

9 MARINE WEST COAST FORESTS

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9.1 Ecoregion Description

The Marine West Coast Forests ecoregion encompasses the Pacific Coast from central Alaska and the Aleutian Islands south through northern California (CEC 1997; Fig. 2.2). The ecoregion description is adapted from CEC (1997). Topography is mountainous bordered by coastal plains. Soils range from very nutrient rich to nutrient poor; nitrogen (N) is the nutrient most commonly limiting to plant productivity, although some areas are naturally very rich in N. The maritime influence of the Pacific Ocean results in high precipitation (600 to 5000 mm yr⁻¹), a long growing season, and moderate mean annual temperatures (5 to 9 °C). Within North America, all of the wettest climates, many of the most productive forests, and all temperate rain forests occur in this ecoregion. Many lowland aquatic ecosystems in this region sustain a diversity of anadromous salmonids that can be a significant source of marine-derived N to rivers and lakes, whereas upland aquatic ecosystems are often nutrient poor. Terrestrial ecosystems range from mild, humid coastal rain forest to boreal forests, to cool sub-alpine forests and alpine conditions at higher elevations. Tree species composition of the temperate coastal forests varies by latitude, and from north to south, and includes white spruce (*Picea glauca*), Sitka spruce (*Picea sitchensis*), western hemlock (*Tsuga heterophylla*), Nootka cypress (Alaska yellow cedar; *Callitropsis nootkatensis*), red alder (*Alnus rubra*), western red cedar (*Thuja plicata*), Douglas-fir (*Pseudotsuga menziesii*), and California redwood (*Sequoia sempervirens*). Many of these trees can reach large size and live to great age. In the drier rain-shadow areas, Oregon white oak (Garry oak; *Quercus garryana*) and Pacific madrone (*Arbutus menziesii*) occur with Douglas-fir. Mountain hemlock (*Tsuga mertensiana*) and Pacific silver fir (*Abies amabilis*) dominate subalpine forests; alpine tundra is dominated by shrubs, herbs, mosses, and lichens.

9.2 Ecosystem Responses to N Deposition

Data are sparse on Marine West Coast Forests ecosystem responses to N deposition, with information available on lichen responses in Oregon, Washington, and southeastern Alaska, and for plant, soil and mycorrhizal responses in white spruce dominated ecosystems in south-central Alaska. Two of the best understood lichen responses to N deposition across the Marine West Coast Forests ecoregion are changes in lichen community composition and accumulation of N in lichen thalli. Declines in the growth and photosynthesis of rare aquatic lichens have also been associated with elevated streamwater nitrate (NO₃⁻) concentrations.

Responses to N emissions from an industrial fertilizer facility by coastal white spruce forests in south-central Alaska include declines in tree health (thinning crowns, chlorotic foliage), changes in forest understory composition, foliar nutritional imbalances (elevated nitrogen:magnesium and nitrogen:calcium), elevated NO₃⁻ in forest floor and mineral soil, and declines in ectomycorrhizal fungal diversity and ectomycorrhizal fungal community change (sporocarp and root-tip level; Lilleskov et al. 2001, 2002; Whytemare et al. 1997). A suite of other responses typical of forested regions (see Chapters 7, 8, and 10) are also likely to be important in the Marine West Coast Forests ecoregion, including: increases in plant foliar N, soil N cycling and NO₃⁻ leaching; decreases in soil carbon:nitrogen; changes in species composition; and increased likelihood of pest outbreak (Brozek 1990, Compton et al. 2003, Perakis et al. 2006, Perakis and Sinkhorn in press, Prescott et al. 1993, 1995).

9.3 Range of Responses Observed

9.3.1 Forest Lichens

Lichen community composition and chemistry are currently the best indicators for assessing ecosystem

responses to N deposition across the Marine West Coast Forests ecoregion (Fenn et al. 2003, 2007; Geiser and Neitlich 2007; Glavich and Geiser 2008; Geiser et al. 2010; Jovan 2008); information on lichen research methodology and terminology can be found in Chapter 4 of this report.

A recent analysis by Geiser and Neitlich (2007) of 1470 epiphytic lichen surveys in western Oregon and Washington shows that lichen community composition and thallus N concentrations shift continuously across the regional N deposition gradient of 0.8 to 8 kg N ha⁻¹ yr⁻¹ (Porter 2007). Oligotrophs and mesotrophs, encompassing nearly all of the larger bodied, endemic, and ecologically important species, comprised more than 90 percent of species richness at 865 of the sites (Geiser et al. 2010). Oligotroph detections peak at N deposition ranges up to 2.5 kg N ha⁻¹ yr⁻¹ (McCune and Geiser 2009). As N deposition increases, the relative proportion of mesotrophic species increases, peaking at deposition rates between 2.5 and 4.0 kg N ha⁻¹ yr⁻¹. At deposition rates of 4.5 kg N ha⁻¹ yr⁻¹, smaller, eutrophic species dominate and oligotrophs and mesotrophs decline to less than 12 percent of species. Oligotrophs play important ecological roles as forage, nesting material, habitat for mollusks and invertebrates, and contribute to N₂-fixation, nutrient cycling, and moderation of humidity; these roles are not filled by eutrophs (McCune and Geiser 2009). Therefore, shifts in lichen community composition, particularly in old-growth forests where lichen biomass is highest, can have broader effects on forest ecosystems and food webs. Responses to N inputs are summarized in Table 9.1.

Lichen community-based air quality scores have been linearly correlated with N deposition estimates from Interagency Monitoring of Protected Visual Environments (IMPROVE), National Atmospheric Deposition Program (NADP), and Community Multiscale Air Quality (CMAQ; see Chapter 4 for descriptions). Geiser and Neitlich (2007) used a nonmetric multi-dimensional scaling ordination technique to separate climate from pollution effects on lichen communities and score air quality at each of the western Oregon and Washington survey sites. In this analysis, air score increased as air quality worsened. Air

scores were directly and positively correlated with N concentrations in ammonium ion (NH₄⁺) wet deposition at eight NADP sites (mg L⁻¹ yr⁻¹; r² = 0.55, p < 0.0001; Geiser and Neitlich 2007), and with ammonium sulfate ((NH₄)₂SO₄) and ammonium nitrate (NH₄NO₃) concentrations in fine particulates at 12 IMPROVE sites in the study area (µg m⁻³ yr⁻¹; r² = 0.93, p < 0.0001; Geiser et al. 2010). After accounting for precipitation, lichen air scores were also positively correlated with total kg N ha⁻¹ yr⁻¹ in wet deposition measured by NADP (r² = 0.64, p = 0.006) and modeled by CMAQ (r² = 0.23, p < 0.005). CMAQ deposition estimates were matched to each of 1470 air scores by overlaying lichen survey coordinates on 36 x 36 km CMAQ grid cell modeled output. When dry deposition was included to estimate total deposition, still accounting for precipitation volume, the strength of correlations between air scores and CMAQ output improved (r² = 0.35, p < 0.005).⁸ Responses of lichens to N concentrations are summarized in Table 9.2.

Field exposure of the air-pollution tolerant epiphytic lichens *Parmelia sulcata* and *Platismatia glauca* to 2 kg wet N ha⁻¹ yr⁻¹ in the western Oregon and Washington Cascades resulted in mainly beneficial effects (shifts in carbon balance between cellular compartments, stimulation of carbon uptake, and improved protection from chlorophyll degradation), though some sensitive lichens were sparse at these sites (Ra et al. 2004, 2005). This deposition rate, while relatively low in comparison to polluted areas, is more than double the estimated background deposition of 0.8 kg N ha⁻¹ yr⁻¹ for Oregon and Washington (Porter 2007).

The sensitive and conspicuous epiphytic macrolichen, Menzies' cartilage (*Ramalina menziesii*), is distributed throughout the Coast Ranges and inland valleys from southern California to southeastern Alaska. Fumigation with nitric acid (HNO₃) at levels from 7 to 25 µg

⁸The 36 x 36 km resolution for deposition estimates is low. Point measurements, especially in grid cells that include urban areas or steep deposition gradients, can be very different than the grid cell mean. Therefore correlations of 0.35 seem reasonable, and, because of statistical power generated by 1470 measurements, the equation for the linear relationship is expected to be reliable. Uncertainty lies primarily in the accuracy of the model output.

Table 9.1—Responses to N input for Marine West Coast Forests ecoregion.

Ecosystem	Site	N input $kg N ha^{-1} yr^{-1}$	Indicator	Response	Study
Coniferous forest	Western OR; WA	~1-8	Lichens	Defined regional indicator oligotroph, mesotroph, and eutroph species response to N deposition range	Jovan 2008
Coniferous forest	Western OR; WA	<1.5->4.5	Lichens	Defined regional indicator oligotroph, mesotroph, and eutroph species response to N deposition range	McCune and Geiser 2009
Coniferous forest	Western OR; WA	2.7-9.2 CMAQ 0-7.0 NADP (wet) 0.4-4.4 CMAQ (wet)	Lichens	Regional scale lichen community composition shifts from oligotroph dominance to mesotroph and eutroph dominance with increasing lichen thallus N; N input increases with precipitation	Geiser et al. 2010
Coniferous forest	Western OR; WA Cascades	2 (wet)	Lichens	N tolerant lichens <i>Parmelia sulcata</i> and <i>Platismatia glauca</i> from moderately polluted sites had increased chlorophyll and lipid content, cytoplasmic lipid droplets, and algal cell wall thickness, indicating altered storage allocation among cellular compartments	Ra et al. 2004
Coniferous forest	Western OR; WA Cascades	2 (wet)	Lichens	Positive physiological effect on tolerant lichens but some sensitive lichens were sparse. <i>Platismatia glauca</i> exposed to moderate levels of fertilizing air pollutants had increased N, S, K, Na; increased chlorophyll and carotenoids; increased OD435/415 ratios; and, in summer, increased CO ₂ uptake and decreased thallus density	Ra et al. 2005
Coniferous forest	S. Central Alaska	5	Tree	Crown thinning, chlorotic foliage	Lilleskov et al. 2001, 2002; Whytemare et al. 1997
Coniferous forest	S. Central Alaska	5	Tree foliar chemistry	Elevated N:Mg and N:Ca ratios	Lilleskov et al. 2002, Whytemare et al. 1997
Coniferous forest	S. Central Alaska	5	Understory and ground cover	Compositional shifts	Lilleskov et al. 2001, Whytemare et al. 1997
Coniferous forest	S. Central Alaska	5	Ectomycorrhizal fungi	Loss of diversity	Lilleskov et al. 2001, 2002
Coniferous forest	S. Central Alaska	5	Ectomycorrhizal fungi	Compositional shifts towards nitrophilic taxa	Lilleskov et al. 2001, 2002
Coniferous forest	S. Central Alaska	5	Soils	Elevated available NO ₃ ⁻	Lilleskov et al. 2001, 2002; Whytemare et al. 1997

Table 9.2—Range of responses to N concentrations for lichen in Marine West Coast Forests ecoregion.

Ecosystem component	Site	N input	Indicator	Response	Study
Coniferous forest	Western OR; WA	>0.06 mg wet NH ₄ ⁺ L ⁻¹ from NADP	Lichens	Declines in sensitive and increases in tolerant and eutrophic epiphytic lichens, increases in lichen thallus N. Lichen N thresholds for <i>Platismatia glauca</i> (0.59% N dry weight)	Geiser and Neitlich 2007
Mountain streams	Coast, Cascade, and Siskiyou Ranges of w OR WA and n CA	Up to 0.1 to 0.33 mg L ⁻¹ during summer field visits	Lichens	Presence of the aquatic lichens <i>Peltigera hydrotheria</i> and <i>Leptogium rivale</i>	Glavich 2009
Mountain streams	Western Cascades, OR	6.3-12.6 mg L ⁻¹ annual mean as NO ₃ ⁻	Lichens	Declines in rate of photosynthesis of <i>Peltigera hydrotheria</i> after 1 year exposure	Davis et al. 2000
Coniferous forest	Western OR and WA	0.51 µg m ⁻³ or 0.26-0.33 kg N ha ⁻¹ yr ⁻¹ in fine particulate N from (NH ₄) ₂ SO ₄ and NH ₄ NO ₃ (IMPROVE)	Lichens	Shifts in lichen community composition dominance from oligotrophs to eutrophs	Geiser et al. 2010; Glavich and Geiser 2008
Deciduous forest	CA Coast Range	7-25 µg HNO ₃ m ⁻³ fumigation	Lichens	Increased membrane permeability, decomposition of photosynthetic pigments, and decreased carbon exchange capacity in <i>Ramalina menziesii</i>	Riddell et al. 2008

m⁻³ (equivalent to concentrations observed in the Los Angeles basin) were unequivocally damaging and associated with increased membrane permeability, decomposition of photosynthetic pigments, and decreased carbon exchange capacity (Riddell et al. 2008).

In southeastern Alaska, response thresholds for N concentration in the epiphytic macrolichens witch's hair (*Alectoria sarmentosa*, 0.56 percent), tube lichen (*Hypogymnia enteromorpha*, 0.75 to 0.89 percent), inactive tube lichen (*Hypogymnia inactiva*, 0.76 percent), and ragged lichen (*Platismatia glauca*, 0.59 to 0.80 percent) have been used to detect hotspots of N deposition in forests near cruise ship docks in Juneau and Skagway's Klondike Goldrush National Historic Park, near popular anchorages for marine fishing vessels at Warren and Coronation Island Wildernesses, and at Greens Creek Gold mine (Dillman et al. 2007, Furbish et al. 2000). Increasing N concentrations in lichens during the past 15 years were attributed to additional N deposition from transpacific sources and increased regional marine traffic (Dillman et al. 2007).

9.3.2 Other Forest Responses

General information on the response of the Marine West Coast Forests to N deposition, beyond information derived from lichens, is relatively sparse. Adverse effects of excess N resulting from industrial NH₃ emissions have been observed in plants, soils, and mycorrhizae along a gradient of N deposition (range: 0.7 to 21.0 kg N ha⁻¹ yr⁻¹) in coastal white spruce forests of south-central Alaska. Although estimates of N deposition along this gradient are not well constrained, and likely varied over ~30 years of industrial emissions, strong gradients in forest response to N were observed. Bulk N deposition of approximately 5 kg ha⁻¹ yr⁻¹ at the time of sampling was associated with declines in tree health (thinning crowns, chlorotic foliage), changes in forest understory composition, foliar nutritional imbalances (elevated N:Mg and N:Ca), elevated NO₃⁻ in forest floor and mineral soil, and declines in ectomycorrhizal fungal diversity and ectomycorrhizal fungal community change (sporocarp and root-tip level; Lilleskov et al. 2001, 2002; Whytemare et al. 1997). Most changes were correlated with increased soil NO₃⁻

availability, which appears to be a useful indicator of long-term response to N inputs in this ecosystem.

Other pathways of N input to forests of the Marine West Coast Forests ecoregion provide insight into forest responses to added N. High rates of biological N₂-fixation by alder and ceanothus (*Ceanothus* spp.) are a distinguishing feature of many forested ecosystems in the region, and commercial forest fertilization with N is also widely practiced. While such N inputs differ from atmospheric deposition in their mode, timing, and rates, they can nevertheless be useful for developing a qualitative understanding of the impacts of elevated N. Forest N fertilization usually occurs as a large dose of urea exceeding 200 kg N ha⁻¹, typically applied once or twice over 40 to 80 years of plantation growth (Stegemoeller et al. 1990). Pure stands of red alder can sustain N₂-fixation rates of 100 to 300 kg N ha⁻¹ yr⁻¹ over 50 to 75 years of growth, adding fixed N to ecosystems primarily in organic forms via above- and belowground tissue turnover, as well as some leakage of NH₄⁺ from nodules (Binkley et al. 1994). Many of the observed effects of N addition by fertilization and biological fixation qualitatively resemble those associated with atmospheric N deposition, including: elevated tissue N content (Brozek 1990), increased nitrification and NO₃⁻ leaching to soils and streams (Compton et al. 2003), loss of divalent base cations from soil and increased soil acidification (Van Miegroet and Cole 1984), and changes in forest understory composition (Prescott et al. 1993, 1995). However, it remains difficult to assess how high rates of N input from biological N₂-fixation and N fertilization translate quantitatively into response thresholds under chronic N deposition.

Responses to atmospheric N inputs in Marine West Coast Forests are likely to be shaped by wide variation in N status across the region. Site N status in the region ranges from ultra N-poor soils developed on young volcanic ash and glacial outwash to extremely N-rich soils that escaped the last glaciation and which rank among the most N-rich areas on Earth. Sites with high N status, particularly in coastal regions, often show negative or no response to N fertilization (Hopmans and Chappell 1994, Peterson and Hazard 1990).

Responses to N fertilization can also vary depending on the dominant tree species considered, for example, with Douglas-fir being more N responsive than western hemlock (Edmonds and Chappell 1993). Variation in N status influences ecosystem response to N₂-fixation and forest fertilization (Binkley et al. 1992, Edmonds and Hsiang 1987), and is therefore likely to shape response to atmospheric N deposition as well. Previous site occupancy by woody N₂-fixers, fire history, and climate all contribute to variations in site N status across the region. Legacy effects of N₂-fixing red alder are particularly significant and can result in addition of >10,000 kg N ha⁻¹ during the 75-year lifespan of a pure stand of red alder. In the moist Oregon Coast Range, historic N₂-fixation by red alder is a proximate cause of exceptionally high soil N accumulation (~30 Mg ha⁻¹), and ultimately yields coniferous forests that exhibit elevated nitrification (80 kg N ha⁻¹ yr⁻¹), NO₃⁻ leaching (25 kg N ha⁻¹ yr⁻¹), soil acidification, base cation depletion, and aluminum (Al) mobilization (Perakis et al. 2006, Perakis and Sinkhorn in press). High foliar N also predisposes large areas of Douglas-fir plantation forests to outbreaks of pathogenic Swiss needle cast fungi (*Phaeocryptopus gaeumannii*) (El Hajj et al. 2004). Further N additions to such areas may have the potential to rapidly intensify symptoms of N saturation; the stages of N saturation in forested ecosystems are described in Chapter 7. Disturbances such as fire and forest harvest can remove ~1,000 kg N ha⁻¹ from a site and may ameliorate some effects of excess N, although N losses may be reversed within several decades where alder or ceanothus dominate early succession.

9.3.3 Aquatic Systems

Currently there is limited information on how atmospheric N deposition shapes aquatic ecosystem N status across the Marine West Coast Forests ecoregion. Transpacific transport of pollutants from Asia to western North America is an issue of increasing concern in the region (Jaffe et al. 1999, Wilkening et al. 2000). Short-lived spikes in N deposition have been associated with increased streamwater NO₃⁻ in old-growth forested watersheds of Olympic National Park, raising concentrations from baseline values < 5 µg N L⁻¹ up to 40 µg N L⁻¹, and illustrating the potential sensitivity of

old-growth forests in the region to added N (Edmonds et al. 1998). The infrequent and highly pulsed nature of these inputs, however, makes it difficult to understand long-term ecological effects and develop watershed N input-output relationships.

Variation in the cover of N₂-fixing red alder across watersheds of the Oregon coast range is related positively to streamwater fluxes of NO₃⁻ and major base cations. Assuming an N₂-fixation rate of 150 kg ha⁻¹ yr⁻¹ for pure alder stands (Binkley et al. 1994), approximately 25 to 30 percent of fixed N is lost as streamwater N export (Compton et al. 2003). These losses correspond to maximum flow-weighted streamwater N concentrations of 2.5 mg L⁻¹, mostly (>90 percent) as NO₃⁻. Studies of N₂-fixing red alder forests in Oregon and Washington (Binkley et al. 1992) and of conifer forests in Oregon (Perakis and Sinkhorn in press) further suggest that the susceptibility of sites to NO₃⁻ leaching increases with soil N status and nitrification rates. Collectively, these results suggest a high potential for atmospheric N deposition to increase NO₃⁻ leaching across some N-rich coastal areas of the Marine West Coast Forests ecoregion, and that some streams are already exposed to naturally elevated NO₃⁻. In colder and more inland areas that support less active N₂-fixation, however, streamwater fluxes of NO₃⁻ remain low, with unknown sensitivity to atmospheric N inputs.

A lichen-indicated critical level for stream-water NO₃⁻ would appear to be between 6.3 to 12.6 mg N L⁻¹. The critical level is the concentration above which adverse effects to sensitive vegetation may occur (UBA 2004). This estimate is based on a year-long laboratory study of the growth and photosynthetic responses of the submerged aquatic hydrothyria lichen (*Peltigera hydrothyria*) collected from the Oregon Cascades and exposed to NO₃⁻ concentrations between 1.6 and 94.5 mg N L⁻¹ (Davis et al. 2000). These levels greatly exceed typical summer NO₃⁻ levels (< 0.05 mg L⁻¹) in the native habitat of this lichen; however stream N could be higher during fall flushes. During a 2-year field study of randomly selected watersheds in western Oregon, western Washington, and northern California, the highest summer stream total N concentrations

associated with occurrence of the aquatic skin (*Leptogium rivale*) and hydrothyria lichens were 0.1 and 0.33 mg N L⁻¹, respectively (Glavich 2009). Gilbert et al. (1997) observed a lower biological diversity in aquatic lichen communities in waters of England where total inorganic N concentrations were just above 0.6 mg L⁻¹.

9.4 Critical Loads Estimates

The onset of elevated soil nitrification and NO₃⁻ availability, conifer nutrient imbalance and decline, declines in ectomycorrhizal fungal diversity, and ectomycorrhizal fungal community change are expected at bulk N deposition of approximately 5 kg ha⁻¹ yr⁻¹ in the northern forests of this region (Lilleskov et al. 2001, 2002; Whytemare et al. 1997). This critical load must be considered expert opinion because historic unrecorded N inputs were likely to have been higher than inputs recorded at the time of study. In addition, it is based on only one study system in southeast Alaska that is almost certainly not representative of all forest types, soils, and climates across the region.

Geiser et al. (2010) created regression models relating lichen-based air scores to NADP and CMAQ measures of N deposition, accounting for precipitation. At an air score response threshold of 0.21, oligotrophs comprised as little as 30 percent, and eutrophs up to 34 percent, of species richness (see Table 4.1). The resulting critical loads were 2.7 to 9.2 kg N ha⁻¹ yr⁻¹ for total deposition (and 0.7 to 4.4 kg N ha⁻¹ yr⁻¹ for wet-only deposition). Because hardwood forests growing on valley floors favor eutrophs by providing a more nutrient-rich canopy drip and higher pH bark substrates compared to the dominant coniferous forests, the response threshold was selected to allow for a natural range in N-availability among clean sites. Critical loads increased with precipitation, which varied approximately tenfold across the study area, because high precipitation volumes dilute the impact of N loading on lichens. The 95 percent confidence interval for the 2.7 to 9.2 kg N ha⁻¹ yr⁻¹ critical load range was 0 to 13.5 kg N ha⁻¹ yr⁻¹. The critical load range can be considered reliable because the systematic sampling design yielded high quality, spatially extensive data. However, while the data for

Table 9.3—Empirical critical loads of nutrient N for the Marine West Coast Forests 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
Western OR and WA forests	2.7-9.2	##	Lichen community composition	Critical load increases with regional range in mean annual precipitation from 45-450 cm	Geiser et al. 2010
SE Alaska forests	5	(#)	Fungal community change; declines in ectomycorrhizal fungal diversity		Lilleskov 1999; Lilleskov et al. 2001, 2002; Whytemare et al. 1997

total deposition are also extensive, they are based on a fairly coarse grid (36 km) with some uncertainties regarding accuracy of the modeled data. Critical loads are summarized in Table 9.3.

9.5 Comparison to Critical Loads from Europe and Other Ecoregions

Many vascular plants and cryptogams of the Tundra, Taiga, and Northern Forests ecoregions also occur in the Marine West Coast Forests ecoregion. For example, extensive peatlands in the Alaskan portion of the Marine West Coast Forests ecoregion support mat-forming lichens and bryophytes and ericaceous shrubs (heaths) typical of boreal ecoregions; alpine areas, especially in Alaska, support tundra species. Therefore, references to Eurasian vegetation responses in these three preceding chapters are potentially relevant to parts of the Marine West Coast Forests. The estimated critical load range for Marine West Coast Forests (2.7 to 9.2 kg N ha⁻¹ yr⁻¹) is consistent with all other U.S. lichen-based estimates reported in this document. The upper-end estimate is consistent with European values, and the lower-end estimate is more than threefold greater than background deposition in the region. For mycorrhizal fungi, the results for this region correspond well with those expected for spruce-fir forests for the northeastern United States (5 to 7 kg N ha⁻¹ yr⁻¹) and are lower than those reported for European forests (10 to 20 kg N ha⁻¹ yr⁻¹).

9.6 Future Research Directions and Gaps in Data

There is a need to understand how the response of forest soils, streams, and their biota will be mediated by underlying variations in site N status in the region. Generally, lowland coastal Oregon and Washington forests and streams possess the highest N status in the region, yet may escape significant elevated N inputs in the near-term due to the lack of significant population and industrial activity along upwind coastal areas. Potential increases in N transport from Asia, however, could make these areas susceptible. On the other hand, naturally N-poor areas of the Oregon and Washington Cascades and glaciated montane forests of the Olympic Peninsula may be sensitive to added N originating from anthropogenic sources in the Willamette Valley and Puget Sound. The wide range in forest ages and in natural and anthropogenic disturbances (e.g., fire, logging) across the region is likely to interact with N status in unknown ways to shape ecosystem responses to N deposition. A better understanding of the relative sensitivity of lichens and other ecosystem response indicators to different forms of atmospheric N deposition would allow more accurate apportionment and prediction of the effects from different emissions sources. Analysis of existing Forest Inventory and Analysis and other U.S. Forest Service lichen community data from southeastern and south-central Alaska will test their application to the northern parts of the ecoregion.

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