Journal of the Nevada Water Resources Association

Spring 2005

A publication of the Nevada Water Resources Association, providing hydrologic information to the people of Nevada and adjacent States

Lake Tahoe Edition

Volume 2, Number 1

Modeled inputs of atmospheric nitrogen to the Lake Tahoe Basin due to gaseous pollutant deposition

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ABSTRACT

Causes of the decline in water quality in Lake Tahoe have been attributed to nutrient additions stemming from various watershed-related changes such as urban development on the one hand to atmospheric nutrient deposition on the other. Several independent estimates exist that quantify the magnitude of these atmospheric sources at the scale of the entire Lake Tahoe Basin (LTB), but there are no estimates of deposition at the scales appropriate for mass balance (i.e., comparison of nutrient inputs vs. outputs for a given watershed). This study leverages ambient measurements of pollutant concentrations and Geographic Information System (GIS) maps of forest canopy characteristics to estimate the variation of gaseous nitrogen (N) pollutant deposition at watershed scales in the LTB. We find that (1) modeled inputs, when integrated over the entire area of the LTB, match those produced by other methods, (2) modeled N deposition to drier east-side watersheds can be twice that of wet deposition, and (3) daytime fluxes of gaseous N are responsible for most modeled dry deposition during the three month summer period of this study. Although uncertainties created by local mountain topography and meteorology prevent definitive deposition measurements, we conclude that dry N deposition likely provides substantial N (average of 1.4 kg-N ha^{-1} or about 1/3 of annual deposition) to some parts of the LTB, especially in areas near the sources of gaseous N pollution (up to 2.1 kg-N ha^{-1}, or 1/2 of annual deposition). The potential effects of this localized N deposition (including possible linkages with aquatic and terrestrial eutrophication) warrant further study.

INTRODUCTION

The Lake Tahoe Basin (LTB) is a high elevation watershed (1900m, 1310 km^2) that lies on the border between California and Nevada in the central Sierra Nevada. The watershed (812 km^2) is a bowl-shaped basin dominated by Lake Tahoe (498 km^2), a large lake famous for its exceptional clarity. The LTB is bordered on the east side by the Carson Range and on the west by the Sierra Nevada crest, which rise 1000m above lake level at some points (Coats and Goldman, 2001). These high ridges define the LTB airshed, channeling and mixing air so that both local and regional pollutants contribute to elevated ambient concentrations of N-containing gases such as nitric acid (HNO_3), ammonia (NH_3), and nitrogen dioxide (NO_2) (Tarnay and others, 2001b; Fraczek and others, 2003).

Deposition of gaseous N compounds due these gaseous pollutants is highly uncertain, but estimates for the average amount of N deposition to the LTB fall between 1.9 kg-N ha^{-1} yr^{-1} (Jassby and others, 1994; Tarnay and others, 2001a). Compared to more urban-influenced settings (e.g., 25-45 kg-N ha^{-1} yr^{-1} in the Los Angeles air basin or 30-40 kg-N ha^{-1} yr^{-1} in some eastern sites), this amount of deposition is relatively low (Lovett and Lindberg, 1993; Munger and others, 1996; Meyers and others, 2000; Fenn and others, 2003b). Most terrestrial ecosystems are N limited, so when they receive such high loads of N from the atmosphere, the availability of N to the biota of that system can exceed the demand, rendering the system "N saturated" (Aber and others, 1998; Aber and Magill, 2004).

Usually, N saturation and its effects are seen only when levels of N deposition are relatively high (e.g., 10 kg-N ha^{-1} yr^{-1} or more); however, the high elevation systems of the West appear to have lower thresholds for ecological effects (e.g., 3-5 kg-N ha^{-1} yr^{-1}) due to the low capacity for the biota at such elevations to take up N (Burns, 2003; Driscoll and others, 2003; Fenn and others, 2003a). The biotic effects of N deposition occur in both terrestrial and aquatic portions of high elevation catchments, and range from changes in vegetation community structure to increased export of inorganic and organic N. Deposition can also cause changes in aquatic food web structure (Fenn and others, 2003a).

Abiotic factors also influence the potential effects of N deposition on a watershed, leading to a strong link between N deposition and N export. Higher elevation watersheds more efficiently "capture" N pollutants from the atmosphere via both dry and wet deposition, and have less biotic demand. This leads to higher N concentrations in runoff and transport of the N pollutants to the bottom of those catchments (Lovett and Kinsman, 1990; Stoddard, 1994). As a result, pollutants can accumulate in the aquatic environments at the bottom of these watersheds (Reiners and Driese, 2001; Reiners and Driese, 2003). Even in the most pristine and remote high-elevation watersheds, it appears that runoff can temporarily create a state of N saturation as spring snowmelt flushes the catchment of its labile N (Sickman and Melack, 1998). The result is that in many mountain environments, including Lake Tahoe and other watersheds in the Sierra Nevada, the abiotic characteristics (i.e., watershed area, soil cover, and runoff), rather than biotic factors, can drive responses to atmospheric N loads as well as N retention and availability, even in undisturbed watersheds (Sickman and others, 2002).

Unlike the watersheds of the Colorado Front Range, most watersheds in the Sierra Nevada show either no trend for N released or even a decrease in N export (Lewis and others, 1999; Clow and others, 2003). One reason for this may be related to climate effects and a gradual warming of the upper elevations, which could be increasing biotic N demand (Sickman and others, 2003). Whatever the reason, the observations imply that atmospheric N deposition in these undisturbed high-elevation settings has not increased, since such increases would likely lead to increases in N export from the catchments.

Lake Tahoe may be a notable exception to this rule, since algal growth in the lake has shifted from N limitation to P limitation over the past 3 decades, a shift that coincides with losses in clarity and increases in primary productivity usually associated with eutrophication (Goldman, 1988; Goldman and others, 1993). Initial research on atmospheric deposition showed that the LTB watersheds (Figure 1) appeared to retain most of their N (Coats and Goldman, 1993; Coats and Goldman, 2001) and that atmospheric deposition (as measured by bulk deposition buckets) appeared to be elevated (Jassby and others, 1994). As a result, this eutrophication was attributed to atmospheric N sources, especially atmospheric N transported from California Central Valley, which is the primary regional source of atmospheric N pollutants.

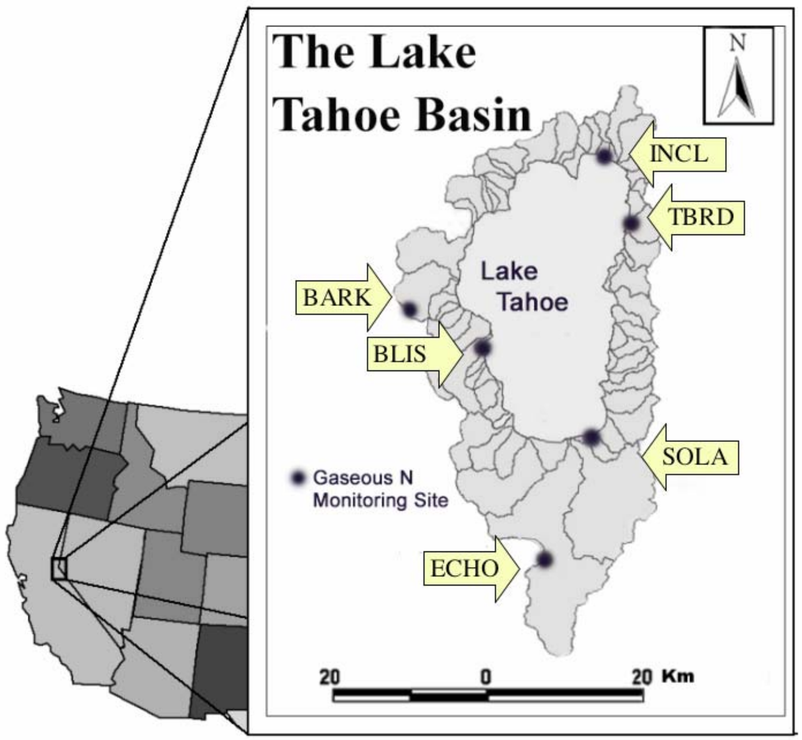

Figure 1. Sub-watershed delineations and gaseous N pollutant monitoring sites (Summer of 2000, Tarnay and others, 2001a; Tarnay and others, 2001b). BLIS = D.L. Bliss State Park, SOLA = South Lake Tahoe; TBRD = Thunderbird Lodge; INCL = Incline Village, and ECHO = Echo Summit

More recent research has shown local sources may also be responsible for substantial fractions of the atmospheric N that deposits to Lake Tahoe and its watershed. One reason is that gaseous pollutants, which are responsible for much, if not most, of deposition during summer months (Tarnay and others, 2001b) attenuate when en route from the Central Valley, especially when they have to travel above 2000m elevation (Carroll and Dixon, 2002; Zhang and others, 2002; Fraczek and others, 2003). Some of this N still contributes to the concentrations of gaseous N pollutants in the LTB. However research also suggests that local sources (i.e., urban development and heavy vehicular traffic during the summer) also contribute substantially to the oxides of nitrogen (NO_x) and gaseous ammonia (NH_3) measured in the LTB. Both Basin-wide (i.e., Jassby and others, 1994; Tarnay and others, 2001a) and site-specific (i.e., Tarnay and others, 2002) deposition estimates have shown that, in combination with wet deposition, dry N deposition can increase annual fluxes of N to the LTB to over 5 kg-N ha^{-1} yr^{-1}, well above levels that have caused effects in other high elevation systems (Williams and Tonnessen, 2000; Fenn and others, 2003a).

The problem with these deposition estimates is that the scale over which they are averaged is too coarse to provide meaningful comparisons with stream-based watershed N outputs (e.g., Coats and Goldman, 2001) or to provide accurate estimates of the maximum amounts of deposition that might be occurring at those scales. On the other hand, site-specific modeling exercises (e.g., Tarnay and others, 2002), even when performed in appropriately flat terrain (which Tahoe is not), cannot be assumed to apply to the surrounding landscape since topography, forest canopies, and concentrations of gaseous pollutants can alter the amount deposited at localized scales substantially (Hicks and others, 1985; Baldocchi and Meyers, 1998).

OBJECTIVES

The objectives of this work are to (1) quantify the potential magnitude of localized dry N deposition to watersheds of the LTB and (2) show the relative contribution of such fluxes to total N deposition at annual time-scales relative to loads known to cause ecological effects in other high elevation systems. Should the estimates show very high deposition, the GIS-based modeling framework developed for this exercise may be useful in developing strategies to reduce N deposition and any associated eutrophication in the LTB.

APPROACH AND METHODS

Our approach to obtaining these quantitative, watershed scale deposition estimates was to incorporate the gaseous N concentrations derived from a 3 month intensive study using annular denuders (Tarnay and others, 2001b) into a previously developed simple deposition model (Tarnay and others, 2002). By integrating this model into a GIS so that other drivers of deposition can be accounted for (i.e., roughly estimated meteorology and forest canopy characteristics), we mapped areas with the highest likely deposition and integrated that deposition over the area of the watersheds in question.

Pollutant deposition modeling

This modeling approach is analogous to “draping” a single layer of “leaf” over the entire area of the LTB, and changing the ratio of leaf area to ground area in every 30 m^2 patch of land. The advantage of this GIS-based “bigleaf” approach is that it quantifies the effect of two major drivers of variability in dry deposition: forest canopy structure and pollutant concentration. These parameters are readily available from monitoring sites in the LTB. As with all bigleaf approaches, the model most likely yields incorrect values in an absolute sense, but useful values in a relative sense (Raupach and Finnigan, 1988).

The disadvantage of this approach is that it does not account for the meteorological values that often alter deposition dramatically in mountain microclimates (e.g., intermittent turbulence and microclimatic gradients, Raupach and Finnigan, 1988; Baldocchi and Meyers, 1998). As a result, single layer models are often “tuned” to empirically fit data using field measurements of fluxes. The particular model used in this exercise was adapted from deposition measurements that were trained in flat agricultural fields (Hicks and others, 1987), adapted by Lindberg and Lovett, (1992) for use in forests, and modified for work in drier western environments (based on cuvette gas-exchange experiments) by Tarnay and others (2002).

The model actually doesn't calculate flux itself, rather a canopy-scale "deposition velocity" (V_d, m s^{-1}) from which flux (F_c, nmol m^{-2} s^{-1}) be calculated, if pollutant concentration (C_{gas}, nmol m^{-3}) is known (Equation 1).

$$F_c = V_d\left(C_{gas}\right)$$ **Equation 1**

Depending on the gas in question and the surface in question (e.g., plant leaves), C_{gas} is sometimes represented as the difference between the concentration of the gas in the free atmosphere and the concentration of the gas in the interior of the leaves (C_{air}-$C_{interior}$).

Calculating deposition velocity at the whole-canopy scale (V_d) starts at the scale of a single leaf, where the deposition velocity (V_g, m s^{-1}) in this single layer model is calculated by assigning a "resistance" to different stages of the process that delivers a gas or pollutant to the surface of a leaf. This process is controlled by diffusion such that gases must (1) be delivered by turbulent eddies (quantified by R_a, s m^{-1}) to a given leaf, (2) diffuse across the laminar layer of air (quantified by R_b, s m^{-1}) next its surface, and (3) "stick" to the surface of the leaf, soil, or other substrate (R_c, s m^{-1}). The R_a and R_b variables are calculated based on meteorological data (i.e., wind speed and standard deviation of wind direction) and on the diffusive properties of the gaseous pollutant in question (Hicks and others, 1987). The R_c variable is highly dependent on the deposition substrate, and often driven by wetness of that substrate (i.e., relative humidity). For plants, which have stomata that allow the diffusion of gases to the interior of the leaf, R_c is actually comprised of several more components representing the transport to the leaf interior, either through the leaf stomata, or through the sometimes-porous leaf cuticle.

By analogy to Ohm's law, the first three processes are calculated in series (Equation 2),

$$V_g = \left(\frac{1}{R_a + R_b + R_c}\right)$$ **Equation 2**

then scaled to the entire forest canopy to obtain the whole-canopy V_d value (Equation 1). The components of R_c, or the different substrates represented by R_c (e.g., soil, plant stems, rock) are calculated in parallel, similar to the resistances in an electrical circuits.

To scale the V_g value (Equation 2) up to a canopy-scale deposition velocity (V_d), the bigleaf approach has to account for the many layers of vegetation to which gaseous pollutants can deposit. It does this using the ratio of leaf area to ground area, or leaf area index (LAI). Since one layer of canopy cover can shade another, models of the extinction (due to shading) of photosynthetically active radiation (PAR) through the forest canopy modify the degree to which LAI increases deposition. Further details on these site-specific calculations are given by Tarnay and others, (2002); further details on the physical underpinnings for this deposition model are given by Hicks and others (1987), while the basic principles behind scaling of leaf level deposition estimates to canopy-scale estimates are treated by Baldocchi and Hicks (1987).

In our analysis of the spatial variability in flux, the meteorological parameters that go into calculating the R_a and R_b are not varied over space, but are derived from the meteorological data from one site (BLIS, Figure 1), which was situated several hundred feet above lake level (Tarnay and others, 2002). The data from BLIS (Figure 1) are used in all 30 m^2 pixels that contain forest canopies—only LAI and pollutant concentrations (C_{gas}, Equation 1) are varied across the area of the LTB.

Forest canopy characteristics

To verify the relationship between LAI and remotely sensed canopy cover in the LTB, we determined LAI allometrically from measurements of diameter at breast height (DBH) at 11 plots in the LTB (1999). The GIS was used to stratify these plots at different canopy cover levels, so that canopy cover (%, from US Forest Service forest inventory analysis [FIA], 1992) could be regressed with the LAI measured at the plots. At each sample plot, we compiled a data table containing species type and DBH. Diameter at breast height for white fir (*Abies concolor*), red fir (*Abies magnifica)* and Jeffrey pine (*Pinus jeffreii)* encountered within each plot was measured. As described by Kittredge (1944), this data table was then used to calculate a total DBH for each 30 x 30 plot. Only DBH for the dominant species within each plot (i.e., white fir, red fir, and Jeffrey pine) were used in the data table.

Allometric relationships from the literature (Kittredge, 1944; Gholz and others, 1976) were used to convert the total DBH at each plot to dry weight biomass. Specific leaf area relationships for white fir, red fir, Jeffrey pine (derived from greenhouse seedlings) were then used to convert from biomass to leaf area (projected). To convert from projected to all-sided leaf area in Jeffrey pine, the general value of 2.6 (Körner, 1995) was used. To convert to all sided leaf area for both red and white fir a more conservative ratio of 2.4 (Turner and others, 2000) was used, since fir needles tend to be flatter than pine needles. Leaf area for each species within the plot was then divided by the total area of the plot (uncorrected for slope) to calculate an LAI for each species. The LAI for each species was then summed to obtain a total LAI for the plot. The stratified canopy cover levels at 15, 30, 60, and 80% corresponded to LAI values (projected) of 1.0, 2.0, 4.3, and 5.8, respectively, with a least squares regression r^2 of 0.84. This relationship was then used to generate a LAI data from the FIA data layer. Each 30 m^2 pixel was assumed to have a homogeneous canopy within it, and deposition velocity was then modeled at each of these LAI levels (e.g., 1.0, 2.0, 4.3, and 5.8) so that the effect of these different LAI values on pollutant flux could be estimated at high resolution as it varied across the landscape (Figure 2).

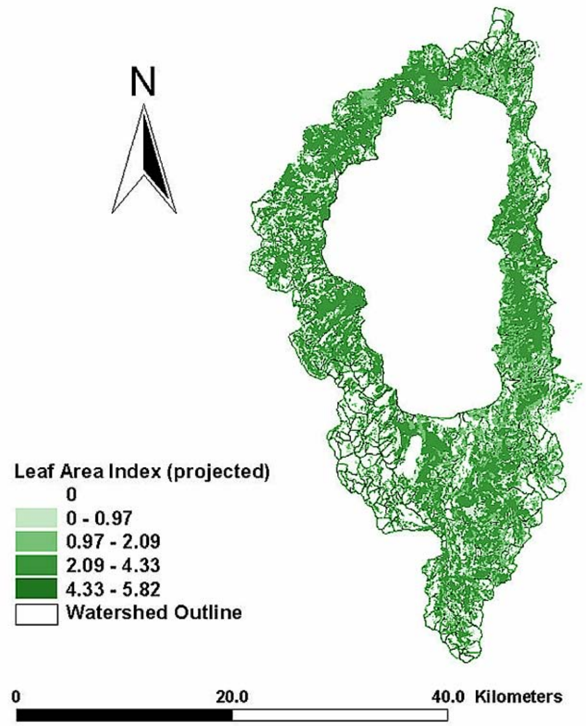

Figure 2. Leaf Area Index (LAI), as calculated from 1992 US Forest Service remote sensing data and 1999 ground-truthing plots.

Interpolated pollutant concentrations

Concentrations of HNO_3 and NH_3 in the LTB for July through September 2000 are reported in Tarnay and others (2001b) and are summarized in Table 1 along with NO_2 concentrations, which are continually monitored by chemiluminescence in the LTB at Incline (INCL), South Lake Tahoe (SOLA), and Echo Summit (ECHO, Figure 1). Chemiluminescent measurement of NO_2 is determined by the difference between total oxidized nitrogen and nitric oxide. Total oxidized nitrogen includes HNO_3, which means that if the sampling line does not scavenge HNO_3 from the air-stream, NO_2 will be overestimated. The amount of scavenging likely varies from sampler to sampler; but since HNO_3 concentrations are less than 10% of NO_2, we assume that this overestimate is negligible (Table 1).

Table 1. Average concentrations of gaseous HNO_3, NO_2, and NH_3 in the Lake Tahoe Basin.

	HNO_3 (nmol N m^{-3})*	NH_3 (nmol N m^{-3})*	NO_2 (nmol N m^{-3})**
24 hr median (± 1 S.E.)	14 (1.2)	14 (3.8)	160 (14)
Day median (± 1 S.E.)	17 (1.6)	21 (4.8)	190 (10)
Night median (± 1 S.E.)	13 (1.2)	9.7 (6.2)	96 (32)
Background/compensation point	6.6-7.1	0.0-40	65-97
Statistically significant day/night difference (p = 0.05)?	yes	yes	yes

*At standard temperature and pressure, taken from Tarnay et al. (2001b)
**Reported values from collocated NO_x sampler (California Air Resources Board)

To obtain ambient pollutant concentration values for each pixel in the GIS, median day and night concentrations at each site were interpolated over the area of the Basin using inverse distance weighting, which assumes that concentrations of pollutants drop off as a power function of increasing distance (ArcMap version 8.1, ESRI, Figure 3).

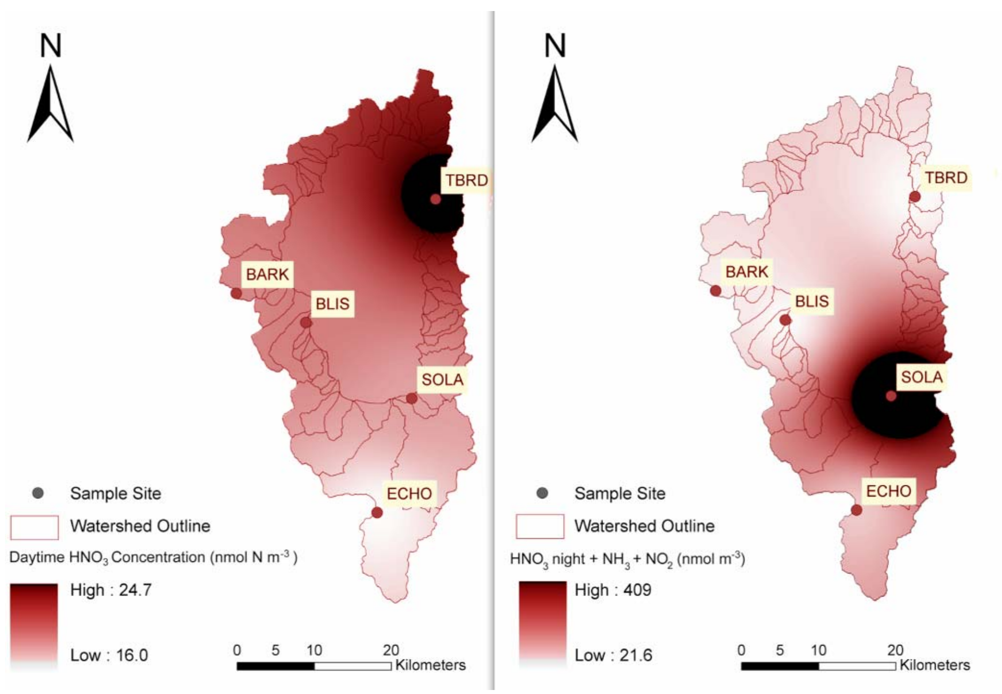

Figure 3. Anecdotal patterns in spatial distribution of gaseous N during summer 2000 (From Tarnay and others, 2001b). Left panel depicts the daytime distribution of HNO_3 (no statistical difference between sites); right panel depicts distribution of other species during day and night (SOLA was statistically different from other sites).

Both NO_2 and NH_3 exhibit compensation points over forest canopies (Hanson and others, 1989; Langford and Fehsenfeld, 1992). In order to ensure that fluxes over terrestrial surfaces accounted for compensation points, we subtracted the highest value of ranges found in the literature (Table 1) from the ambient value in each terrestrial pixel. However, canopies are heterogeneous and discontinuous throughout the LTB (Figure 2), and may not be cohesive enough under the turbulent conditions measured to result in a canopy-scale compensation point. As a result, we also calculated fluxes without these compensation points to gain obtain an upper range for the flux of pollutant gases to terrestrial LTB surfaces.

RESULTS

Average pollutant loads for the whole Lake Tahoe Basin

During the 2-month observation period of this study, the total flux of N from HNO_3, NH_3, and NO_2, integrated over the entire terrestrial area of the LTB, varied from a low of 2.0 kg N ha^{-1} (assuming a canopy compensation point for both NO_2 and NH_3) to a high of 3.8 kg N ha^{-1} (assuming no compensation point for NO_2 and NH_3, Figure 4). These estimates should be considered conservative since loads to the LTB were calculated for summer only—annual values that include fall, winter, and spring deposition are likely higher (Figure 4). To avoid the conflation of these summer-only deposition rates with the more standard annual deposition values, we therefore use units of kg-N ha^{-1} $summer^{-1}$ rather than the more common units of kg-N ha^{-1} yr^{-1} in this discussion.

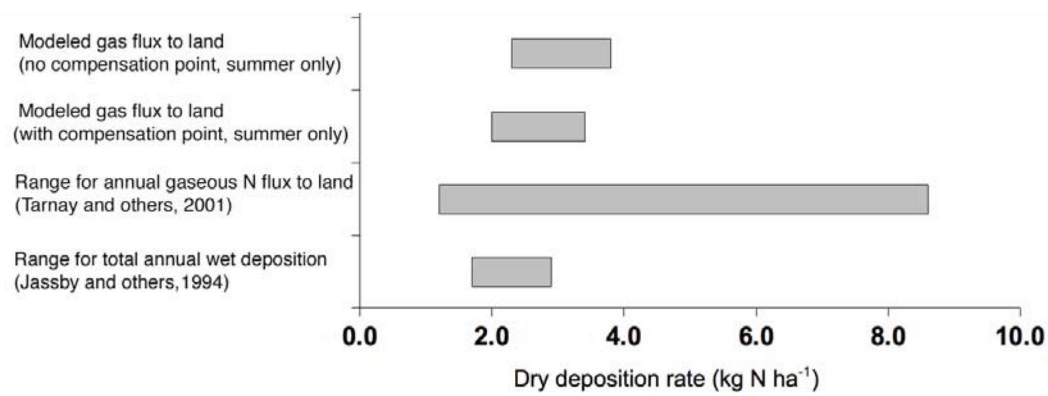

Figure 4. Comparison of summer gaseous N fluxes (with and without compensation points) with previous annual estimates. Modeled estimates are restricted to the summer months (kg-N ha^{-1} $summer^{-1}$) since critical model parameters (i.e., pollutant concentrations) were not available for the fall, winter, and spring months.

Compensation points for both NO_2 and NH_3 need also be considered in the interpretation of these numbers. Given the extreme N limitation reported by Coats and Goldman, (2001), it appears likely that plants will minimize N losses to the atmosphere, so that compensation points for closed canopies will be lower than the values reported for less extremely N limited systems (Hanson and others, 1989; Langford and Fehsenfeld, 1992). In addition, the sparse character of most canopy cover in the LTB will also inhibit the formation of inverted pollutant concentration gradients for these pollutants in all but the most stable boundary layer conditions. Therefore, we assume that terrestrial fluxes are best approximated by higher-range estimates that do not include a canopy compensation point (median = 3.8 kg N ha^{-1} $summer^{-1}$, Figure 4). These rates are similar to the literature-based estimates reported by Tarnay and others (2001) and, with wet deposition added (Figure 3), resemble moderately polluted sites in the southern Sierra Nevada rather than the less polluted montane sites to the north and south of the LTB (Bytnerowicz and others, 2000; Fenn and others, 2003b).

Modeled effects of forest canopies and spatial pollutant patterns on deposition

Application of the BLIS-derived model parameters to land canopy and pollutant values for the rest of the LTB shows that moderately high rates of deposition are not evenly distributed over the entire LTB, but may be concentrated in the immediate vicinity of and downwind of the SOLA monitoring site (Figure 5). The result is a spatial pattern of N loading that is greater on the southern and eastern sides of the Basin than on the north and west sides. According to the model most of this N (80%) was deposited during the daytime, when turbulent boundary layer conditions and open stomata allowed for maximum deposition of all pollutants. Since NO_2 and NH_3 have relatively short atmospheric residence times, most deposition outside the immediate influence of SOLA was probably due to HNO_3, with the overall fraction of HNO_3 increasing with distance from SOLA (Figure 3, right panel).

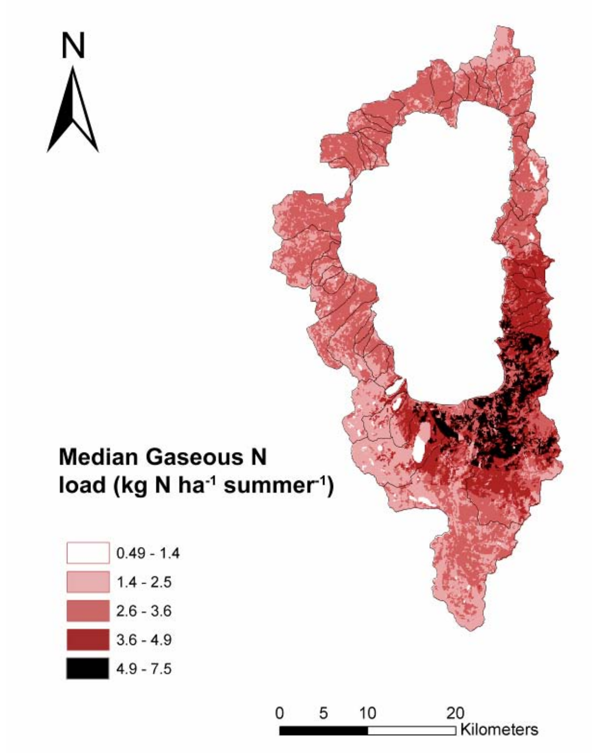

Figure 5. Modeled gaseous N load to the Lake Tahoe Basin during the summer based on 30 m^2 resolution canopy data and spatially interpolated pollutant concentrations

Care should be taken not to literally interpret the variation depicted in Figure 5 for several reasons. Most importantly, the meteorology and other model parameters (e.g., PAR) for each

30 m^2 pixel were not measured, but assumed to be the same as at the BLIS site (Figure 1). In addition, these deposition models, even when applied homogenous forest canopies and stable atmospheric conditions, have been shown to underestimate or overestimate actual fluxes by +/- 30% (Lindberg and others, 1990). Further, these models, having been trained in homogeneous terrain at lower elevations, are more likely to underestimate flux than overestimate it, and as such provide a conservative estimate of dry deposition. At these fine (i.e., 30 m^2) scales, the depicted estimates represent a conservative measure of the potential variability introduced by variations in canopy and pollutant concentrations. Actual fluxes are likely more variable and, in many sites, higher.

In terms of watershed retention, averaging over the larger area (including non-forested) of the various sub-watersheds of the LTB is thus more meaningful and likely more accurate. Integrating the modeled fluxes at these scales shows that most dry deposition occurs on the eastern side of the LTB, where loads can be as high as 5.1 kg N ha^{-1} summer^{-1} (Figure 6). In their 2001 paper, Coats and Goldman show that in spite of these elevated atmospheric fluxes, the streams in question release very small amounts of inorganic N into Lake Tahoe, a feature characteristic of extremely N limited, high elevation environments (Perakis, 2002). Historically, wet deposition is relatively small on this side of the Basin due to rain shadow effects that reduce precipitation by half (1.7 vs. 2.9 kg N ha^{-1} yr^{-1}) in east-side watersheds (Jassby and others, 1994; Johnson and others, 1997; Reuter and Miller, 2000). Thus, any new anthropogenic dry season N fluxes would substantially increase the total amount of atmospheric N received by east-side watersheds, especially in the areas near to the SOLA site (Figure 6).

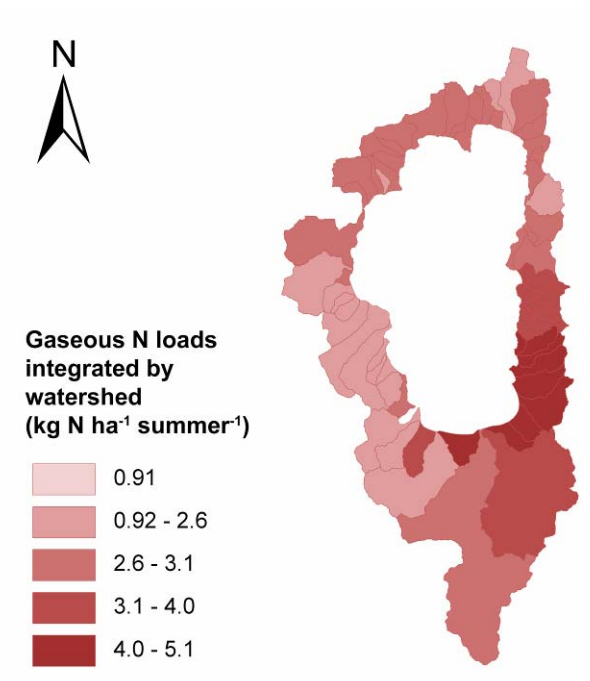

Figure 6. Gaseous N loads, integrated by watershed, in the Lake Tahoe Basin for the (July-September 2000)

Potential ecosystem effects

Our calculations indicate that if these areas were receiving low wet deposition amounts (i.e., 1.7 kg N ha^{-1} yr^{-1}) and high dry deposition (5.1 kg N ha^{-1} summer^{-1}), up to 75% of their atmospheric N load would be derived from dry deposition of gaseous N pollutants. The differential effects of dry N deposition along different physico-chemical pathways may be significant. On one hand, NH_3 and NO_2, which provide much of the atmospheric N in the urban impacted sub-watersheds (Figure 7b), are absorbed by plants directly through leaf stomata (Taylor and Constable, 1994; Stulen and others, 1998). This constitutes a direct foliar source of N for plants, which may then take up less N through their roots (Grulke and others, 1998). At the ecosystem scale, such changes in nutrient availability may affect ecosystem structure by favoring nitrophilic invasives or by allowing less root vs. microbial N uptake (Fenn and others, 2003a). The fact that these pollutants deposit primarily during the summer growing season may enhance this effect.

HNO_3 may also provide some N via the foliar pathway (Krywult and Bytnerowicz, 1997; Calanni and others, 1999), but the majority of the deposited HNO_3 will adsorb to the outside of the leaf, where much of the deposited N can be easily washed off (Marshall and Cadle, 1989; Janson and Granat, 1999; Wesely and Hicks, 2000). The labile nitrate from this HNO_3 deposition may more directly affect aquatic systems as a result. Additions of inorganic N have also been shown to substantially change soil chemistry and the resulting production of dissolved organic nitrogen (DON) in some forest ecosystems (Marcus and others, 1998; McDowell and others, 2004; Pregitzer and others, 2004).

Given (1) unknown contribution of DON to eutrophication in the LTB (Fenn and others, 2003a), (2) the potential for urbanization to change the bio-availability of DON exported from forest watersheds (Seitzinger and Sanders, 1997; Seitzinger and others, 2002), and (3) the continuing alteration of litter N content resulting from fire suppression (Johnson, 1992; Johnson and others, 2001), we believe that the specific linkages between atmospheric N deposition, DON accumulation, and DON export in the LTB may warrant further investigation.

CONCLUSIONS

As Raupach and Finnegan (1988) note, bigleaf (or single layer inferential) models "are incorrect but [sometimes] useful." In this exercise, we use the bigleaf approach in a GIS framework to assess how dry N deposition varies across the different sub-watersheds within the LTB. We find that the model results, integrated over the entire LTB area, closely match previous estimates, and are not necessarily high enough to cause concerns about ecological effects based on threshold established in other high elevation watersheds. However, the modeled deposition at more local scales (30 m^2) can be much higher, up to 7.5 kg-N ha^{-1} for the 3-month summer period. Integrated over the area of the individual sub-watersheds of the LTB, the modeled fluxes are lower (up to 5 kg-N ha^{-1} summer^{-1}), but still high enough that they might exceed thresholds for causing ecological effects, which may include aquatic eutrophication, changes in soil chemistry, and/or changes in plant community structure.

Our spatial analysis shows that these effects are mostly likely to occur in watersheds near South Lake Tahoe and on the drier east side of the LTB, where N loads during the period examined comprised up to 75% of total deposition (assuming 1.7 kg-N ha^{-1} yr^{-1} of wet deposition and 5.1 kg-N ha^{-1} yr^{-1} dry deposition). Given that these inputs occur during the plant and phytoplankton growing season, and given that the modeling techniques used are likely conservative (if spatially uncertain), we conclude that further research looking for the linkages between dry N deposition and eutrophication of aquatic and terrestrial systems in the LTB is warranted.

ACKNOWLEDGEMENTS

The authors gratefully acknowledge the support of the National Science Foundation (Nevada EPSCoR program) and the Desert Research Institute and thank Mary Miller and James Sutherland for their help in the gathering and processing of samples. The input and expertise of Tim Minor and Julide Koracin were also greatly appreciated.

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