

8 NORTHWESTERN FORESTED MOUNTAINS

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8.1 Ecoregion Description

The Northwestern Forested Mountains are ecologically diverse and geographically widespread, encompassing the mountain ecosystems of central and northwestern North America (CEC 1997; Figure 2.2). The ecoregion description is adapted from CEC (1997). Geographically, they extend from the Rocky Mountains and the Sierra Nevada north through the Siskiyou, the east side of the Cascade Range, and then east of the Coast Ranges to interior Alaska. Climatically, the region is characterized by a transition from a moist, maritime climate in the northwest, to a continental and drier climate in the Rockies in the southeast. Orographically generated rainfall creates both rain shadows and wet belts, often in close proximity.

The vegetation of the ecoregion is extremely diverse, with distinct community zonation occurring along elevation gradients. Alpine communities at the highest elevations contain various forb, lichen, and shrub associations. Subalpine communities include lodgepole pine (*Pinus contorta*), subalpine fir (*Abies lasiocarpa*), Pacific silver fir (*Abies amabilis*), grand fir (*Abies grandis*), and Engelmann spruce (*Picea engelmannii*). Mid-elevation forests are characterized by ponderosa pine (*Pinus ponderosa*), Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), lodgepole pine, and quaking aspen (*Populus tremuloides*) in the east, and by western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), Douglas-fir (*Pseudotsuga menziesii*), and western white pine (*Pinus monticola*) in the west and southwest. White and black spruce (*Picea glauca* and *P. mariana*) dominate the Alaskan portion of the ecoregion. Vegetation of the interior valleys in the southern portion of the region includes big sagebrush (*Artemisia tridentata*), rabbitbrush (*Chrysothamnus* spp.), and antelope bitterbrush (*Purshia tridentata*).

8.2 Ecosystem Responses to N Deposition

The wide diversity of communities and climates, along with diverse bedrock geologies, results in substantial variation in ecological responses to nitrogen (N) deposition. Unfortunately, little information exists on the response of many areas in the region. Documented responses to elevated N deposition within this bioregion include alteration of soils (carbon:nitrogen (C:N) ratios, base cation composition, and N cycling rates, including mineralization and nitrification), plant, lichen, and algal chemistry (N concentration and nitrogen:calcium (N:Ca), nitrogen:magnesium (N:Mg), nitrogen:phosphorus (N:P), and C:N ratios; lichen thallus N concentrations), surface water N concentration and acid-neutralizing capacity, catchment N leaching rate, and changes in the community composition of plants, lichens, and phytoplankton. The region provides several important ecosystem services that may be compromised by N deposition, including water supply for much of western North America, timber, forage for livestock, and recreation.

8.3 Range of Responses Observed

8.3.1 Forest

Much of the research on ecological responses to N deposition in this ecoregion has been conducted in the Colorado Front Range of the Rocky Mountains, where a strong connection between regional urban and agricultural emissions and adjacent wildlands responses has been documented (e.g., Rocky Mountain National Park; Baron et al. 2000, Gebhart et al. 2011). Responses to N deposition are shown in Table 8.1. Rueth and Baron (2002) evaluated characteristics of subalpine Engelmann spruce stands spanning the Continental Divide across a N deposition gradient from 1-2 to 3-5 kg ha⁻¹ yr⁻¹. Deposition reported is total (wet + dry) at the eastern end of the deposition

Table 8.1—Observed and modeled responses to N inputs for the Northwestern Forested Mountains ecoregion. Reliability rating: ## reliable; # fairly reliable; (#) expert judgment

Ecosystem	Site	N deposition <i>kg ha⁻¹ yr⁻¹</i>	Reliability	Response	Comments	Study
Coniferous forest	Alaska	1.2-3.7	(#)	Lichen community composition	Application of western Oregon and Washington model	Geiser et al. 2010
Alpine lake	S. Rockies/Loch Vale RMNP	1.5	##	Diatom assemblages	Wet deposition estimate	Baron 2006
Coniferous forest	Northwestern Forested Mountains, non-Alaska	2.5-7.1	##	Lichen community composition	Application of western Oregon and Washington model, Sierra thresholds	Geiser et al. 2010
Alpine	S. Rockies/ Rocky Mountain National Park	3-4	(#)	Soil chemistry		Baron et al. 1994
Coniferous forest	Sierra Nevada	3.1-5.2	##	Lichen community composition	N measured as canopy throughfall	Fenn et al. 2008
Subalpine Forest	S. Rockies/12 stands of paired high and low N deposition	<4	##	Foliar chemistry, mineralization, nitrification, initial increases in N leaching below the organic layer	Effects are cumulative over time	Rueth and Baron 2002
Subalpine Forest	S. Rockies/fertilized plots in Fraser and Loch Vale	<4	##	Foliar chemistry, mineralization, nitrification	Effects are cumulative over time	Rueth et al. 2003
Alpine vegetation	S. Rockies/ Niwot Ridge	4	##	Species composition ; individual species responses		Bowman et al. 2006
Alpine lake	S. Rockies/Niwot	4.0	#	Episodic freshwater acidification		Williams and Tonnesson 2000
Hardwood and coniferous forests	Columbia River Gorge	4-5	#	Lichen community composition	N measured as canopy throughfall	Geiser et al. in 2010
Alpine stream	S. Rockies/Loch Vale RMNP	5	(#)	Episodic freshwater acidification		Baron and Hartman ^a , model results
Alpine and subalpine lakes and streams	Sierra Nevada	> 5	##	NO ₃ ⁻ leaching	Increased NO ₃ ⁻ leaching not observed at the highest deposition (5 kg ha ⁻¹ yr ⁻¹)	Sickman et al. 2001

Alpine stream	S. Rockies/Loch Vale RMNP	7.3	(#)	Chronic freshwater acidification		Baron and Hartman ^a , model results
Alpine stream	S. Rockies/Loch Vale RMNP	8	(#)	Freshwater acidification	ANC=20 µeq/L	Sullivan et al. 2005 (modeled)
Alpine vegetation	S. Rockies/ Niwot Ridge	10	###	Species composition; whole community		Bowman et al. 2006
Subalpine stream/lake	S. Rockies/Loch Vale RMNP	12	(#)	Freshwater acidification	ANC=20 µeq/L	Sullivan et al. 2005 (modeled)
Alpine stream	S. Rockies/Loch Vale RMNP	14	(#)	Freshwater acidification	ANC=0 µ/L	Sullivan et al. 2005 (modeled)
Mixed conifer forest	Sierra Nevada	17	###	NO ₃ ⁻ leaching; reduced fine root biomass	Critical load not exceeded in NW forests; critical load based on SW Sierra Nevada and San Bernardino Mts	Fenn et al. 2008
Alpine terrestrial	S. Rockies / Niwot Ridge	>20	#	NO ₃ ⁻ leaching, soil nitrogen fluxes		Bowman et al. 2006
Subalpine stream/lake	S. Rockies/Loch Vale RMNP	21	(#)	Freshwater acidification	ANC=0 µeq/L	Sullivan et al. 2005 (modeled)

^aBaron, J.S.; Hartman, M.D. Unpublished data. On file with Jill Baron, Research ecologist, U.S. Geological Survey, Natural Resource Ecology Laboratory, Colorado State University, Fort Collins CO 80523-1499

gradient; wet deposition only is reported in the west where dry deposition was assumed to be insignificant (Rueth and Baron 2002). At the eastern, high end of the deposition gradient, forest stands had significantly lower organic soil horizon C:N, higher N mineralization rates, higher potential net nitrification rates and foliar N concentration, and higher N:P, N:Ca, and N:Mg ratios. Nitrate (NO_3^-) leaching was elevated at the eastern, high end of the deposition gradient (Baron⁶, Rueth and Baron 2002). An earlier study reported elevated foliar N:P ratios in high elevation bristlecone pine (*Pinus aristata*) stands receiving N deposition of $6 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Williams et al. 1996).

Four years of low-level ($25 \text{ kg ha}^{-1} \text{ yr}^{-1}$) ammonium nitrate (NH_4NO_3) fertilization to old-growth coniferous forests with different initial levels of atmospheric N deposition showed different responses based on initial conditions (Rueth et al. 2003). The low deposition site with initial organic horizon C:N of 36 and N pool of 605 kg N ha^{-1} showed no significant increase in N mineralization rates. At this low deposition site, foliar and soil organic horizon C:N decreased significantly with fertilization. In contrast, N mineralization rates and extractable inorganic N in soil increased significantly at the higher N deposition site with greater initial soil N (C:N of 24, N pool of 991 kg N ha^{-1}). At this higher N deposition site, foliar and soil organic horizon percentages of N did not change (Rueth et al. 2003).

In mixed conifer forests of the southwestern Sierra Nevada (See Chapter 13, Mediterranean California), NO_3^- leaching increases at a deposition of $17 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, as determined from empirical data and from biogeochemical modeling (Fenn et al. 2008). At this level of N deposition, fine root biomass was reduced by 26 percent. The highest throughfall N deposition inputs reported for the northern and eastern regions of the Sierra Nevada range are $7 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Fenn et al. 2008). Mixed conifer forests in California with these levels of N deposition are highly conservative of N, with no appreciable hydrologic NO_3^- leaching or

gaseous N losses from soil (Fenn et al. 2008). Effects on soils and vegetation could be occurring in the most polluted northern and eastern regions, but are likely to be subtle and may not have been observed for lack of in-depth study. However, Community Multi-scale Air Quality (CMAQ) simulations of N deposition indicate that N inputs may be as high as 11 to $17 \text{ kg ha}^{-1} \text{ yr}^{-1}$ along portions of the western edge of the central and northern Sierra Nevada range (Fenn et al. 2010). If deposition actually occurs at these levels in some sites, N enrichment of soil and plants are expected. Increased NO_3^- leaching and gaseous N loss may also occur at the upper end of this deposition range (Fenn et al. 2008). Effects of N deposition on biodiversity of understory or shrub communities in the Sierra Nevada range have not been studied, although plant community composition of many nutrient-poor low biomass ecosystems in California are likely impacted by deposition levels $\leq 10 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Weiss 2006). Weiss (2006) estimated that $10,000 \text{ km}^2$ in California are vulnerable to plant community impacts from N deposition $> 10 \text{ kg ha}^{-1} \text{ yr}^{-1}$.

The most notable effect of N deposition in the low to moderately polluted forested regions of the Sierra Nevada range (N deposition of 3 to $7 \text{ kg ha}^{-1} \text{ yr}^{-1}$) is the highly significant alteration of lichen communities (Fenn et al. 2008).

Ectomycorrhizal fungi. There is no information on ectomycorrhizal community responses to N in the Northwestern Forested Mountains ecoregion. We postulate that, given the climate, species composition and soils, white and black spruce dominated ecosystems may have similar deposition thresholds to those seen in spruce forests of the same species in coastal Alaska (see Chapter 9, West Coast Marine Forests). Similarly, deposition thresholds for Engelmann spruce-subalpine fir dominated ecosystems may be similar to those of the spruce-fir forests of the northeastern United States (see Chapter 7, Northern Forests).

Forest Lichen Communities. In the California Sierra Nevada range, a strong ammonia (NH_3) deposition gradient was positively correlated with NH_3 indicator lichen species, or eutrophs ($r = 0.93$). The greatest lichen

⁶Baron, J.S. Unpublished data. Research ecologist, U.S. Geological Survey, Natural Resource Ecology Laboratory, Colorado State University, Fort Collins CO 80523-1499

community impacts were observed in the portions of the Sierra Nevada range in the southwest and near the Tahoe Basin, and the Modoc Plateau, near urban areas and at popular recreation areas in Sequoia and Yosemite National Parks (Jovan and McCune 2006). Increased N content of the lichen epiphyte *Letharia vulpina*, as defined by a threshold concentration of 1.0 percent, was also observed in these areas (Jovan and Carlberg 2006). Thallus N concentrations above this threshold are correlated with decreasing representation of oligotrophic species relative to mesotrophic and eutrophic species within epiphytic macrolichen communities of the Sierra Nevada (Fenn et al. 2008). Thallus N concentrations below the threshold represent the clean site range; at these sites no community shifts are detected. Information on lichen research methodology and terminology can be found in Chapter 4 of this volume.

Within the mixed conifer forests in California, which extend from the west side of the California Sierra Nevada from Tahoe National Forest south through Sequoia-Kings Canyon National Park, simple indices of three N-indicator groups—the oligotrophs (adapted to low nutrient availability), the mesotrophs (requiring moderate nutrient levels and tolerating some anthropogenic enhancement but not extreme levels of N deposition) and the eutrophs (nitrophilous species that respond favorably to enhanced N deposition)—were used to identify N loads that correspond with major shifts in Sierra Nevada lichen communities. At 3.1 kg N ha⁻¹ yr⁻¹ (95 percent confidence interval: 0.70 to 5.5 kg ha⁻¹ yr⁻¹; Fenn et al. 2008; see also Table 8.2), the N concentration threshold in the lichen *Letharia vulpina* was exceeded, and the lichen community composition was already shifting from oligotrophic to more N-tolerant mesotrophic and eutrophic species. At an estimated N deposition of 5.7 kg ha⁻¹ yr⁻¹ the lichen community had shifted from the natural state of oligotroph dominance. This change is of particular concern because oligotrophic species are important components of winter food webs, hydrologic and nutrient cycles, and wildlife habitat (McCune et al. 2007, McCune and Geiser 2009). The data from this study predict a complete extirpation of oligotrophs from the lichen community at an N load of about 10.2

kg N ha⁻¹ yr⁻¹. This work demonstrates that known biological impacts are occurring at N deposition levels as low as 3 to 5 kg ha⁻¹ yr⁻¹, levels which are exceeded over large areas of the forests of California: most of the forested region of the San Bernardino and San Gabriel Mountains, the western portions of the Sierra Nevada, and large portions of other mountain ranges in southern California (Fenn et al. 2003a, b; Fenn et al. 2008, Fenn et al. 2010).

Geiser et al. (2010) applied a model developed for Oregon and Washington forests to Sierra Nevada lichen communities, using the same response thresholds as Fenn et al. (2008) and median site precipitation (111 cm). At 2.5 to 3.8 kg N ha⁻¹ yr⁻¹, some sensitive oligotrophic species were modeled to be absent; at 4.6 to 5.7 kg N ha⁻¹ yr⁻¹, the lichen community would shift away from oligotroph dominance. These results are very close to the results obtained by Fenn et al. (2008). Fenn et al. (2008) did not account for precipitation in the Sierra Nevada study, as recommended by Geiser et al. (2010) for Oregon and Washington forests. Because mean annual precipitation across Sierra Nevada study sites varied only twofold (79 to 165 cm), it was probably less influential there than in Oregon and Washington, where precipitation varied tenfold. If the entire precipitation range in the ecoregion were considered, the Geiser et al. 2010 model predicts a lichen response threshold of 2.5 to 7.1 kg N ha⁻¹ yr⁻¹ for the Northwestern Forested Mountains ecoregion in the contiguous United States (30 to 203 cm), and 1.2 to 3.7 kg N ha⁻¹ yr⁻¹ (30 to 80 cm) for Alaska (See Table 4.1 for details).

Along the Snake River and its tributaries in Idaho and Oregon, within Hells Canyon National Recreation Area, comparatively high NH₄NO₃ in fine particulates originating from the agriculturally intensive Snake River basin accounted for increased lichen N throughout the Recreation Area relative to other remote sites in Oregon, Washington, Wyoming, and Colorado. Average seasonal NH₃ concentrations between 4.2 and 7.7 parts per billion (ppb), with 2-week averages peaking at 9 to 18 ppb, were associated with significantly higher cover of the nitrophilous lichens *Xanthomendoza* and *Xanthoria*,

Table 8.2—Responses to N input for lichens of the Northwestern Forested Mountains ecoregion.

Ecosystem	Site	N levels generating observed response	Measured response	Comments	Study
Coniferous forest	Central-southern Sierra Nevada	3.1 kg N ha ⁻¹ yr ⁻¹	Some sensitive lichen species absent	Critical load based on exceedance of a N concentration threshold in the lichen <i>Letharia vulpina</i>	Fenn et al. 2008
Coniferous forest	Central-southern Sierra Nevada	2.5-3.8 kg N ha ⁻¹ yr ⁻¹	Some sensitive lichen species absent	Application of Oregon/Washington model using Fenn et al. 2008 response threshold and median precipitation	Geiser et al. 2010
Coniferous forests	Sierra Nevada (Tahoe)	4-5 kg N ha ⁻¹ yr ⁻¹	Lichen N > 1.03% (<i>Letharia vulpina</i>).	In the SW Sierras, Tahoe and Modoc Plateau	Jovan and Cariberg 2006, Jovan and McCune 2006
Coniferous forest	Central-southern Sierra Nevada	5.7 kg N ha ⁻¹ yr ⁻¹	50 % sensitive lichen species absent	Critical load based on exceedance of a N concentration threshold in the lichen <i>Letharia vulpina</i>	Fenn et al. 2008
Coniferous forest	Central-southern Sierra Nevada	4.6-5.7 kg N ha ⁻¹ yr ⁻¹	50 % sensitive lichen species absent	Application of Oregon/Washington model using Fenn et al. 2008 response threshold and median precipitation	Geiser et al. 2010
Coniferous forests	Sierra Nevada (Sequoia NP)	6-15 kg N ha ⁻¹ yr ⁻¹	Lichen N > 1.03% (<i>Letharia vulpina</i>).	N was mostly NH ₃ ; N content was elevated in the SW Sierras, Tahoe and Modoc Plateau	Jovan and Cariberg 2006, Jovan and McCune 2006
Coniferous forest	Central-southern Sierra Nevada	10.2 kg N ha ⁻¹ yr ⁻¹	Extirpation of all sensitive lichen species	Critical load based on exceedance of a N concentration threshold in the lichen <i>Letharia vulpina</i>	Fenn et al. 2008
Coniferous forest	Columbia River Gorge OR/WA	<11.5 kg ha ⁻¹ in 4.5 months	Loss of sensitive species; lichen N > 1.5% (<i>Xanthoparmelia cumberlandia</i>).	Lichen N and measured N in throughfall were correlated. Lichen N concentration exceeded the threshold at all sites	Fenn et al. 2007
Deciduous riparian forests	Hells Canyon NRA, Snake River OR/ID	4.2-7.7 µg NH ₃ m ⁻³ annual mean; 5-20 µg NH ₃ m ⁻³ summer biweekly peaks	Increased cover of eutrophs; lichen N > 1.5% (<i>Xanthoparmelia cumberlandia</i>)	4.2-7.7 µg NH ₃ m ⁻³ annual mean; 5-20 µg NH ₃ m ⁻³ summer biweekly peaks	Geiser et al. 2008

and higher lichen N concentrations in *Xanthoparmelia cumberlandia* (Geiser et al. 2008). Similarly, abundance of the lichen epiphyte, *Xanthomendoza fallax*, in northern Utah and southern Idaho, increased with proximity to peak urban (Logan, Utah) and agricultural NH₃ emission centers, where summertime NH₃ ranged from 7.3 to 92.2 µg m⁻³. Much of the effect was attributed to long distance transport and deposition of NH₄⁺ (Rogers et al. 2009).

Topographic and meteorological conditions make the Columbia River Gorge a conduit for N-containing pollutants generated by the Portland, Oregon to Vancouver, Washington metropolis to the west and the agriculturally intensive Columbia Basin to the east. Nitrogen pollution contributes to decreased visibility and highly acidic fog events and potentially impacts valuable historic petroglyphs (Fenn et al. 2007, Geiser et al. 2008). Lichen N and measured N in throughfall were correlated throughout the Columbia River Gorge, and elevated levels were measured, especially at eastern- and western-most sites. Lichens indicating N-enriched environments were more abundant and lichen N concentrations were two to three times higher in the Columbia River Gorge than surrounding national forests; throughfall N deposition ranged from 11.5 to 25.4 kg ha⁻¹ over 4.5 months (Fenn et al. 2007). Lichen N concentrations exceeded regional background ranges.

8.3.2 Alpine Terrestrial

Alpine ecosystems are particularly sensitive to increased availability of N due to inherently low rates of N cycling, low rates of primary production, and thin, poorly weathered soils (Fenn et al. 1998). Most of the studies examining the responses of alpine ecosystems come from the Colorado Front Range of the Rocky Mountains. Baron et al. (1994) used the CENTURY model to estimate N uptake by plants and soils. They estimated that increases in the export of NO₃⁻ began at lower inputs (3 to 4 kg ha⁻¹ yr⁻¹) in the alpine than adjacent subalpine ecosystems, which had larger biological sinks for N.

Bowman et al. (2006) used an experimental approach to examine N impacts on alpine vegetation. Noting that previous studies using high N inputs resulted in

modest increases in production and large changes in species composition, they established experimental plots in dry meadow tundra with an ambient control of 6 kg ha⁻¹ yr⁻¹ and experimental N additions of 20, 40, and 60 kg ha⁻¹ yr⁻¹. Diversity increased at all levels of N treatment relative to the control, while changes in net N mineralization, nitrification, and leaching occurred at inputs greater than 20 kg N ha⁻¹ yr⁻¹ (Bowman et al. 2006). Nearly identical results were obtained from a similar study in Rocky Mountain National Park (Bowman and Murgel⁷). Changes in plant species composition associated with N deposition have the potential to enhance rates of N cycling, leading to a nonlinear increase in N leaching and acidification of soils (Bowman and Steltzer 1998).

8.3.3 Alpine Freshwater Lakes and Streams

Because many alpine freshwaters are strongly N-limited, they are highly responsive to even slight changes in nutrient availability (Elser et al. 2009a, b). Nutrient data collected from regional lake surveys and analyzed as part of the Surface Water Chapter (18) of this monograph indicate that 45 percent of Rocky Mountain lakes and 35 percent of Sierra Nevada/Cascade lakes were N limited in 1985 (see Table 18.1). Slight amounts of additional N from deposition or experiments have elicited an increase in algal biomass and changes in the composition of Rocky Mountain and Sierra Nevada lake species assemblages (Bergström and Jansson 2006; Goldman 1988; Interlandi and Kilham 1998; Lafrancois et al. 2003a, 2003b; Michel et al. 2006; Morris and Lewis 1988; Nydick et al. 2004). Lake sediment records, including algal diatoms, organic compounds, and stable isotopes, indicated that the onset of change due to N fertilization from atmospheric deposition occurred in the decade 1950-1960 (Das et al. 2005; Enders et al. 2008; Wolfe et al. 2001, 2003).

Two species of diatom that are favored by high N availability, *Asterionella formosa* and *Fragilaria crotonensis*, now dominate the flora of at least several

⁷Bowman, W.D.; Murgel, J. Unpublished data on file with William Bowman, Professor, Department of Ecology and Evolutionary Biology, University of Colorado, Boulder, CO 80309-0334

alpine and montane Rocky Mountain lakes (Baron et al. 2000, Interlandi and Kilham 1998, Saros et al. 2003, Saros et al. 2005, Wolfe et al. 2001, Wolfe et al. 2003). Other species have shown negative growth responses to increased N; two of these, *Tetracyclus glans* and *Staurosirella pinnata*, were reported by Michel et al. (2006). In studies of remains of diatoms in lake sediment, typical oligotrophic species such as *Aulacoseria perglabra*, *Cyclotella steligera*, and *Achnanthes* spp. declined coincident with the rise in dominance of *A. formosa* and *F. crotonensis* (Wolfe et al. 2001, 2003).

Several studies have noted higher surface water NO_3^- concentrations in the Colorado Front Range compared with other lakes of the Southern Rocky Mountains, especially lakes east of the Continental Divide (Baron et al. 2000, Elser et al. 2009b, Musselman et al. 2004). Another study reported acid neutralizing capacity (ANC) in surface water $< 0 \mu\text{mol/L}$ as a result of acidifying inputs of N deposition, suggesting that current deposition levels are having an observable impact on catchments in the Front Range (Williams and Tonnesson 2000).

In contrast to the elevated NO_3^- leaching observed with relatively low N deposition inputs in the Colorado Front Range, similar N deposition levels (2 to 5 $\text{kg ha}^{-1} \text{yr}^{-1}$) in Sierra Nevada high elevation catchments have not been reported to increase NO_3^- leaching or NO_3^- levels in lakes. Even when atmospheric deposition is low, high elevation watersheds in the Sierra Nevada appear to naturally export low levels of NO_3^- during early snowmelt, particularly in catchments with low amounts of soil cover (Fenn et al. 2003b, Sickman et al. 2001). The greater response of Rocky Mountain catchments to atmospheric deposition may be due to climatic differences. For example, the greater extent of soil freezing in the Rockies likely reduces the N retention capacity of these watersheds (Sickman et al. 2002). However, the combination of a snowmelt NO_3^- pulse, coupled with a short growing season and limited soils and vegetation, results in temporal asynchrony between N availability and N demand, suggesting that high elevation ecosystems in the Sierra Nevada will respond quickly to increased N deposition (Fenn et al. 2003b).

8.4 Critical Load Estimates

8.4.1 Forest

The empirical N critical load for subalpine forests of the Rocky Mountains is $\leq 4 \text{ kg N ha}^{-1} \text{yr}^{-1}$, based on the incipient effects of N saturation (organic horizon and foliar N enrichment, higher potential net N mineralization rates, and NO_3^- leaching; more information about N saturation can be found in Chapter 7) reported by Rueth and Baron (2002) and Baron.⁶ We expect that more productive forest types with deeper soils in the Pacific Northwest, e.g., Douglas-fir, would have higher critical load thresholds. A modeling study conducted for subalpine forests of the Colorado Front Range projected an increase in forest soil N leachate to streams at N deposition values $> 4 \text{ kg ha}^{-1} \text{yr}^{-1}$ (Baron et al. 1994). In mixed conifer forests of the Sierra Nevada, the empirical critical load for incipient NO_3^- leaching is $17 \text{ kg ha}^{-1} \text{yr}^{-1}$ (Fenn et al. 2008). Critical loads are summarized in Table 8.3.

Critical loads for lichens ranged from 1.2 to 7.1 $\text{kg N ha}^{-1} \text{yr}^{-1}$. The western Sierra Nevada critical loads range, 3.1 to 5.2 $\text{kg N ha}^{-1} \text{yr}^{-1}$ (Fenn et al. 2008), is considered reliable. The most protective response threshold was based on N concentrations above the threshold value in the lichen, *Letharia vulpina*, in mixed conifer forests of California (Fenn et al. 2008). At this level of N deposition, community composition of epiphytic macrolichens was already shifting from oligotrophic to more N-tolerant species. The higher threshold marked a shift to domination by eutrophic species. Using the Sierra Nevada (Fenn et al. 2008) response thresholds, a more conservative response threshold for Alaska (justified in Table 4.1), and relevant precipitation ranges, the western Oregon and Washington model (Geiser et al. 2010) predicted critical loads of 2.5 to 7.1 for the Northwestern Forested Mountains ecoregion in the contiguous United States and 1.2 to 3.7 $\text{kg N ha}^{-1} \text{yr}^{-1}$ for Alaska.

Provisional N critical loads using ectomycorrhizal fungal abundance and diversity as indicators ranged from 5 to 10 $\text{kg ha}^{-1} \text{yr}^{-1}$, based on critical loads determined using the same indicators in similar forest types in the marine west coast forests and northern forests ecoregions

Table 8.3—Empirical critical loads of nutrient N for the Northwestern Forested Mountains ecoregion.
Reliability rating: ## reliable; # fairly reliable; (#) expert judgment

Ecosystem component	Critical load for N deposition <i>kg ha⁻¹ yr⁻¹</i>	Reliability	Response	Comments	Study
Alpine lakes	1.5	##	Changes in diatom assemblages	Wet deposition estimate	Baron 2006
Lichens	1.2-3.7	(#)	Community change in mixed-conifer forests in Alaska	Application of western Oregon and Washington model	Geiser et al. 2010
	2.5-7.1	##	Community change in mixed-conifer forests		Fenn et al. 2008, Geiser et al. 2010
Subalpine forest	4	##	Increase in organic horizon N, foliar N, potential net N mineralization, and soil solution N, initial increases in N leaching below the organic layer		Baron et al. 1994, Rueth and Baron 2002
Alpine lakes	4.0	#	Episodic freshwater acidification		Williams and Tonnesson 2000
Alpine vegetation	4-10	##	Plant species composition		Bowman et al. 2006
Ectomycorrhizal fungi	5-10	(#)	Ectomycorrhizal fungi community structure in white, black, and Engelmann spruce forests	Expert judgment extrapolated from Marine West coast spruce and northern spruce-fir forest	Lilleskov 1999; Lilleskov et al. 2001, 2002, 2008
Mixed conifer forest	17	##	NO ₃ ⁻ leaching	Critical load based on SW Sierra Nevada and San Bernardino Mts	Fenn et al. 2008
	17	#	Reduced fine root biomass	Critical load based on SW Sierra Nevada and San Bernardino Mts	Fenn et al. 2008

(Chapters 9 and 7; Lilleskov 1999; Lilleskov et al. 2001, 2002, 2008).

8.4.2 Alpine Terrestrial

Using the rates of changes in species abundances, Bowman et al. (2006) estimated individual plant species responded to inputs at 4 kg ha⁻¹ yr⁻¹, while whole community responses were detectable at 10 kg N ha⁻¹ yr⁻¹. Estimates of critical load based on changes in net N mineralization, nitrification, and NO₃⁻ leaching were above 20 kg N ha⁻¹ yr⁻¹ (Bowman et al. 2006), higher than the 3 to 4 kg N ha⁻¹ yr⁻¹ predicted by Baron et al. (1994) using a modeling approach.

8.4.3 Alpine Freshwater Lakes and Streams

A hindcasting technique suggested the tipping point for change for aquatic diatoms occurred at about 1.5 kg N

ha⁻¹ yr⁻¹ as wet deposition (Baron 2006). A meta-analysis of lakes across the northern hemisphere, including alpine lakes of the western United States, suggests lake productivity increases at deposition values of 2.5 kg N ha⁻¹ yr⁻¹ (Bergström and Jansson 2006). Williams and Tonnesson (2000) compared catchment yields of N with deposition in the Green Lakes Valley, and estimated a N critical load for acidification of 4 kg ha⁻¹ yr⁻¹ wet deposition. Aquatic critical loads for alpine and other lakes are discussed in greater detail in Chapter 18.

8.5 Comparison to European Critical Loads

8.5.1 Forest

Northwestern forests can most easily be compared to the Scandinavian forests, which were the subject of a recent workshop (Nordin 2007). Using empirical results from

N fertilization experiments, a critical load for changes in understory plant species composition of less than 10 kg N ha⁻¹ yr⁻¹ was suggested for forest ecosystems, while for more sensitive parts of the ecosystem, such as bogs and poor minerotrophic mires, a critical load of less than 8 kg ha⁻¹ yr⁻¹ was suggested (Nordin 2007). Bobbink et al. (2003) recommended a critical load of 10 to 15 kg N ha⁻¹ yr⁻¹ for forests based on increased N mineralization and nitrification.

8.5.2 Alpine Terrestrial

The synthesis by Bobbink et al. (2003) included estimates of critical loads for alpine ecosystems based on vegetation responses in fertilization experiments. For ecosystems on soils derived from granitic parent material, an estimated critical load of 10 kg N ha⁻¹ yr⁻¹ was reported, while for soils on basic parent material, a critical load of 15 to 20 kg N ha⁻¹ yr⁻¹ was estimated. These estimates are 2 to 5 times higher than those estimated for alpine sites in the southern Rockies discussed above.

8.6 Future Research Directions and Gaps in Data

Improved methods for quantifying N deposition and increased efforts to monitor trends over time are critically important for the Northwestern Forested Mountains ecoregion (Fenn et al. 2009), especially considering the projected increase in atmospheric N deposition to the region. Given the highly variable terrain, orographic effects, difficulties of measuring winter deposition in alpine sites, potentially high inputs of dry deposition (particularly to coniferous stands), and isolated or concentrated emission source areas, deposition rates are often underestimated. As a result, it is difficult to extrapolate deposition rates, even to areas that are physically close to each other but different in climate, elevation, or vegetation. Using wet-only estimates for deposition, and the problems associated with underestimating current deposition rates, may result in setting critical loads too low. In contrast, empirical critical loads are often based on linking observed responses with shorter-term N additions and the time lag in ecological responses to N deposition could lead us to set critical loads too high. The results

of long-term fertilization studies would improve understanding of the rate of change possible in these high elevation ecosystems. Effects of N deposition on plant biodiversity and mycorrhizae have not been studied over most of the Northwestern Forested Mountains ecoregion. Plant biodiversity often responds to low to moderate levels of chronic N deposition. Such sensitive indicators will be most valuable, considering the low to moderate N deposition levels that occur over most of this region.

Far more is known for the more heavily impacted Colorado Front Range than for other parts of the Rocky Mountains. Additional studies are needed to better understand the sensitivity of alpine ecosystems in other parts of the west, particularly in the Teton and Wind River Ranges, which may have elevated N deposition due to agricultural activities and oil and gas extraction. Existing plant and lichen data collected systematically by the U.S. Forest Service Forest Inventory and Analysis program in California, Oregon, Washington, Idaho, Montana, Wyoming, Utah, Colorado, and New Mexico can potentially provide further insights into regional critical loads if community composition responses can be correlated with N deposition estimates.

Fire is an essential component of western forests, and many stands have suffered severe dieback due to bark beetle irruptions. More research is needed on the interactions between fire, bark beetle attack, and atmospheric deposition (Grulke et al. 2009). For freshwaters, surveys and experiments are needed to determine the effects of N deposition on algal biodiversity in N-limited lakes. No studies to date have adequately addressed this question. Finally, climate change will affect every aspect of the N cycle, from emissions and agricultural practices to climate-mediated ecological and biogeochemical N response rates. Modeling and empirical studies should be used to address whether the already low estimates of critical loads for the Northwestern Forested Mountains ecoregion will need to be adjusted to changes in temperature and precipitation regimes.

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