Empirical and simulated critical loads for nitrogen deposition in California mixed conifer forests

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Critical loads for N deposition effects on lichens, trees and nitrate leaching provide benchmarks for protecting California forests.

Abstract

Empirical critical loads (CL) for N deposition were determined from changes in epiphytic lichen communities, elevated NO3-leaching in stream water, and reduced fine root biomass in ponderosa pine (Pinus ponderosa Dougll. ex Laws.) at sites with varying N deposition. The CL for lichen community impacts of 3.1 kg ha-1 year-1 is expected to protect all components of the forest ecosystem from the adverse effects of N deposition. Much of the western Sierra Nevada is above the lichen-based CL, showing significant changes in lichen indicator groups. The empirical N deposition threshold and that simulated by the DayCent model for enhanced NO3 leaching were 17 kg N ha-1 year-1. DayCent estimated that elevated NO3-leaching in the San Bernardino Mountains began in the late 1950s. Critical values for litter C:N (34.1), ponderosa pine foliar N (1.1%), and N concentrations (1.0%) in the lichen Letharia vulpina ((L.) Hue) are indicative of CL exceedance.

Keywords: Ecosystem protection; Atmospheric deposition; Epiphytic lichens; Bimonitoring; Nitrogen saturation; Harmful effects

1. Introduction

Globally humans have altered the nitrogen (N) cycle to a greater degree than the much more publicized carbon cycle. As nitrogen (N) compounds cascade through the environment, there are many effects on the environment and aquatic and terrestrial ecosystems (Galloway et al., 2003). A major objective of research efforts on the impacts of N deposition is to quantify the N deposition inputs at which unacceptable impacts occur to sensitive ecosystem components. The critical loads (CL) approach has been widely used to determine the threshold deposition inputs above which unacceptable impacts begin to occur. The use of CL promises to be instrumental as we progress towards air quality management and regulation more firmly rooted in ecosystem protection. Critical loads can be determined for acidification or eutrophication effects in both aquatic and terrestrial ecosystems. Nitrogen and sulfur deposition both contribute to soil acidification effects, while eutrophication or N enrichment effects in terrestrial ecosystems are caused by N deposition. In this study we focus on eutrophication responses although considerable soil acidification has also occurred in high deposition areas in the San Bernardino Mountains (Breiner et al., 2007; Wood et al., 2007).

The largest coordinated effort in developing and implementing CL has been the efforts of the United Nations - Economic Commission for Europe, Internati onal Cooperative Programme (UNECE ICP, 2004). These methods have been adopted by researchers in other countries as well. In the United States there are no large coordinated programs to establish

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suitable CL and no regular extensive and intensive monitoring efforts designed to evaluate CL methodologies. However, projects for developing CL are increasing in North America and a wide variety of approaches have been used in both Canada and the US (Baron, 2006; Ouimet et al., 2006; McNulty et al., 2007; Porter and Johnson, 2007; Williams and Tonnessen, 2000).

The principal methods for CL determination include steady state simple mass balance models (SMB), dynamic models, and empirical observations. Empirical methods are based on site specific data showing deposition levels at which "harmful" biological effects are observed (Bobbink et al., 2003). A major limitation of the empirical approach is that results may only be applied with confidence to sites with highly similar biotic and environmental conditions. Empirical CL are important, however, for validating CL values determined indirectly with SMB and dynamic simulation models. Empirical CL are based on realistic conditions under which adverse biological impacts occur. Simulation modeling methods are particularly useful for evaluating the effects of different combinations of deposition levels, temporal dynamics, disturbance regimes, stand characteristics, and other variables on CL.

The SMB model is the most widely applied CL method in the ICP European network (UNECE ICP, 2004). Models such as the SMB are used to map CL at large scales, but must make many assumptions regarding nutrient cycling and soil chemical processes and lack the resolution to provide accurate site-specific CL. The approach is thus best applied over broad regions. Even when the SMB model is parameterized with site specific data, the capacity of the SMB models to provide reliable estimates of the CL for soil acidification or eutrophication has been questioned (Cresser, 2000; Rodriguez-Lade et al., 2007; Skeffington, 1999), and the biological linkages for the calculated CL are even more uncertain in many cases unless verified by empirical or mechanistic data (De Vries et al., 2007; UNECE ICP, 2004). Notwithstanding the widespread use of relatively standardized CL methodologies in Europe and elsewhere, questions still remain regarding the best approach for determining CL that reflect ecologically relevant thresholds of N deposition.

The major objectives of this study are to determine the 'N as a nutrient' CL for key endpoints in mixed conifer forests in California and to compare different empirical and modeling approaches for determining the N as a nutrient CL. Empirical CL were determined for effects on lichen communities and for NO3 leaching in association with previously published results on reduced fine root biomass (Gruulke et al., 1998) from sites spanning N deposition gradients. Epiphytic lichens are among the most sensitive bioindicators of N in forested ecosystems (Blett et al., 2003; Bobbink et al., 2003; Fenn et al., 2003b; Glavich and Geiser, in press). Consequently, the implementation of lichen-derived CL is likely to prevent undesired impacts to the broader forest ecosystem. Protecting the integrity of lichen communities, moreover, ensures preservation of their complex ecological roles in the nutrient cycling, hydrological flux and supply of wildlife forage and nesting materials to the forest (McCune et al., 20(7). Finally, information on CL will be discussed in regard to their implications for resource protection and land management.

2. Materials and methods

2.1. Site locations

Twenty-four mixed conifer forest sites in California exposed to a wide range of N deposition inputs (Fig. 1; Table 1) were included in this study. Dominant overstory species were ponderosa pine (Pinus ponderosa Laws) or Jeffrey pine (P. jeffreyi Grev. and Balf.) and commonly associated species included California black oak (Quercus kelloggi Newh.), white fir (Abies concolor Gord. & Glend.), and incense cedar (Calocedrus decurrens (Torr.) Florin). Fifteen sites were located in the Sierra Nevada Mountains and nine in the San Bernardino Mountains. The primary use of these forests is recreational and protection of habitat and water sources. Little known logging has occurred in the study sites since the early 20th century and except for fires that occurred in the vicinity of some of the San Bernardino Mountain sampling sites in the fall of 2003, the sites have not been affected by fire in the last 70 years. In all instances, we did not collect samples close to burned areas. Soil parent material is weathered or decomposed granitic rock, except at Lassen where the soil is of volcanic origin. The soils are generally sandy loam in texture and percent base saturation ranges from 70 to 100% (Perin et al., 1996) except in the most polluted western San Bernardino Mountain sites where base saturation has decreased in recent years. Nitrogen deposition in canopy throughfall was measured year-round at the Sierra Nevada sites for one or two years from 2000 to 2003 (Fenn et al., 2003a,c; and M.E. Fenn, unpublished data) and in the San Bernardino Mountains sites for 2-5 years from 2000 to 2005 (Fenn and Poth, 2004; Fenn et al., 2003a; and M.E. Fenn, unpublished data). Throughfall fluxes were measured using ion exchange resin throughfall collectors (Fenn and Poth, 2004). The deposition data do not include deposition fluxes during fire episodes.

2.2. Empirical critical load or nitrate leaching and fine root biomass impacts

The empirical threshold for NO3 leaching is based on the relationship between throughfall N deposition and an empirically-derived clean-site threshold value for streamwater NO3. The threshold was derived by comparing temporal profiles of streamwater NO3 concentrations in relatively pristine low N deposition catchments against profiles of catchments receiving varying levels of chronic N deposition inputs. Based on 28 streams from relatively low pollution catchments in southern California (Penn and Poth, 1999; Riggan et al., 1985) we determined that peak NO3 concentrations in runoff rarely exceed 0.2 mg NO3-N L-1 (14.3 μM). This concentration corresponds with those reported for relatively unpolluted streams in Sweden (De Vries et al., 2007). Thus, we selected 14.3 μM as the critical threshold value for identifying catchments that are beginning to exhibit symptoms of N excess or N saturation. The empirical N threshold for NO3 leaching is thus the N deposition input at which this threshold peak NO3 concentration is exceeded.

The N deposition input leading to NO3 leaching was determined by a linear regression equation between throughfall N deposition and streamwater NO3 data collected from 11 sites in the San Bernardino and southern Sierra Nevada Mountains encompassing a range of annual N deposition inputs (5.7–71.1 kg ha−1 year−1). Streamwater samples were collected as grab samples (n = 3 per sampling date) during two surveys done in February and March 2007, when runoff rates were highest. The throughfall sampling site within each of the 11 catchments was located in the mixed conifer zone. Accessible stream sampling sites were in the mixed conifer zone at 7 of the sites (SL, CP, CA, BF, GV, HP, GH; Table 1) or at lower elevation scrub oak or chaparral sites in 4 of the catchments (MH, PI, BR, CP; Table 1). An implied assumption in developing an empirical CL associated with NO3 leaching is that atmospheric deposition is the sole source of elevated runoff NO3. In selecting streamwater sampling sites for CL determinations, catchments disturbed by
forest fires, stand thinning, or other anthropogenic activities that stimulate net nitrification or otherwise release NO) to the system were excluded. Because a CL by definition refers to atmospheric inputs causing a harmful biological effect, we tested the relationship between reduced fine root biomass in ponderosa pine (see Gruelke et al., 1998) and stream water NO\(_3^-\) concentrations or N deposition at sites across the N deposition gradient in the San Bernardino Mountains. This linkage must be considered preliminary because the fine root data are only available from three sites (Gruelke et al., 1998), although the sites span a N deposition range of 7.5-39.3 and 71.1 kg ha\(^{-1}\) year\(^{-1}\). The reduction in fine root biomass is dramatic with increasing air pollution exposure and is strongly correlated with streamwater NO\(_3^-\) \(r^2 = 0.75\) and N deposition \(r^2 = 0.82\); Table 2. Further work at a greater number of sites will be needed to verify the quantitative nature of the response of fine roots to N enrichment. Ozone is also believed to contribute to the decreased C allocation to fine roots (Gruelke et al., 1998) and could be considered as a confounding factor. However, inasmuch as the intent of the empirical CL is to estimate the N deposition level at which harmful effects occur to the ecosystem under realistic field conditions, ozone exposure effects should be included because in California forests ozone and N are co-occurring pollutants.

2.3. Litter C:N as a predictor of the N deposition CL

Litter was sampled under five ponderosa or Jeffrey pine trees at a subset of the study sites (LN, SL, GF, MH, CP, SP, DW, CO). Litter was collected at two diametrically opposed points located at approximately the midpoint of the radius from the stem of a mature tree to the canopy drip line. The entire depth of the forest floor or organic horizon was sampled except for the undecomposed superficial L-layer litter. Litter from the two sampling points per tree was combined into one replicate sample. The samples were analyzed for total C and N by standard combustion methods with a Carbo Erba analyzer (Fein et al., 1996).

2.4. Empirical critical loads for lichen responses

A preliminary CL for lichens of mixed conifer forests in the Sierra Nevada was developed by cross-walking throughfall N measurements with N concentration in *Letharia vulpina* (Fig. 2) and indices describing lichen community composition \(n = 11\). Nitrogen concentration in *L. vulpina* represents an easily measured biological response to N that correlates closely with local deposition. An abundance of *L. vulpina* data from forests in the Pacific Northwest allowed us to compare collections spanning from polluted to remote sites and determine natural ranges for lichen N concentrations. With this information we defined a “clean site” threshold for N enhancement (analogous in concept to the threshold value used for streamwater NO\(_3^-\); see Section 2.2), which we used to anchor the lichen CL. Lichen samples were collected between 1994 and 2004 primarily from forests on the east side of the Cascade Crest. Sampling and laboratory protocols followed Geiser (2004). Raw data are available on-line at http://gis.nacse.org/lichenair. Laboratory and field replicates were averaged for each site visit, yielding a final sample size of
A small number of sites were sampled in both 1994 and 2004; these were treated as independent measurements. A smooth curve was fitted (kernel standard of 0.0883) to a histogram of the distribution of N concentrations in all 535 samples. The fitted distribution showed a major peak (Fig. 3) representing the clean sites, which made up the bulk of samples. Values in the skewed upper portion of the distribution represent the enhanced-N population (i.e., sites affected by local agricultural or urban-industrial emissions). Confirmation that "clean sites" were exposed to relatively low N levels was evident in composition of the epiphytic lichen communities, namely, in the absence of nitrophytic lichens and the presence of air pollution sensitive species (see also distribution maps of sensitive and N indicator lichens in Oregon and Washington presented in Fenn et al., 2003, 2007).

We calculated a clean site threshold value of 1.0% N (Fig. 3). This threshold represents the uppermost limit, or 100% quantile, of the expected range of values for % N in L. vulpina collected from clean sites. Assuming that concentrations of N in lichens from clean sites follow a normal distribution and that most sites were clean, the peak density of the fitted curve, 0.60% N was used to represent the 50% quantile for the clean sites. The skewed upper part of the distribution represents samples from N enhanced sites. The 100% quantile for % N in the clean population was then calculated by doubling the distance between the 0% quantile (0.20% N) and the 50% quantile, i.e., 2 x (0.60% N - 0.20% N) = 1.0% N. Nitrogen concentrations above this threshold are considered enhanced.

### Table 1

<table>
<thead>
<tr>
<th>Site</th>
<th>Coordinates</th>
<th>N deposition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sierra Nevada sites</td>
<td></td>
<td></td>
</tr>
<tr>
<td>(LN) Lascan</td>
<td>40° 19' 19.20&quot;</td>
<td>121° 26' 27.60&quot;</td>
</tr>
<tr>
<td>(WC) White Cloud</td>
<td>39° 18' 59.01&quot;</td>
<td>120° 59' 55.80&quot;</td>
</tr>
<tr>
<td>(BC) Blackwood Canyon</td>
<td>39° 06' 45.00&quot;</td>
<td>120° 10' 60.00&quot;</td>
</tr>
<tr>
<td>(FL) Fallen Leaf</td>
<td>38° 55' 4.00&quot;</td>
<td>120° 01' 40.00&quot;</td>
</tr>
<tr>
<td>(SP) Sly Park</td>
<td>38° 43' 48.00&quot;</td>
<td>120° 32' 60.00&quot;</td>
</tr>
<tr>
<td>(SM) Five Mile</td>
<td>38° 02' 60.00&quot;</td>
<td>120° 17' 60.00&quot;</td>
</tr>
<tr>
<td>(WA) Wawona</td>
<td>37° 31' 48.00&quot;</td>
<td>119° 40' 12.00&quot;</td>
</tr>
<tr>
<td>(SL) Shaver Lake</td>
<td>37° 08' 33.95&quot;</td>
<td>119° 15' 09.10&quot;</td>
</tr>
<tr>
<td>(WO) Wofferton</td>
<td>36° 35' 38.40&quot;</td>
<td>118° 44' 13.20&quot;</td>
</tr>
<tr>
<td>(GP) Giant Forest</td>
<td>36° 34' 00.84&quot;</td>
<td>118° 46' 37.92&quot;</td>
</tr>
<tr>
<td>(HHA) Hell's Half</td>
<td>37° 25' 58.37&quot;</td>
<td>119° 14' 45.07&quot;</td>
</tr>
<tr>
<td>(MH) Mountain Home</td>
<td>36° 13' 23.28&quot;</td>
<td>118° 42' 46.84&quot;</td>
</tr>
<tr>
<td>(BR) Breckinridge</td>
<td>35° 28' 33.15&quot;</td>
<td>118° 37' 34.71&quot;</td>
</tr>
<tr>
<td>(PI) Piutes</td>
<td>35° 29' 31.65&quot;</td>
<td>118° 23' 44.95&quot;</td>
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<tr>
<td>(GR) Greenhome</td>
<td>35° 39' 42.42&quot;</td>
<td>118° 35' 24.40&quot;</td>
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San Bernardino Mountain sites

<table>
<thead>
<tr>
<th>Site</th>
<th>Coordinates</th>
<th>N deposition</th>
</tr>
</thead>
<tbody>
<tr>
<td>(HV) Holcomb Valley</td>
<td>34° 17' 35.27&quot;</td>
<td>116° 54' 43.15&quot;</td>
</tr>
<tr>
<td>(CO) Camp Osceola</td>
<td>34° 09' 43.54&quot;</td>
<td>116° 51' 26.77&quot;</td>
</tr>
<tr>
<td>(GV) Green Valley</td>
<td>34° 15' 04.11&quot;</td>
<td>117° 03' 02.36&quot;</td>
</tr>
<tr>
<td>(BP) Barton Flats</td>
<td>34° 09' 33.48&quot;</td>
<td>116° 52' 49.15&quot;</td>
</tr>
<tr>
<td>(CA) Camp Angelus</td>
<td>34° 08' 43.87&quot;</td>
<td>116° 58' 40.75&quot;</td>
</tr>
<tr>
<td>(DW) Dogwood</td>
<td>34° 14' 14.62&quot;</td>
<td>117° 12' 18.40&quot;</td>
</tr>
<tr>
<td>(HP) Heups Peak</td>
<td>34° 13' 53.74&quot;</td>
<td>117° 08' 09.19&quot;</td>
</tr>
<tr>
<td>(SP) Strawberry Peak</td>
<td>34° 13' 49.63&quot;</td>
<td>117° 14' 18.31&quot;</td>
</tr>
<tr>
<td>(CP) Camp Paivika</td>
<td>34° 14' 23.95&quot;</td>
<td>117° 19' 34.22&quot;</td>
</tr>
</tbody>
</table>

### Table 2

<table>
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<tr>
<th>Parameters</th>
<th>Regression equation</th>
<th>Number of sites sampled</th>
</tr>
</thead>
<tbody>
<tr>
<td>Throughfall N deposition (γ)</td>
<td>y = 8.69 + 0.58x</td>
<td>0.98 11</td>
</tr>
<tr>
<td>vs. streamwater NO3 ( x )</td>
<td>y = 4.07 + 0.07x</td>
<td>0.72 8</td>
</tr>
<tr>
<td>Bulk N deposition (γ)</td>
<td>y = 148.08 - 3.81x</td>
<td>0.79 8</td>
</tr>
<tr>
<td>vs. streamwater NO3 ( x )</td>
<td>y = 35.65 - 0.11x</td>
<td>0.74 6</td>
</tr>
<tr>
<td>Litter C:N (γ)</td>
<td>y = 35.65 - 0.11x</td>
<td>0.74 6</td>
</tr>
<tr>
<td>vs. streamwater NO3 ( x )</td>
<td>y = 0.81 - 0.012x</td>
<td>0.82 3</td>
</tr>
<tr>
<td>Fine root biomass (γ)</td>
<td>y = 0.68 - 0.0068x</td>
<td>0.75 3</td>
</tr>
<tr>
<td>vs. streamwater NO3 ( x )</td>
<td>y = 0.68 - 0.0068x</td>
<td>0.75 3</td>
</tr>
</tbody>
</table>

* Units for throughfall & bulk N deposition are kg ha⁻¹ year⁻¹. Streamwater NO3 concentrations are as μg L⁻¹, and fine root biomass are g per soil pit (20 × 20 cm; only data from the top 20 cm are included here) (Gruelke et al., 1998).
4.96 epiphytic macrolichen found within a 0.38 ha plot (USDA FS, 2005). Each collected on a systematic grid of permanent plots. The FIA data consist of 2 h
(2) the proportion of total lichen abundance in neutrophytes; (3) the proportion (acidophyte index minus neutrophyte indices). These same indices have been used in related studies to assess N effects on lichen communities in the western US (Jovan, 2008; Jovan and McCune, 2006). Relationships between measurements of throughfall N (\(\text{NH}_4^+ + \text{NO}_3^-\)), calculated indicator indices, and N concentration in L. vulpina were examined with simple linear least squares regression. Correlation strength was evaluated with coefficients of determination.

2.4.2. Data integration

Canopy throughfall measurement points (\(n = 11\)) were matched with nearby topographically similar sites where L. vulpina was analyzed for N in a previous pollution bioindication study (Jovan andCarlberg, 2007) and where epiphytic lichen communities were surveyed by the FIA between 1998 and 2003 (Jovan andMcCune, 2006). Geographic distance from throughfall measurement points to sites where one or both of the lichen variables were measured ranged from < 1 to 23 km (average 10 km). In three cases, L. vulpina and lichen community data were co-located. Although distance between the lichen plots and the throughfall plots were on average 10 km apart, on the scale of the Sierra Nevada Mountains, N deposition is expected to be highly similar between the paired plots within a given catchment. Lichen community data used in this analysis were restricted to plots dominated by conifers (> 85% basal area) to decrease effects of stand composition on community composition.

2.5. N as a nutrient CL determined from the simple mass balance steady state model

Theoretical critical loads were calculated using the SMB model to compare with the empirical CL determined for streamwater NO\(_3^-\). According to the Critical Load Mapping Manual (MM; UNECE ICP, 2004) the following equation was used:

\[
\text{CL}_{\text{MM}}(N) = N_i + N_{\text{net}}(1 - f_{\text{de}}),
\]

where \(N_i\) is the critical load of nutrient \(N, \text{Ni}\) is the long-term immobilization of \(N\) in soil organic matter, \(N_{\text{net}}\) is the net removal of \(N\) in harvested vegetation and animals, \(N_{\text{de}}\) is the acceptable leaching of \(N\) and \(f_{\text{de}}\) is the denitrification fraction. The \(N_{\text{de}}\) term was calculated as the product of the precipitation surplus (\(Q\), \(\text{m}^3\ \text{ha}^{-1}\ \text{yr}^{-1}\)) and the acceptable \(N\) concentration (\(\text{IN}_{\text{acc}}\), \(\text{eq m}^{-3}\)). Following MM prescriptions, \(N_i\) and \(\text{IN}_{\text{acc}}\) were set to 1 kg \(\text{N ha}^{-1}\ \text{yr}^{-1}\) and 0.2 mg N L\(^{-1}\), respectively, although calculations were also performed using 3.5 kg \(\text{N}\) ha\(^{-1}\) year\(^{-1}\) (Downing et al., 1993) and 3 mg N L\(^{-1}\) (De Vries et al., 2007), respectively (Table 4) because these are considered to be more realistic values. To avoid negative \(Q\) values (precipitation surplus) that are often observed in Mediterranean systems, this term was calculated from the long term average flow volume of a perennial stream (USGS Monitoring Station no. 11063680) draining the Devil Canyon/Camp Paivika watershed (\(-1600\ \text{m}^3\ \text{ha}^{-1}\ \text{year}^{-1}\); USGS, 2007). Similarly, \(N_i\) was considered to be 0 as recommended by the MM for long-term net uptake of unmanaged forests. Finally \(f_{\text{de}}\) was considered to be 0.1, as corresponds to sandy loam soils (Downing et al., 1993).

2.6. DayCent biogeochemical model simulations

Critical loads were also calculated using the DayCent biogeochemistry simulation model, DayCent version 4.5 was used in this study to simulate historical N saturation symptoms such as N trace gas emissions from soil and NO\(_3^-\) leaching and to estimate the N deposition threshold for such losses. DayCent (Del Grosso et al., 2008; Parton et al., 1998, 2001) is the daily time step version of the CENTURY model (Parton et al., 1993). It simulates the biogeochemical processes of carbon, nitrogen, phosphorus, and sulfur associated with soil organic matter (SOM) in multiple ecosystem types. One feature of DayCent important to this study is the recent improvement of N cycling algorithms, including the simulation of \(\text{N}_2\), NO, and \(\text{N}_2\) emissions resulting from nitrification and denitrification. Other improvements relate to

\[
PAA_i = \frac{\sum_{j=1}^{s} a_{ij} l_j}{\sum_{j=1}^{s} l_j}
\]

where \(s\) is the number of species in plot \(i\), \(a_{ij}\) is the abundance of species \(j\) in plot \(i\), and \(l_j\) indicates whether each species is an acidophyte \((n = 1)\) or not \((n = 0)\). By changing \(l_j\) the same equation is applied for the neutrophyte and nitrophyte indices. Four plot-level indices were calculated using the lichen community data based on the proportion of lichen abundance in a particular N indicator group: (1) the proportion of total lichen abundance in acidophytes \((\text{PAA}\) in Eq. (1)); (2) the proportion of total lichen abundance in neutrophytes; (3) the proportion of total lichen abundance in nitrophytes; and (4) acidophyte dominance (acidophyte index minus neutrophyte index).

Fig. 3. Histogram of % N in Letharia vulpina, dry weight, sampled from urban, rural, and remote Oregon and Washington forests east of the Cascade Crest between 1994 and 2004 (N = 535). Two populations, clean and N-enhanced, were apparent when we fit a smoothed distribution (blue line) to the histogram. Assuming that concentrations of N in lichens from clean sites follow a normal distribution and that most sites were clean, the peak density of the fitted curve, 0.60% N was used to represent the 50% quantile for the clean sites. The skewed upper part of the distribution represents samples from N enhanced sites. The 100% quantile for % N in the clean population was then calculated by doubling the distance between the 0% quantile (0.20% N) and the 50% quantile, i.e. 2 \(\times\) (0.60% N - 0.20% N) = 1.0% N. The 100% quantile, 1.0% N, was selected as the upper limit for the expected background range, or 'clean site threshold.' Nitrogen concentrations above this threshold are considered enhanced.
detailed soil N mineralization-immobilization processes associated with various organic matter pools (including microbial), which are distinguished by their decomposition rates. Required inputs to the model include daily maximum/minimum temperature and precipitation, site-specific soil properties and hydraulic characteristics, and current and historical land use. Model runs were initialized and stabilized by repeatedly using site-specific weather data for 900 years (i.e., initially started from 1000 AD), background N deposition (0.1 kg N ha\(^{-1}\) year\(^{-1}\)) and fire occurrence every 100 years. For the final model runs Daycent was set for the stand to regenerate after fire in 1901 and was run until 2060 without further fire occurrence, thus simulating the current conditions of fire suppression. The model provides approximations of historical tree growth, litter C, SOM and C/N ratios in the study area (Allen et al., 2007; Fenn et al., 2005; Grulke and Balduman, 1999; Grulke et al., 2001).

Some DayCent parameters, mostly related to tree growth, N adsorption by litter, and decomposition of litter and the soil organic matter component referred to as the chemically-resistant fraction, were modified in order to achieve reasonably well-fitted outputs with observations of tree biomass and allocation, litter and soil organic matter accumulation, and C/N ratios.

The model was parameterized for a high (CP) and low (BF) deposition site with gradually increasing N deposition beginning in 1930 to the current throughfall N deposition levels (70 and 8.8 kg N ha\(^{-1}\) year\(^{-1}\) at CP and BF, respectively). Inputs for the estimated historical rate of increase in N deposition since 1930 are shown in Table 5. Inasmuch as historical N deposition data are not available for the study sites, we estimated historical inputs and temporal patterns of deposition based on intermittent measurements at CP and BF (Fenn and Bytnerowicz, 1993, 1997; Fenn and Poth, 2004) were also used as guidelines in estimating historical deposition. Also taken into account were historical population growth trends and vehicle miles driven in the South Coast Air Basin and in Riverside and San Bernardino Counties (Cox et al., 2006). California Air Resources Board data on NO\(_x\) emissions since 1975 were also used for evaluating trends in NO\(_x\) deposition (Cox et al., 2006). Ozone concentration data at Crestline (immediately adjacent to CP) since 1963 (Bytnerowicz et al., 2008) were used as proxy for estimating temporal trends in NO\(_x\) deposition at CP. Deposition of NH\(_3\) was based on periodic measurements of atmospheric NH\(_3\) taken from 1988 to 2005 at CP and BF (Bytnerowicz et al., 1999; Bytnerowicz et al., 2000; Fenn and Bytnerowicz, 1993, 1997; Fenn and Poth, 2004) were also used as guidelines in estimating historical deposition. Also taken into account were historical population growth trends and vehicle miles driven in the South Coast Air Basin and in Riverside and San Bernardino Counties (Cox et al., 2006). California Air Resources Board data on NO\(_x\) emissions since 1975 were also used for evaluating trends in NO\(_x\) deposition (Cox et al., 2006). Ozone concentration data at Crestline (immediately adjacent to CP) since 1963 (Bytnerowicz et al., 2008) were used as proxy for estimating temporal trends in NO\(_x\) deposition at CP. Deposition of NH\(_3\) was based on periodic measurements of atmospheric NH\(_3\) taken from 1988 to 2005 at CP and BF (Bytnerowicz et al., 1999; Bytnerowicz et al., 2000; Fenn and Bytnerowicz, 1993, 1997; Fenn and Poth, 2004) were also used as guidelines in estimating historical deposition. Also taken into account were historical population growth trends and vehicle miles driven in the South Coast Air Basin and in Riverside and San Bernardino Counties (Cox et al., 2006). California Air Resources Board data on NO\(_x\) emissions since 1975 were also used for evaluating trends in NO\(_x\) deposition (Cox et al., 2006). Ozone concentration data at Crestline (immediately adjacent to CP) since 1963 (Bytnerowicz et al., 2008) were used as proxy for estimating temporal trends in NO\(_x\) deposition at CP.

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For model runs to estimate the CL for NO\textsubscript{3} leaching, deposition was increased from a background level of 0.1 kg N ha\textsuperscript{-1} year\textsuperscript{-1} to 3 kg N ha\textsuperscript{-1} year\textsuperscript{-1} in 1930 and increased linearly to 10 kg N ha\textsuperscript{-1} year\textsuperscript{-1} by 1940. In 1940, N deposition was set at 19 different levels for each site ranging from 10 to 80 kg N ha\textsuperscript{-1} year\textsuperscript{-1} and maintained at these levels for simulations as long as 120 years in order to simulate dose responses to increasing N deposition. For model simulations using this set range of inputs and for model runs using historical N deposition, annual N deposition was set to vary randomly by \(<10\%\). Simulations were also run with 40\% reduced precipitation at CP. The effect of a 117\% increase in precipitation was tested at BF by using CP precipitation levels at BF (annual average of 105.9 cm at CP vs. 48.7 cm at BF). The increased precipitation scenario was tested at BF to determine how much of the lower N loss patterns at BF compared to CP was due to the much lower precipitation at BF. Based on simulated occurrences of peak stream water NO\textsubscript{3} concentrations above the clean site threshold of 0.2 mg L\textsuperscript{-1} (see Section 2.2), we determined the beginning year and conditions under which CL exceedance began. We included a stipulation that the critical NO\textsubscript{3} leachate concentration must be exceeded at least once every 10 years on average to consider the threshold exceeded. Spurious NO\textsubscript{3} exceedances that occur rarely due to extreme climatic events for example, do not indicate a catchment above the CL by our definition.

3. Results

3.1. Empirical critical load for nitrate leaching and fine root biomass

The gradient in throughfall N deposition at the study sites ranged from a relatively pristine site in the northern Sierra Nevada (1.4 kg N ha\textsuperscript{-1} year\textsuperscript{-1}) to heavily polluted sites in the San Bernardino National Forest in southern California with deposition rates as high as 33-71 kg N ha\textsuperscript{-1} year\textsuperscript{-1} (Table 1). Peak NO\textsubscript{3} concentrations at 11 of these sites also varied, ranging from 0.07 to 107 ueq L\textsuperscript{-1}. Based on the linear regression ($r^2 = 0.98$) of stream water NO\textsubscript{3} vs. throughfall N deposition (Table 2), the N deposition at which the critical threshold NO\textsubscript{3} concentration (14.3 ueq L\textsuperscript{-1}) occurs is 17 kg N ha\textsuperscript{-1} year\textsuperscript{-1}, which can be considered as the preliminary empirical threshold for NO\textsubscript{3} leaching at these sites (95\% confidence interval: 15.19 kg N ha\textsuperscript{-1} year\textsuperscript{-1}). It should be noted that the unusually high correlation coefficient is possibly due to the occurrence of nine sites with N deposition somewhat clustered at less than 20 kg ha\textsuperscript{-1} year\textsuperscript{-1} and two sites with deposition much greater than 20 (36 and 71 kg ha\textsuperscript{-1} year\textsuperscript{-1}).

A similar relationship was evaluated for bulk deposition and streamwater NO\textsubscript{3} concentration ($r^2 = 0.72$; Table 2). In this case, the estimated CL for bulk deposition of N was 4.68 kg ha\textsuperscript{-1} year\textsuperscript{-1} (95\% confidence interval: 3.62-5.8 kg ha\textsuperscript{-1} year\textsuperscript{-1}). Based on previously published root biomass data (Gruelke et al., 1998) and throughfall and streamwater NO\textsubscript{3} data from this study at the same sites, we found a 26\% reduction in fine root biomass at a throughfall N deposition of 17 kg ha\textsuperscript{-1} year\textsuperscript{-1} and streamwater peak NO\textsubscript{3} concentration of 10.8 ueq L\textsuperscript{-1} (Table 2). As mentioned previously, this is a very preliminary result because the root biomass, N deposition and streamwater data are from only three sites across the N deposition gradient in the San Bernardino Mountains. These tentative results indicate that a significant reduction in fine root biomass occurred at the N deposition threshold (17 kg ha\textsuperscript{-1} year\textsuperscript{-1}) at which elevated NO\textsubscript{3} leaching was observed and that approximately 11 ueq L\textsuperscript{-1} can be considered as a tentative critical value for peak streamwater NO\textsubscript{3} concentrations associated with standing root biomass reductions. Medium-sized roots were impacted to an even greater degree than fine roots during the early July root growth flush (Gruelke et al.; 1998); but since the effect was equally strong at the moderate deposition site compared to the most polluted site, an estimate of the CL for this effect was not possible.

3.2. Litter C:N as a predictor of the N deposition CL

Litter C:N values ranged from 24.2 at Camp Paivika in the western San Bernardino Mountains to 39.7 at the relatively pristine Lassen National Park in the northern Sierra Nevada. Litter C:N values were highly correlated with both throughfall N ($r^2 = 0.79$, Table 2) and streamwater NO\textsubscript{3} ($r^2 = 0.74$, Table 2). The linear regression predicted a litter C:N value of 34.1 at the critical peak streamwater NO\textsubscript{3} level of 14.3 ueq L\textsuperscript{-1}.

3.3. Empirical critical loads for lichen responses

The relationship of N concentration in L. vulpina to throughfall N was approximated by a linear model (Fig. 4). Enhanced N in L. vulpina, defined as exceedance of the 1.0\% N clean site threshold, corresponds with an empirical CL of 3.4 kg N ha\textsuperscript{-1} year\textsuperscript{-1} in throughfall (95\% confidence interval: 0.70-3.5 kg ha\textsuperscript{-1} year\textsuperscript{-1}). At this load, acidophytes, neutrophiles, and nitrophiles are predicted to comprise an average of about 40\%, 21\%, and 11\% of lichen abundance.

![Fig. 4](image-url)

**Fig. 4.** Simple linear regression of canopy throughfall nitrogen (N) vs. % N (dry weight) in the epiphytic macrolichen *Letharia vulpina*. Linear regression fits through the corrected data points defined as exceedance of the 1.0% N clean site threshold of 1.0% N content in *L. vulpina* (95% confidence interval: 0.70-5.5 kg ha\textsuperscript{-1} year\textsuperscript{-1}).
respectively. The nitrophyte contribution appears to depend also on hardwood basal area as discussed more below; the estimate presented here assumes a hardwood basal area of 0.

With increasing N in throughfall, N accumulation in *L. vulpina* increases and we witness a steady decline in the relative abundance of acidophytes in lichen communities (Fig. 5a and d). Based on the linear relationship between the acidophyte index and throughfall N measurements, the relative abundance of acidophytes is predicted to decline by about 5.6% for every 1 kg ha⁻¹ year⁻¹ increase in N deposition. In contrast, the relative abundance of neutrophytes is predicted to increase by about 3.2% for every 1 kg ha⁻¹ year⁻¹ increase (Fig. 5b). This relationship is less clear than that of the acidophytes, although throughfall N still explains well over half of the variability in the neutrophyte index ($r^2 = 0.58$). If we consider the response of both acidophyte and neutrophyte indicator groups in tandem, acidophyte dominance would likely shift to neutrophyte dominance when throughfall N reaches about 5.2 kg N ha⁻¹ year⁻¹ (95% inverse prediction interval: 4.0-7.9 kg N ha⁻¹ year⁻¹; Fig. 5c).

Proportion of nitrophyte abundance (PNA; data not shown) also increased with increasing N in throughfall deposition, although the correlation to throughfall data is much weaker than for the other two indicator groups ($PNA = 0.029(- \text{throughfall N}) + 0.0685; r^2 = 0.33, p = 0.06$). It is likely that this relationship is more variable due to the natural preference for and greater abundance of nitrophytes on hardwood substrates, which were present in only four of the 11 sites. Inclusion of a variable that accounts for the proportion of stand basal area in hardwoods greatly improves the linear model ($r^2 = 0.77$).

3.4. CL determined from simple mass balance

When the default parameterization of the SMB model for conifer species on a sandy loam soil was used (Table 4), which are very conservative values, a CL value of 1.4 kg ha⁻¹ year⁻¹ of total N deposition was derived. However, when a new parameterization was carried out considering more realistic values provided in the literature for $N_{\text{car}}$ (the long-term immobilization of N in soil organic matter) and $[N]_{\text{le}}$ (the acceptable leaching of N), the original estimate was increased by more than 6 times to 8.8 kg N ha⁻¹ year⁻¹. Even this higher calculated estimate is well below the empirically derived CL of 17 ± 2 kg N ha⁻¹ year⁻¹ for peak streamwater NO₃ concentrations and root biomass reduction.

3.5. CL information from the DayCent biogeochemical model

Based on historical N deposition scenarios and the criteria for NO₃ threshold exceedance (peak stream water NO₃ above 0.2 mg L⁻¹ at least once every 10 years on average), the threshold was first exceeded at CP around 1956 (Fig. 6b),
which is 16 years after elevated N deposition began in 1940. When deposition at CP was set at the empirical CL of 17 kg ha\(^{-1}\) year\(^{-1}\) beginning in 1940, streamwater NO\(_3\) did not exceed the threshold until 1978, a period of 38 years (Fig. 6a).

With 40% reduced precipitation and N deposition set to 17 kg ha\(^{-1}\) year\(^{-1}\), elevated NO\(_3\) leaching did not occur at CP. With 40% reduced precipitation and historical N deposition inputs, the NO\(_3\) threshold was reached after 31 years in 1971 (Fig. 6b).

At BF, a constant N deposition of 17 kg ha\(^{-1}\) year\(^{-1}\) did not result in NO\(_3\) concentrations above the threshold except when precipitation was more than doubled by using the precipitation inputs from CP. In the latter case, NO\(_3\) leaching first exceeded the threshold after 16 years in 1956 (Fig. 6c). Nitrate leaching never exceeded the threshold at BF under historical N deposition rates, although when using the CP precipitation inputs peak NO\(_3\) concentrations were >=0.2 mg L\(^{-1}\) on four occasions beginning in 1984 (Fig. 6d).

At the empirical CL, annual N export as NO\(_3\) peaked at 1-2 kg ha\(^{-1}\) year\(^{-1}\) in high years at CP but values rarely reached 1.0 kg ha\(^{-1}\) year\(^{-1}\) with 40% reduced precipitation (Fig. 7a). At the empirical CL at BF, NO\(_3\) export was always below 0.5 kg ha\(^{-1}\) year\(^{-1}\) with normal precipitation (Fig. 7c). With doubled precipitation (CP precipitation levels) at BF NO\(_3\) export was sometimes greater than at CP (Fig. 7a,c). Under historical N deposition scenarios, elevated NO\(_3\) export began to occur regularly at CP (ranging from 3 to 17 kg ha\(^{-1}\) year\(^{-1}\)) after 1965, 25 years after elevated N deposition began (Fig. 7b). At BF historical N deposition did not affect NO\(_3\) export at either precipitation level, although NO\(_3\) export was increased from maximal values of 0.2 kg ha\(^{-1}\) year\(^{-1}\) to 0.5 kg ha\(^{-1}\) year\(^{-1}\) (and two points near 1 kg ha\(^{-1}\) year\(^{-1}\)) when CP precipitation inputs were applied to the model (Fig. 7d).

At the empirical CL, annual gaseous losses of N showed an immediate response within the first year of increasing N deposition to 17 kg ha\(^{-1}\) year\(^{-1}\) at both sites (Fig. 8a,c), and emissions were greater at BF (maximum values of 9 kg ha\(^{-1}\) year\(^{-1}\) or greater near the latter years) than at CP (generally 6 kg ha\(^{-1}\) year\(^{-1}\) or less). Under the estimated historical N deposition scenarios, current maximum values for gaseous N emissions at CP were approximately 15 kg ha\(^{-1}\) year\(^{-1}\) compared to 4 kg ha\(^{-1}\) year\(^{-1}\) at BF (Fig. 8b,d). Reduced precipitation significantly reduced gaseous emissions at CP, particularly at the empirical CL of 17 kg ha\(^{-1}\) year\(^{-1}\) (Fig. 8a). At BF increased precipitation had the effect of increasing interannual variation with the N deposition set at 17 kg ha\(^{-1}\) year\(^{-1}\) (Fig. 8c).

The estimated N deposition threshold for NO\(_3\) leaching based on simulated NO\(_3\) concentrations in seepage water were 17 and 30 kg ha\(^{-1}\) year\(^{-1}\) at CP and BF (Fig. 9a,b). This is based on N deposition occurring over a range of values for 30-60 years and with at least one occurrence per decade on average of a peak streamwater NO\(_3\) value over the 0.2 mg L\(^{-1}\) threshold. Increasing the time period of elevated N deposition resulted in lower threshold values. Reducing precipitation by 40% at CP
resulted in an estimated threshold of 20 kg ha\(^{-1}\) year\(^{-1}\) (Fig. 9c), while doubling precipitation at BF, yielded a threshold of 7 kg ha\(^{-1}\) year\(^{-1}\) over a 30-60 year time frame. The threshold at BF increased to 16 kg ha\(^{-1}\) year\(^{-1}\) with doubled precipitation and a time frame of less than 30 years (Fig. 9d).

When considering annual NO\(_3\) export across a range of N deposition levels, increases in the mean and maximum NO\(_3\) export values were more responsive at CP than at BF at N deposition below 55 kg ha\(^{-1}\) year\(^{-1}\). At BF, NO\(_3\) export values increased slowly as a linear response with N deposition of 10-55 kg ha\(^{-1}\) year\(^{-1}\), then NO\(_3\) export at BF increased exponentially from 55 to 80 kg ha\(^{-1}\) year\(^{-1}\) (Fig. 10b,d). At current N deposition levels at CP (70 kg ha\(^{-1}\) year\(^{-1}\)) the simulated mean NO\(_3\) export is approximately 2 kg N ha\(^{-1}\) year\(^{-1}\) after 30-60 years of N deposition, with a maximum N\(_3\) export of approximately 12 kg ha\(^{-1}\) year\(^{-1}\) (Fig. 10a,c).

Patterns of simulated gaseous N emissions (annual mean and maximum values) in response to increasing N deposition were similar for the BF and CP sites, except that fluxes at BF were more than double of those at CP. At the empirical CL of 17 kg ha\(^{-1}\) year\(^{-1}\), simulated mean gaseous losses of N were approximately 4.5 and 1.8 at BF and CP, respectively (Fig. 11a,b). With the critical value for mean annual N emissions over a 30-60 year period set at 2 kg ha\(^{-1}\) year\(^{-1}\), the threshold for N deposition would be 10 and 20 kg ha\(^{-1}\) year\(^{-1}\) at BF and CP, respectively (Fig. 11a,b). If the critical value for maximum annual N emissions is set at 2 kg ha\(^{-1}\) year\(^{-1}\), the threshold for N deposition would be 6 and 13 kg ha\(^{-1}\) year\(^{-1}\) at BF and CP (Fig. 11c,d).

4. Discussion

4.1. Empirical CL for nitrate leaching and fine root biomass impacts

Elevated concentrations of streamwater NO\(_3\) is the cardinal sign of an N saturated ecosystem and a large amount of empirical data are available relating N deposition and NO\(_3\) export (Aber et al., 2003; Fenn and Poth, 1999; Gunderson et al., 2006; MacDonald et al., 2002; Van der Salm et al., 2007). The throughfall N deposition threshold established in this study for streamwater leaching of 17 kg N ha\(^{-1}\) year\(^{-1}\) is consistent with estimates from large European watersheds (Dise and Wright, 1995; Gunderson et al., 2006). Likewise, Stoddard et al. (2001) found that for catchments at stage 2 of N saturation (characterized by episodic elevated NO\(_3\) and high base-flow NO\(_3\) during the growing season; Stoddard, 1994), the average N deposition for the network of ICP Waters sites was ca. 14 kg ha\(^{-1}\) year\(^{-1}\) (see Fig. 4 of their paper). Thus, general patterns for thresholds of N deposition leading to NO\(_3\) leaching losses are quite robust (MacDonald et al., 2002), although exceptions occur because varying environmental conditions and land use and management history can alter NO\(_3\) export patterns in relation to N deposition. The threshold for NO\(_3\) leaching of 17 kg N ha\(^{-1}\) year\(^{-1}\)
in this study is derived from and applicable to watersheds on predominantly granitic soils in the southwestern Sierra Nevada and the San Bernardino Mountains, yet results are highly similar to those from the temperate regions cited above. In the European ICP Forests network, data from Mediterranean climate watersheds were sometimes outliers (van der Salm et al., 2007), although data from these sites were under represented. Outliers were particularly apparent in mid-latitude and Mediterranean sites that receive moderate N deposition, have an above average temperature and relatively high C:N ratios of the organic layer (van der Salm et al., 2007), and probably had different precipitation patterns. Thus, the NO3 leaching response of the sites in our study appear to respond to N deposition similar to those of temperate forests and not to fit the profile of the outlier Mediterranean sites in the ICP Forests network receiving moderate N deposition.

A weakness in relating N deposition to streamwater NO3 export is that N deposition is usually only measured at one or at most a few points within the catchment and may not be representative of the entire catchment (Aber et al., 2003), whereas, streamwater NO3 concentrations are a result of N cycling, N deposition and hydrologic processes that occur over the entire catchment. Accuracy of empirical thresholds determined for NO3 leaching or other system-level responses may be improved by calibration with models that estimate N deposition over the landscape (Weathers et al., 2006). Notwithstanding this scaling problem, however, the relationship between N deposition measured at select points and stream water NO3 is useful for estimating the empirical threshold for an N saturation response (Aber et al., 2003). The lichen-based or fine root biomass CL can be determined from small plots, thus providing information at finer scales of resolution. The same can be done for NO3 leaching if sufficient soil solution or soil seepage samples are collected to reveal temporal patterns of NO3 loss, although installation and maintenance of replicate soil solution samplers would require considerable effort and cost.

Peak streamwater NO3 concentrations are observed during periods of high stream flow (Fenn and Poth, 1999; Riggan et al., 1985). In montane catchments in California with elevated N deposition, episodic NO3 concentrations exceed the NO3 threshold of 14.3 μM (0.2 mg NO3-N L⁻¹) during high flow events virtually every year. Peak concentrations in the more polluted sites in southern California are usually an order of magnitude greater than this threshold. In contrast, NO3 concentrations can be very low in mid-elevation coniferous forests in California exposed to low or moderate N deposition. For example, NO3 concentrations in Log Creek, a perennial stream in a coniferous catchment in Sequoia National Park, peak NO3 concentrations were approximately 1.0 μM over a 10-year period (Williams and MeNack, 1997). The NO3...
Fig. 9. DayCent simulated trends in annual peak NO$_3$-N concentrations in streamwater that occurred within various time periods since elevated N deposition began in 1940. In these simulations N deposition since 1940 ranged from 0 to 80 kg N ha$^{-1}$ year$^{-1}$. The model was parameterized for the CP and BF sites in the San Bernardino National Forest. The prescribed N deposition values were set with <10% random variation.

Fig. 10. DayCent simulated annual NO$_3$-N export in streamwater that occurred within various time periods since elevated N deposition began in 1940. (a) mean value at CP site, (b) mean value at BF site, (c) the max. values at CP site, and (d) the max. value at BF site. Note that the y-axis scales differ among the plots. See Fig. 9 for a description of N deposition scenarios.
threshold selected for peak NO$_3$ runoff is a highly conservative value, indicative of watersheds at the incipient stage of the trajectory towards above-normal N losses from the system. Considering catchment surveys in North America and Europe, Gundersen et al. (2006) recommended an annual mean streamwater NO$_3$ threshold of 0.5 mg L$^{-1}$ (36 uM) as indicative of a "leaky" system. Because this is a mean value, it is not directly comparable to our peak NO$_3$ value (14.3 uM or 0.2 mg NO$_3$N L$^{-1}$), but the comparison indicates that the NO$_3$ threshold derived for our study is a highly protective value for determining the empirical CL for streamwater NO$_3$.

However, using peak NO$_3$ runoff concentrations has functional virtue because high runoff events are the period and process when the greatest sensitivity to ecosystem disturbance is apparent. Furthermore, seasonality is pronounced in these Mediterranean catchments and using annual means may make it difficult to detect incipient N saturation. The chosen peak NO$_3$ threshold is based on reviews of "natural" NO$_3$ leaching concentrations in relatively pristine catchments in Sweden (De Vries et al., 2007) and in many regions of North America (Fenn et al., 2002; Stoddard, 1994; Stottlemyer and Hanson, 1989; Stottlemyer and Toczydłowski, 1991; Williams et al., 1996), but with a particular emphasis on catchments in California (Chorover et al., 1994; Fenn and Poth, 1999; Meixner and Fenn, 2004; Riggin et al., 1985; Williams and Melack, 1997).

The previously documented negative response of standing fine root biomass in ponderosa pine across the air pollution gradient in the San Bernardino Mountains was used to estimate a preliminary N as a nutrient CL with linkage to a harmful biological effect. Establishment and quantification of the causal connection between N deposition or NO$_3$ concentration in seepage water and a biological effect is often the most difficult step in determining the N CL (De Vries et al., 2007). The response of decreasing fine and medium root biomass with increasing air pollution exposure across the San Bernardino Mountains is a clear and dramatic response, although data were only collected at three sites across the gradient (Grulke and Balduman, 1999; Grulke et al., 1998). Nitrogen deposition or greater N fertility (Nadelhoffer, 2000) and ozone (Grulke et al., 2001) are known to cause decreased fine root biomass and the observed response in the San Bernardino Mountains is thought to be a result of the combined effects of ozone and N enrichment. Although, the effects of ozone on fine root biomass can be considered as a confounding factor in determining a N deposition CL, we consider that the empirical CL for N should be determined under conditions of simultaneous exposure to ozone because in polluted California mixed conifer forests N deposition and ozone exposure are inseparable. The fine root biological response to N deposition and ozone is clear, but data from more sites are needed.

Fig. 11. DayCent simulated annual N gaseous emissions that occurred within various time periods since elevated N deposition began in 1940: (a) mean value at CP site, (b) mean value at BF site, (c) the maximum value at CP site, and (d) the maximum value at BF site. Note that the y-axis scales are different for CP and BF. See Fig. 9 for a description of N deposition scenarios.
to better quantify the response to N deposition. From currently available data we estimate that at the N deposition threshold for elevated NO₃ leaching (17 kg ha⁻¹ year⁻¹) standing fine root biomass is reduced by 26% during the early season root growth flush period (Gruulke et al., 1998). Based on this preliminary evidence, we propose the N deposition threshold of 17 kg ha⁻¹ year⁻¹ for NO₃ leaching as a first estimate of the CL associated with reduced fine root biomass in ponderosa pine in California mixed conifer forests.

4.2. Predictors or indicators of CL exceedance for N as a nutrient

Like the European studies mentioned above (van der Salm et al., 2007), our results also illustrate that Mediterranean forests can have higher forest floor C:N ratios than temperate forests, even in N saturated sites. In temperate forests the C:N threshold of 25 was a strong predictor of sites predisposed to high net nitrification rates and high levels of NO₃ leaching under chronic N deposition, even with N deposition as low as 10 kg ha⁻¹ year⁻¹ (Gundersen et al., 2006; MacDonald et al., 2002). In our study the critical value for litter C:N associated with elevated NO₃ leaching was 34.1, 36% higher than in the European network and higher than an infertile red spruce site in Vermont where the onset of forest floor nitrification occurred below a C:N ratio of 31 (McNulty et al., 1991). In a survey of sites in northeastern North America, elevated net nitrification activity was correlated with C:N ratios between 20 and 25 (Aber et al., 2003), again indicating that forest floor C:N ratios are significantly higher in Mediterranean forests, even with elevated N deposition.

Breiner et al. (2007) reported that the critical value for foliar N in ponderosa pine at the N deposition threshold for stream water NO₃ leaching was equivalent to the critical level for foliar N (1.1% N) determined experimentally for ponderosa pine nutritional needs (Powers, 1981). By comparison, in European temperate forests the foliar N threshold was 1.4% and with an N concentration below this value, NO₃ leaching did not occur (Gundersen et al., 2006). The relationship between N deposition and litter C:N or foliar N (Gundersen et al., 2006) and NO₃ leaching suggests that these simple N status indicators can be used to identify sites where the CL is likely exceeded. These indicators are less temporally variable than streamwater NO₃ and their implementation could be useful in locating areas at risk for the effects of excess N. However, the critical values indicative of CL exceedance for foliar N tend to be lower and values for litter C:N higher in southern California forests than in temperate forests in Europe and northeastern North America (Aber et al., 2003; Breiner et al., 2007; Gundersen et al., 2006; MacDonald et al., 2002). Further data are needed to see if this pattern holds for Mediterranean ecosystems in general (van der Salm et al., 2007).

4.3. Empirical critical loads for lichen responses

We found evidence that a distinct shift in lichen community composition, from sensitive to more tolerant species, was in progress at the N critical load of 3.1 kg N ha⁻¹ year⁻¹. These community changes culminate in a shift away from the natural state of acidophyte dominance at a fairly moderate N load of 5.2 kg ha⁻¹ year⁻¹. The maintenance of healthy acidophyte populations is of particular concern because they are an integral part of food webs and are used as nesting material and habitat for insects, mollusks, birds, and animals, in the Sierra Nevada (McCune et al., 2007). The bearded lichen, Bryoria fremontii (TUCK). Brodo & D. Hawksw, is an example of an acidophyte that serves as forage for a wide variety of species in our study region, including the northern flying squirrel (Glaucomys sabrinus), Douglas squirrel (Tamiasciurus douglasii), black-tailed deer (Odocoileus hemionusi), and a variety of invertebrates (McCune et al., 2007). These foragers are themselves key parts of the diets of many insectivores and carnivores. A reduced abundance of acidophytes clearly has direct ecological consequence. Our data predict complete extirpation of acidophytes from the lichen community at an N load of about 10.2 kg ha⁻¹ year⁻¹. The midway benchmark in which the relative abundance of acidophytes in the lichen community is expected to drop by 50%, given the model, is 5.7 kg N ha⁻¹ year⁻¹ (note that even if no acidophytes are lost, a drop in relative abundance can still be caused by an increase in non-acidophytic species, which inflates the index denominator). At the other extreme, we see what could he an effect at even the lowest N levels; the N load that initiates acidophyte depression potentially lies outside the range of our data (≤ 1.2 kg N ha⁻¹ year⁻¹) although more samples are needed from low N levels to better understand this relationship.

Our lichen CL of 3.1 kg N ha⁻¹ year⁻¹ is much lower than those adopted by the UNECE for Europe (10-15 kg ha⁻¹ year⁻¹) for lichens in temperate and boreal forests (Bobbink et al., 2003). However, the European CL range is primarily based on expert judgment (with an admitted shortage of lichen-based CL research), focuses on boreal forests, includes effects on algae, and the CL seems primarily based on extirpation of species. Bobbink et al. (2003) concluded that the disappearance of epiphytic lichen species from oak woodlands in Scotland and NW England in response to varying N deposition suggests a CL of 11-18 kg ha⁻¹ year⁻¹. Our prediction of extirpation of acidophytes at 10 kg N ha⁻¹ year⁻¹, falls within the European CL range for lichens in forests (10-15 kg ha⁻¹ year⁻¹; Bobbink et al., 2003), although we argue that these CL values are much too high if the CL allows sensitive and biologically valuable taxa to be extirpated from the forest.

Lichens (Fenn et al., 2003b, 2007; van Herk et al., 2003) and diatoms (Baron, 2006) appear to be the most sensitive terrestrial and aquatic responders to low levels of N enrichment. As such, lichens are useful bioindicators for determining a CL for terrestrial biotic community changes. Lichens can be considered as the "canary in the mine" or early warning indicators of biological change from air pollution, and N deposition in particular. In areas where the lichen CL is exceeded, biological effects are already occurring and further ecological impacts can be expected as N accumulates in the system and if N deposition increases. Based on the lichen CL
determined in this study, we conclude that known biological impacts on lichen communities are clearly occurring at N deposition levels as low as 3-5 kg ha\(^{-1}\) year\(^{-1}\). In California, this level of deposition or greater occurs over most of the forested region of the San Bernardino and San Gabriel Mountains, the western portions of the Sierra Nevada, and large areas of other mountain ranges in southern California. Critical load exceedance can be evaluated by either surveying the lichen community and employing the N indicator indices or by determining %\(\text{N}\) in *L. vulpina*.

### 4.4. Simple mass balance CL

Theoretical critical loads for nutrient N were calculated using the steady state 5MB approach (UNECE ICP, 2004). The calculated CL values (1.4 and 8.8 kg N ha\(^{-1}\) year\(^{-1}\)) were well below the empirical N deposition CL of 17 kg ha\(^{-1}\) year\(^{-1}\) which leads to incipient NO\(_3\)\(^-\) leaching and approximately 25% fine root biomass reduction in these ecosystems. De Vries et al. (2007) concluded that the use of the critical N concentration in soil seepage water suggested in the UNECE manual is likely to lead to calculated CL lower than empirical values. Many uncertainties remain to be resolved to parameterize the SMB model, particularly in linking the impairment of soil-processes with biological impacts (UNECE ICP, 2004).

The DayCent results and field data (Fenn and Poth, 2001) suggest that trace gas N emissions, represented as the denitrification fraction calculated for the SMB model, are underestimated. Emissions of NO\(_3\), largely a byproduct of enhanced nitrification rates in the upland forest and denitrification rates in the riparian zones, are important emissions sources not accounted for in the SMB calculations (Fenn et al., 1996; Fenn and Poth, 2001). DayCent and empirical data also suggest that acceptable leaching may be overestimated by the revised SMB model estimate (Table 4). The N immobilization term in the SMB model refers to long term N accumulation in soil organic matter and would appear to be underestimated, but no data from our sites are available to evaluate this. However, average N immobilization determined for the Nordic forests was 8 kg ha\(^{-1}\) year\(^{-1}\) (De Vries et al., 2007), more than double our highest calculated value.

Another factor contributing to an underestimate of the calculated CL may be the enhanced N storage in the forest floor and in woody biomass and necromass at our sites as a result of the combined effects of ozone and chronic N deposition (Fenn et al., 2003c). These N pools are not normally considered within the N immobilization term of the 5MB model and some N from these pools is slowly released via decomposition. In these summer dry Mediterranean ecosystems, litter decomposition is slow and recurring fire is normally a major N release mechanism (Johnson et al., 2008). However, because of long term fire suppression, N deposition and co-occurring ozone exposure, large pools of organic N accumulate in the forest floor and in woody material, suggesting that in a functional sense, N immobilization in our sites is likely much greater than the SMB model predicts (Fenn et al., 2003c, 2005). Our best estimates (from empirical data, modeling and literature) of the terms used to derive the 5MB CL are given in the final column of Table 4. Based on our best estimate of the empirical CL of 17 kg N ha\(^{-1}\) year\(^{-1}\), we estimated by difference the N immobilization term to be 12 kg ha\(^{-1}\) year\(^{-1}\), compared to values of 1.0 and 3.5 kg ha\(^{-1}\) year\(^{-1}\) using the original and revised SMB protocols (Table 4).

Another issue for Mediterranean forests is how the 5MB model does not account for the potentially significant influence of forest management practices, prescribed fire, or wildfire events on plant nutrient uptake (\(N_u\)) in wild or recreational areas. For instance, fire would likely produce a sharp increase in N CL because N uptake would be large in the early stages of regrowth and the ecosystem may not reach steady state if forest fires are recurring, especially considering that ponderosa pine forests may not reach an old growth steady state condition until 250 years (Law et al., 2003), far from the 100 years allowed in the SMB model to reach a steady-state condition. Also important for Mediterranean forests is the strong seasonality of N mobilization during wet season precipitation and runoff events leading to temporal extremes of increased risk of NO\(_3\) leaching. Another factor is large inter-annual variability from drought cycles that may last 4-7 years, which leads to boom or bust periods of N accumulation and release particularly if disturbance such as fire occurs (Rigge et al., 1994).

### 4.5. DayCent model thresholds for N loss

The simulated estimate of the N deposition threshold for NO\(_3\) leaching at CP (high N deposition site) was 17 kg ha\(^{-1}\) year\(^{-1}\) (same as the empirical estimate) compared to 30 kg ha\(^{-1}\) year\(^{-1}\) at BF (low N deposition site). These results apply to model runs with elevated N deposition for 30-60 years, a time frame considered relevant to the historical period of N deposition in the region. Shorter periods of N deposition resulted in higher threshold values, while longer periods of N deposition yielded lower values.

A higher N deposition threshold for NO\(_3\) leaching at BF appears caused by lower precipitation at that site. When CP precipitation was used for BF, the threshold after 30-60 years of deposition was 7 kg ha\(^{-1}\) year\(^{-1}\). The threshold at BF (with CP precipitation) is lower than at CP presumably due to the lower plant and microbial biomass and thus lower N demand and retention capacity at BF. Thus, more N is available in soil with potential to be leached or emitted as trace gases.

It should be recognized that DayCent is not a watershed-scale model and the leached NO\(_3\) values actually represent NO\(_3\) concentrations in seepage water beneath the soil profile. Seepage water frequently has higher NO\(_3\) concentrations than streamwater because denitrification, particularly in the riparian zone and in-stream consumptive processes, can reduce NO\(_3\) concentrations (Gundersen et al., 2006). However, in some cases N deposition can enter the stream directly during snowmelt producing elevated concentrations not observed in seepage waters (Stoddard, 1994). For comparison purposes and because of the inherent uncertainty in the DayCent simulations, we are treating the NO\(_3\) leaching model outputs...
as comparable to the empirical winter runoff NO$_3^-$ data. In fact, the model results for CP under the normal precipitation and historical N deposition scenario show only 41% of the years from 1965 to 2005 where the NO$_3^-$ concentrations exceed the threshold (14.3 umol L$^{-1}$). In contrast, streamwater NO$_3^-$ concentration data from catchments draining the CP site far exceeded the threshold every year since 1995 when data were available (Fenn and Poth, 1999; Meixner and Fenn, 2004).

Under historical N deposition at CP, simulated threshold NO$_3^-$ concentration was first exceeded at CP in 1956, 16 years after elevated deposition began in 1940. Thus, our best estimates are that elevated NO$_3^-$ leaching, the cardinal symptom of a N saturated catchment, began around 1956. However, this may be a conservative estimate of the time to N saturation or CL exceedance considering that under the high N deposition loading at CP, the model underestimates peak NO$_3^-$ concentrations in many years. Li et al. (2006) also reported that DayCent underestimated stream NO$_3^-$ export in January in a small chaparral catchment in Sequoia National Park. The earliest field data in the region showing chronic NO$_3^-$ leaching is from the San Dimas Experimental Forest in the San Gabriel Mountains. Streamwater NO$_3^-$ concentrations and annual N export values similar to those near CP have been reported since 1979 (Riggan et al., 1985, 1994; Meixner et al., 2006). Because of the unusually high streamwater NO$_3^-$ concentrations that have been reported for San Dimas since 1979 (as high as 500 ueq L$^{-1}$; Riggan et al., 1985) and the historically-elevated N deposition at San Dimas (Bytnerowicz et al., 1987), N saturation conditions likely developed there prior to the 1970s. Although the San Dimas data are from low-elevation chaparral catchments located closer to Los Angeles than our San Bernardino mountain sites, data from San Dimas support the estimation of N saturation conditions occurring in catchments in the Los Angeles Basin by around 1960 if not sooner. Considering that we estimate N deposition at CP to be 35 kg ha$^{-1}$ year$^{-1}$ by 1960, this would not be an unexpected result.

Reduction of precipitation by 40% at CP was associated with pronounced reductions in stream water NO$_3^-$ concentrations and gaseous N losses when N deposition was set at the empirical N deposition threshold of 17 kg ha$^{-1}$ year$^{-1}$. Stream water NO$_3^-$ concentrations were less affected under historical N deposition inputs, but gaseous losses of N at CP were greatly reduced by a 40% reduction in precipitation. The latter effect may be due to soil drying and less favorable conditions for nitrification and denitrification. Doubled precipitation at BF caused dramatic increase in NO$_3^-$ leaching but mainly affected gaseous losses by increasing the interannual extremes. Thus, it seems that the greatest effect of increasing precipitation is to increase NO$_3^-$ flushing and leaching from the soil profile at sites with moderate N deposition.

Gaseous losses of N in the model responded within the first 2 years in most cases to N deposition levels of 17 kg ha$^{-1}$ year$^{-1}$ or greater. At BF under historical N deposition (currently 8.8 kg ha$^{-1}$ year$^{-1}$), trace gas emissions increased slowly and did not maintain elevated fluxes until around 1975, 35 years after deposition began. At CP and BF the percent of N deposition re-emitted as N trace gases in the year 2000 according to DayCent was 20 and 22% respectively. These results agree with European studies demonstrating that temperate forests show increased NO and N$_2$O fluxes when throughfall N deposition was greater than 15 kg ha$^{-1}$ year$^{-1}$ and that 17-20% of the atmospherically deposited N was re-emitted as NO and N$_2$O (Gasche and Papan, 2002; Skiba et al., 2004).

4.6. Comparison of CL approaches

We consider the empirical approach to produce the most reliable estimates of N deposition CLs. As expected, the empirical CL for lichen effects (3-5 kg ha$^{-1}$ year$^{-1}$) was much lower than for N saturation responses based on NO$_3^-$ leaching or fine root biomass reduction (empirical value of 17 kg ha$^{-1}$ year$^{-1}$) or gaseous N losses (DayCent simulated CL of 20 and 10 kg N ha$^{-1}$ year$^{-1}$ at the CP and BF sites, respectively). Lichen-based CL values can function as early warning sentinels because lichens respond to low levels of N deposition and the response is much more rapid than the broader biogeochemical cycle indicators such as foliar N, litter C:N or streamwater NO$_3^-$ concentrations. This is not surprising, considering that lichens take up N exclusively from the atmosphere and have no means to regulate uptake.

The revised CL estimate from the SMB model was 8.8 kg ha$^{-1}$ year$^{-1}$, which is similar to current N deposition at BF and approximately 50% lower than the empirical CL for fine root biomass and DayCent simulated and empirical thresholds for NO$_3^-$ leaching. The forest at BF is still highly N-limited and exhibits highly conservative N cycling with no observable impacts on fine root biomass (Gruulke et al., 1998). The SMB model needs further refinement to more accurately estimate the N as a nutrient CL in mixed conifer forests of California. Each of the component terms of the SMB model entail considerable uncertainty (Heywood et al., 2006).

The DayCent biogeochemical model simulations provided an estimate of the time to develop N saturation symptoms and also predicted N storage pool sizes associated with NO$_3^-$ export. These types of predictions are difficult to make with empirical data because historical N deposition and NO$_3^-$ export data are limited and it is difficult to collect N budget data from replicated sites across N deposition gradients. Empirical data from the CP and BF sites on stream water NO$_3^-$ export, trace gas emissions, and N pool sizes (Fenn and Poth, 1999, 2001; Fenn et al., 2005) indicate that the DayCent model produced reasonable simulations of the major N cycling processes. In summary, the two empirical CL approaches and the DayCent model each contributed useful information toward understanding CL for important ecosystem responses to chronic N deposition.

4.7. Special considerations for Mediterranean ecosystems

Although the empirical and simulated thresholds for NO$_3^-$ leaching were similar to values from northern temperate...
forests, some characteristics of the Mediterranean forests in southern California may have an important influence on N loss and CL in these ecosystems. DayCent modeling results suggest that annual precipitation is an important factor affecting N losses as NO$_3^-$ and as trace gas emissions.

The CL and N loss thresholds determined in this study were based on empirical and modeling results in the absence of fire for the last 100-160 years. However, these are ecosystems with estimated fire return intervals of 15-29 years in the absence of fire suppression (McBride and Laven, 1976), but possibly as long as 50 years (Minnich, 1999). Prescribed fire and particularly wildfires that have occurred in some of these forests in recent years can have significant effects on stand condition and N cycling. These processes raise questions surrounding the forest steady state concept which is the basis for the SMB CL calculations. The strong seasonality of the Mediterranean climate is characterized by prolonged summer drought when N deposition accumulates on canopies and in the upper soil profile, followed by heavy winter rains that mobilize much of the accumulated N during the period of decreased plant N demand. This seasonality also suggests possible seasonal differences in the CL. The empirical and simulated NO$_3^-$ leaching approach used in this study is based on peak NO$_3^-$ concentrations in winter runoff and is designed to account for this seasonality. But further research is needed to evaluate the consequences of this strong seasonality in N fluxes on the harmful biological effects which CL are designed to prevent.

Another important factor possibly affecting the CL in our study sites is the co-occurrence of elevated ozone. Ozone causes major physiological disruption of the dominant pine species in these forests, reducing fine root biomass, altering water relations, and affecting nutrient uptake and allocation within the plant and the ecosystem (Fenn 2003c; Grulke and Balduman, 1999; Grulke et al., 1998), which can affect the CL for N as a nutrient. Ozone and N deposition also predispose trees to bark beetle attacks and mortality (Jones et al., 2004), which should ultimately be considered in CL evaluations as well, but data are needed from a greater number of sites.

4.8. Management implications of CL findings

Simulated streamwater NO$_3^-$ concentrations suggested that with longer deposition periods, the N deposition threshold leading to NO$_3^-$ export decreases. For example, the simulated threshold at CP after 30-60 years of deposition was 17 kg N ha$^{-1}$ year$^{-1}$ but decreased to 13 kg N ha$^{-1}$ year$^{-1}$ after 90-120 years of N deposition. At BF, comparative threshold values were 30 and 22 kg N ha$^{-1}$ year$^{-1}$. This suggests that because N deposition has already been elevated for the past 60-70 years, future N deposition at N saturated sites such as CP will likely need to be reduced to less than 13 kg ha$^{-1}$ year$^{-1}$ to return the ecosystem to a condition of conservative N cycling. It is still uncertain how many years N deposition levels would need to remain below a given threshold to restore forests to a conservative N cycle due to the current large N stores within these ecosystems.

DayCent simulations indicated that N losses from the ecosystem begin to accelerate when total forest N pools in soil, forest floor and living biomass reach ca. 2000 kg ha$^{-1}$. Nitrogen stocks at CP are estimated to be two to three times greater than this threshold (Arbaugh et al., 1999; Fenn et al., 2005). However, the model indicates that mineral soil N is the largest and most important N pool driving N losses, and that N losses as leached NO$_3^-$ and as gaseous emissions accelerate above a threshold value of 1400 kg N ha$^{-1}$ year$^{-1}$ at CP. Current N stores in the top 10 cm of the mineral soil at CP and BF are 1625 and 1390 kg ha$^{-1}$, respectively (Penn et al., 2005). At CP, the N pool in the forest floor is of similar magnitude to the mineral soil and modeling results and field N cycling data for CP (Fenn et al., 2005) show that N losses are strongly related to N in the forest floor as well. Simulations suggest a threshold N pool in the forest floor associated with N loss of 450 kg N ha$^{-1}$ compared to field values at CP and BF of 1370 and 580 kg N ha$^{-1}$ (Fenn et al., 2005). These data and simulation results support the conclusion that N losses at CP will only be brought under control after N capital on the site has been reduced by a large but uncertain amount. Similarly, we estimate that N deposition may require as much as a five-fold reduction.

5. Conclusions

Our conservative, low critical load (3.1 kg ha$^{-1}$ year$^{-1}$) for lichen responses is expected to protect the integrity of the lichen community, and thus preserve its ecological roles in the food web, nutrient cycling, and wildlife usage in conifer forests of the Sierra Nevada. Protecting lichens, the most sensitive ecosystem component, from adverse effects of N deposition also provides de facto protection for more tolerant ecosystem components. This lichen CL is exceeded over large portions of the western Sierra Nevada, including as far north as the Tahoe National Forest. Sensitive lichen species have been extirpated from polluted forest sites in the Los Angeles Basin, but because of steep N deposition gradients, concentrations of NO$^+_3^-$ in surface runoff and groundwater are mainly impacted in regions at least partially exposed to corridors of polluted air mass transport. The empirical and simulated throughfall N deposition threshold for NO$_3^-$ leaching in mixed conifer forests and the preliminary CL for a 26% reduction in fine root biomass is estimated at 17 kg N ha$^{-1}$ year$^{-1}$. This CL is exceeded in much of the western and central San Bernardino Mountains, along the southern face of the San Gabriel Mountains in the Los Angeles Air Basin, and in several areas of the southwestern Sierra Nevada Mountains. Simulation modeling results conservatively estimate that elevated NO$^+_3^-$ leaching began in the late 1950s at a polluted site in the western San Bernardino Mountains, which corresponds with the first reports of foliar ozone injury at this site in 1953.

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