucifolius	10-15	5	30	65	grass, sedge, forb	15
64	Erigeron simplex	0-5	5	30	65	grass, sedge, forb	15
65	Boechera lyallii	0-5	5	30	65	grass, sedge, forb	15