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# Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States

L.H. Pardo, M.J. Robin-Abbott, C.T. Driscoll, editors



## Abstract

Human activity in the last century has led to a substantial increase in nitrogen (N) emissions and deposition. This N deposition has reached a level that has caused or is likely to cause alterations to the structure and function of many ecosystems across the United States. One approach for quantifying the level of pollution that would be harmful to ecosystems is the critical loads approach. The critical load is defined as the level of a pollutant below which no detrimental ecological effect occurs over the long term according to present knowledge.

The objective of this project was to synthesize current research relating atmospheric N deposition to effects on terrestrial and aquatic ecosystems in the United States and to identify empirical critical loads for atmospheric N deposition. The receptors that we evaluated included freshwater diatoms, mycorrhizal fungi and other soil microbes, lichens, herbaceous plants, shrubs, and trees. The main responses reported fell into two categories: (1) biogeochemical, and (2) individual species, population, and community responses.

The range of critical loads for nutrient N reported for U.S. ecoregions, inland surface waters, and freshwater wetlands is 1 to 39 kg N ha<sup>-1</sup> y<sup>-1</sup>. This broad range spans the range of N deposition observed over most of the country. The empirical critical loads for N tend to increase in the following sequence for different life forms: diatoms, lichens and bryophytes, mycorrhizal fungi, herbaceous plants and shrubs, trees.

The critical loads approach is an ecosystem assessment tool with great potential to simplify complex scientific information and effectively communicate with the policy community and the public. This synthesis represents the first comprehensive assessment of empirical critical loads of N for ecoregions across the United States.

## Cover Photos

Front, left to right, top: elevated nitrogen inputs to a prairie grassland in Minnesota (control) resulted in a decrease in species richness and an increase in invasive grasses (N addition). Photos by David Tilman, University of Minnesota, used with permission.

Front, left to right, bottom: elevated nitrogen inputs to a high elevation spruce fir forest (control) in Vermont resulted in decreased growth and increased mortality (high treatment). Photos by Linda Pardo, U.S. Forest Service.

Back top: The endangered checkerspot butterfly (*Euphydryas editha bayensis*). Photo by Stuart Weiss, used with permission.

Back bottom: The threatened purple pitcher plant (*Sarracenia purpurea* L.). Photo by Lingli Liu, used with permission.

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# **Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States**

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# 12 NORTH AMERICAN DESERTS

E.B. Allen, L.H. Geiser

## 12.1 Ecoregion Description

North American deserts and semi-deserts extend from British Columbia to Mexico (CEC 1997). The ecoregion description is adapted from CEC (1997). They are characterized by aridity (130 to 380 mm annual precipitation); shrub and succulent vegetation with trees in riparian areas and higher elevation woodlands; and dry, low organic matter soils that are high in calcium carbonate (CaCO<sub>3</sub>) in some regions. The ecoregion can be subdivided into cold northern deserts and semi-deserts (Columbia Plateau, Snake River Basin, Great Basin, greater Colorado Plateau) and warm southern deserts (Mojave, Sonoran, Chihuahuan) (see Table 12.1).

The cold deserts and semi-deserts are arid to semi-arid with marked seasonal temperature extremes, a winter moisture regime and some snow. They lie in the rain shadow of the Cascade and Sierra Mountains to the west, and are blocked from moist Gulf Coast air masses by the Rocky Mountains to the east. The Columbia/Snake River Plateaus are loess- and volcanic ash-covered plains; their expansive grassland and sagebrush steppes have been largely converted to agriculture. The Great Basin contains hundreds of north-south mountain

ranges separated by broad valleys, whereas the Colorado and Arizona/New Mexico plateaus consist of canyons, cliffs, buttes, and mesas of sedimentary rock. Sagebrush (*Artemisia* spp.) dominates these deserts, with saltbush (*Atriplex* spp.) and greasewood (*Sarcobatus* spp.) on more alkaline soils.

The warm deserts have higher temperatures and greater evaporation rates. The Mojave is the driest and hottest of the North American deserts; its recorded high temperature, in Death Valley, California, is 57 °C. Characteristic plants are the creosote bush (*Larrea tridentata*) and Joshua tree (*Yucca brevifolia*). The Sonoran Desert of Arizona and California is subtropical and receives most of its precipitation in the summer monsoon season; gentler rains may occur in the winter months. Its structurally diverse vegetation includes the paloverde tree (*Parkinsonia* spp.), saguaro cactus (*Cereus gigantea*), cholla (*Cylindropuntia* spp.), and agave (*Agave* spp.). The Chihuahuan Desert extends into southern New Mexico from Mexico. Rain occurs primarily in the summer, and supports smaller scrub vegetation characterized by American tarwort (tarbush; *Flourensia cernua*), creosote bush (*Larrea tridentata*),

**Table 12.1—Location of North American Deserts of the United States.**

Level 2 Ecoregion <sup>a</sup>	Desert	Location	Level 3 Ecoregion <sup>a</sup>
Cold deserts	Columbia Plateau	Eastern Washington	Columbia Plateau
	Snake River Plain	Southern Idaho	Snake River Plain
	Great Basin	Southern Oregon, extreme southern Idaho, north and central Nevada, western Utah	Northern Basin and Range, Central Basin and Range
	Greater Colorado Plateau	Western Wyoming, western Colorado, eastern Utah, northern Arizona, northern New Mexico	Wyoming Basin, Colorado Plateau, Arizona/New Mexico Plateau
Warm deserts	Mojave	Southeastern California, southern Nevada, northeastern Arizona	Mojave Basin and Range
	Sonoran	Southwestern Arizona, extreme southern California	Sonoran Basin and Range
	Chihuahuan	Southern New Mexico	Chihuahuan Desert

<sup>a</sup>CEC 1997

soaptree yucca (*Yucca elata*), grama grasses (*Bouteloua* spp.), and tobosagrass (*Pleuraphis mutica*). Piñon-juniper woodlands occur at the higher elevations above both cold and warm deserts.

Small mammals, birds, and reptiles are the most abundant wildlife of the ecoregion. Livestock grazing is widespread and, together with irrigated agriculture and population growth, has had significant ecological and hydrological effects. Major urban areas with higher nitrogen (N) deposition, which affects the adjacent and downwind deserts, are Phoenix, Salt Lake City, Las Vegas, Tucson, Albuquerque, Spokane, and Los Angeles (Fenn et al. 2003).

## 12.2 Ecosystem Responses to N Deposition

Responses to N deposition in deserts include alterations of species composition, often increases in biomass of exotic species, and decreases in native species. In this ecoregion of altitudinal, latitudinal, and landform diversity, vegetation types vary, but low growing shrubs and grasses predominate. Recently, invasive exotic species have been increasing in desert ecosystems (Allen et al. 2009, Brooks 2003). This may be due to increased N deposition, as exotic grasses may respond to higher soil N levels differently than native species (Allen et al. 2009). Such changes can have serious management implications. For example, in the agriculturally intensive Snake River Plain and in the Great Basin, extensive cheatgrass (*Bromus tectorum*) invasions contribute to increased fire frequency, that in turn favors even greater cover of cheatgrass. Loss of native plants adapted to longer fire intervals and the poorer nutritional quality of cheatgrass in turn reduces the carrying capacity of lands (Whisenant 1990). Fire frequency has similarly increased in the Mojave Desert in areas undergoing exotic grass invasions (Brooks et al. 2004, Brooks and Matchett 2006). Experimental N fertilization resulted in a depletion of soil moisture in sagebrush steppe, which could potentially affect future plant production (Inouye 2006).

Biotic soil crusts (consisting of bacteria, green algae, microfungi, and crustose lichens) and mycorrhizal fungi constitute important parts of desert ecosystems.

Responses to increased N deposition include reductions in photosynthesis and nitrogen fixation rates; reductions in photosynthetic pigments and ultraviolet (UV) protective pigments such as scytonemin; increases in electrolyte leakage, increases in microbial N and denitrification; shifts in community composition favoring eutrophs; and increases in microbial activity in nonvegetated soils.

## 12.3 Range of Responses Observed

Several studies have used N fertilizer in the desert to measure a biomass response and then compared this response to N inputs along N deposition gradients (Allen et al. 2009, Baez et al. 2007). Responses to N inputs are summarized in Table 12.2. A response to a certain level of N fertilization is assumed to be similar to the response along the gradient, assuming that factors besides N inputs are equal. The difficulty with this approach is that gradients may have multiple factors that change among sites in addition to N inputs, making interpretation of results difficult. The combination of N fertilization plus gradient approaches provide the most reliable information currently available. The most useful N fertilization studies for setting critical loads are those in the range of threshold responses, as reported below. For instance, N fertilizer of 72 kg ha<sup>-1</sup> yr<sup>-1</sup> at the Jornada Long-Term Ecological Research (LTER) site in New Mexico showed growth increases by native winter annual forbs and decreases by summer annual forbs (Gutierrez and Whitford 1987), but this value is well above the response threshold determined from studies reviewed below. Similarly, 60 kg N ha<sup>-1</sup> yr<sup>-1</sup> increased mineralization rates at the Central Arizona Project (CAP-LTER; Hall et al. 2009). Alternatively, input of 40 kg N ha<sup>-1</sup> yr<sup>-1</sup> over 3 years did not increase creosote bush biomass at the Mojave Global Change Facility near Las Vegas (Barker et al. 2006) and the authors concluded that annual variability in precipitation was the most important variable for creosote bush productivity.

### 12.3.1 Plant Community Responses

Baez et al. (2007) used an increase in the biomass of the dominant native blue grama grass (*Bouteloua gracilis*) as an indicator of elevated N in desert grassland at the

**Table 12.2—Responses to N input for North American Deserts ecoregion. Reliability rating: ## reliable; # fairly reliable; (#) expert judgment**

Site	Response threshold <i>kg N ha<sup>-1</sup> yr<sup>-1</sup></i>	Reliability	Response	Comments	Study
Riparian habitat, Hells Canyon NRA, Columbia Plateau	3	(#)	High cover of eutrophic lichens, enhanced N concentrations in lichen thalli	Uncertainty regarding modeled N estimate	Geiser et al. 2008, Porter et al 2007
Piñon-juniper and creosote scrub, Joshua Tree National Park, Sonoran and Mojave Desert	8.4	#	Increased biomass of invasive grasses	N fertilization of 5 kg ha-1 y-1 plus background N deposition of 3 kg ha-1 y-1 in wet year	Allen et al. 2009
Creosote scrub, Joshua Tree National Park, Sonoran Desert	3-8	#	Production of fine fuel of Mediterranean splitgrass sufficient for fire	DayCent model	Rao et al. 2010
Piñon-juniper Joshua Tree National Park, Mojave Desert	3-6	#	Production of fine fuel of red brome sufficient for fire	DayCent model	Rao et al. 2010
Desert grassland Sevilleta LTER	2.5-22	#	Increase in blue grama biomass	Response observed at 20 kg N ha-1 y-1 fertilization; ambient is 2.45 kg N ha-1 y-1	Baez et al. 2007
Sagebrush steppe	7.4	#	Depletion of soil moisture	No increase in aboveground production	Inouye 2006
Desert grassland, Colorado Plateau	22	Not reliable: fertilization level too high	Increase in tiller density of the dominant cool season grass species Indian ricegrass	No increase in warm season galleta grass. Fertilizer addition of 20 kg N ha-1 y-1; ambient deposition 2 kg N ha-1 y-1	Schwinning et al. 2005
Desert scrub, Mohave Desert	32	Not reliable: fertilization level too high	Increase in biomass of the invasive species Russian thistle	Short term—2 yrs fertilization	Brooks et al. 2003
		Not reliable: fertilization level too high	Increase in exotic invasive grasses and a decline in native forbs	Creosote bush dominated ecosystem	

Sevilleta LTER site in New Mexico. Plots fertilized with 20 kg N ha<sup>-1</sup> for one season had increased biomass of blue grama compared to controls. Bulk N deposition increased from 1.7 to 2.4 kg ha<sup>-1</sup> yr<sup>-1</sup> in the control plots between 1989 to 2004, adding an additional total of 5.9 kg N ha<sup>-1</sup> to the control plots from increased N deposition during these 16 years. However, there was no change in biomass of blue grama over this period, suggesting that 5.9 kg N ha<sup>-1</sup> over 16 years was below the threshold for increased biomass in this species. Thus, Baez et al. (2007) concluded that the threshold for elevated biomass of blue grama lies somewhere between 2.45 and 20 kg N ha<sup>-1</sup> yr<sup>-1</sup>; including ambient deposition raises the upper response threshold to approximately 22 kg N ha<sup>-1</sup> yr<sup>-1</sup>.

Zhu et al. (2006) calculated 72.3 kg ha<sup>-1</sup> N content in urban Phoenix soils compared to exurban desert soils that had 9.4 kg N ha<sup>-1</sup>, accompanied by elevated carbon (C) in urban soils. This increase in soil N and C was hypothesized to have occurred during the last several decades of rapid urban expansion. A more recent study also showed elevated N and C in remnant native vegetation fragments within the urban Phoenix CAP-LTER site that promoted changes in N cycling, such as increased denitrification potential and higher microbial N (Hall et al. 2009). Nitrogen deposition in the Phoenix area was measured using wet and dry bucket collectors. The highest measured values were less than 6 kg N ha<sup>-1</sup> yr<sup>-1</sup>, which the investigators describe as an underestimate because the dry bucket technique is not accurate (Lohse et al. 2008). The Community Multiscale Air Quality (CMAQ) model, which includes both wet and dry deposition, showed a maximum deposition of 26 kg N ha<sup>-1</sup> yr<sup>-1</sup> for the Phoenix area (Fenn et al. 2003). Impacts of N deposition on vegetation were not reported in these studies in the Phoenix area, although the PALS (Patch Arid Land Simulator) model showed nearly a doubling in total plant production with simulated N addition between 2 and 26 kg ha<sup>-1</sup> yr<sup>-1</sup> (Shen et al. 2008). The model shows that even a small increase in N can increase biomass, but it is not clear how this can be used to determine a response threshold. However, measurements comparing urban to rural areas show no significant change in plant biomass due to elevated N, likely because of multivariate gradients that

include elevated ozone (O<sub>3</sub>), temperature, and other factors.<sup>14</sup> Additional research on responses to fertilizer N are under way at the CAP-LTER site.

Schwinning et al. (2005) used fertilizer levels of 0, 10, 20, and 40 kg N ha<sup>-1</sup> for 2 years in a desert grassland on the Colorado Plateau and measured an increase in tiller (grass shoot) density of the dominant cool season grass species, Indian ricegrass (*Oryzopsis hymenoides*), at 20 kg N ha<sup>-1</sup> yr<sup>-1</sup>. This increase was not observed in the warm season grass species, galleta grass (*Hilaria jamesii*). The most noticeable change in this study was an increase in biomass of the invasive species Russian thistle (*Salsola iberica*) at the highest N level. The authors concluded that elevated N deposition will promote an invasion of Russian thistle and a shift in dominance to cool season grasses. However, current N deposition at this site may be as low as 2 kg ha<sup>-1</sup> yr<sup>-1</sup>. Because the study was carried out for only 2 years, the levels of N inputs may be higher than threshold response levels for long-term N deposition.

Working in creosote bush desert scrub, Brooks (2003) fertilized at a rate of 32 kg N ha<sup>-1</sup> yr<sup>-1</sup> for 2 years and found an increase in exotic invasive grasses and a decline in native forbs. Allen et al. (2009) studied the impacts of elevated N by fertilization along an N deposition gradient in Joshua Tree National Park in creosote bush scrub and piñon-juniper woodland. The N deposition rates along the gradient ranged from 3.4 to 12.4 kg N ha<sup>-1</sup> yr<sup>-1</sup> as measured by bulk deposition samplers and also as modeled (Tonnesen et al. 2007). Fertilization rates were 5 and 30 kg N ha<sup>-1</sup> between 2003 and 2005. There were no vegetation biomass responses in 2003, a dry year. In 2004, a year with moderate precipitation, two of the four sites had increased invasive grass biomass at the treatment of 30 kg N ha<sup>-1</sup>, but not with 5 kg N ha<sup>-1</sup>. The strongest response to N occurred in the wettest year, 2005, when invasive grass biomass increased under both 5 and 30 kg N ha<sup>-1</sup> fertilizer in the lowest and highest N deposition sites. Conversely, native

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<sup>14</sup>Hall, S.J. Unpublished data comparing plant response to N deposition in urban and rural areas. On file with: Department of Ecology, Evolution, and Environmental Science, School of Life Sciences, Arizona State University, Tempe, AZ 85287-4701

forbs decreased in production at these sites following fertilization. However, another site with low N deposition and sparse invasive grass cover had a positive response by native forbs to N fertilization, suggesting that native forbs can respond to N in the absence of grasses (Allen et al. 2009).

To determine the N input level that results in vegetation change, the biogeochemical model daily Century (DayCent; Rao et al. 2010) was used to model invasive grass and forb productivity under varying levels of N deposition, soil texture, and precipitation between 2003 and 2008 (Rao et al. 2009, 2010). Fires have become more frequent in the Mojave and Sonoran Deserts, and there may be a relationship between fire and increased grass biomass under enhanced N deposition (Allen et al. 2009, Brooks 2003, Brooks and Matchett 2006). Fire risk, which was calculated as the probability of producing enough fine fuels to carry a fire ( $1 \text{ T ha}^{-1}$  fine fuel), increased exponentially when deposition was above  $3.0 \text{ kg N ha}^{-1}\text{yr}^{-1}$ ; fire risk leveled at  $5.7 \text{ kg N ha}^{-1}\text{yr}^{-1}$  in piñon-juniper woodland, and at  $8.2 \text{ kg N ha}^{-1}\text{yr}^{-1}$  in creosote bush scrub (Rao et al. 2010).

Elevated N may also increase soil water use by plants, as reported by Inouye (2006) for big sagebrush (*Artemisia tridentata*) fertilized at 6 and  $12 \text{ kg N ha}^{-1}\text{yr}^{-1}$  for six years in Great Basin semi-desert in southern Idaho. Soil moisture was depleted to 140 cm and was not recharged except in the wettest years. No aboveground increase in productivity was measured, so the moisture depletion may have been due to increased root productivity. The site receives  $1.4 \text{ kg N ha}^{-1}\text{yr}^{-1}$  wet deposition as measured at a nearby NADP (National Atmospheric Deposition Program) station, but dry deposition is not known. Since there was no significant difference between the two fertilizer levels, the lower level of  $6 \text{ kg N ha}^{-1}\text{yr}^{-1}$  (plus the background  $1.4 \text{ kg N ha}^{-1}\text{yr}^{-1}$  wet deposition) can be considered the response threshold for moisture depletion by sagebrush. Schwinning et al. (2005) observed an earlier onset of drought stress and increased water use with  $20 \text{ kg N ha}^{-1}\text{yr}^{-1}$  of fertilizer plus the estimated background deposition of  $2 \text{ kg N ha}^{-1}\text{yr}^{-1}$ , confirming the observations that elevated N can promote increased water use in desert vegetation.

### 12.3.2 Arbuscular Mycorrhizal Fungi

Mycorrhizal fungi are important for plant nutrient acquisition, drought stress tolerance, and the hydraulic lifting of water, which are critical in arid lands (Allen 2007). Their diversity, abundance, and functioning are altered by N deposition. A Phoenix-area study showed shifts in mycorrhizal species composition in urban and exurban sites that were dominated by different plant species, but did not relate these species shifts to N deposition (Bills and Stutz 2009). Data from the Sevilleta LTER site, which is located at the junction of cold desert, warm desert, and shortgrass steppe, suggest that long-term N fertilization at  $100 \text{ kg ha}^{-1}\text{yr}^{-1}$  can lead to changes in the community structure and function of arbuscular mycorrhizal fungi associated with grasses from semi-arid environments (Corkidi et al. 2002, Egerton-Warburton et al. 2007). However, this level of N fertilization is well beyond the critical load, greatly exceeds current deposition amounts (wet plus dry), and exceeds the threshold for changes in mycorrhizal functioning observed in Mediterranean California ecosystems. Results of mycorrhizal response to N deposition in the Great Plains and Mediterranean ecoregions (Chapters 11 and 13) also have relevance for the more mesic end of the cold and warm desert ecosystem types.

### 12.3.3 Lichens, Biotic Soil Crusts, and Soil Microbes

Lichens make a significant contribution to biological diversity in North American deserts and semi-deserts; a recent compendium documented 1971 species from the greater Sonoran Desert alone (Nash et al. 2002, 2004, 2008). However, the long periods of metabolic inactivity due to insufficient hydration, dominance of communities by crustose forms with minimal surface exposure, minimal dry deposition inputs from canopy drip, and lower growth rates, can make it difficult to detect community-level shifts. Even after a decade of exposure, Marsh and Nash (1979) found no evidence of community effects on lichens in the vicinity of the large Four Corners Power Plant in the Colorado Plateau. Nevertheless, physiological measurements indicate that wetted desert lichens can be as sensitive to air pollution as lichens from other climates. Lichens in the vicinity of a coal-fired power plant in Page, Arizona, a major point



source of sulfur dioxide (SO<sub>2</sub>), nitrogen oxides (NO<sub>x</sub>), and metals, exhibited increased electrolyte leakage, chlorophyll degradation, and reduced N fixation (Belnap 1991).

Enhanced concentrations of N in thalli of the saxicolous lichen, *Xanthoparmelia cumberlandia*, and high cover of eutrophic lichens on netleaf hackberry (*Celtis laeviagata* var. *reticulata*), a small tree of riparian areas, were observed throughout Hells Canyon National Recreation Area (Geiser et al. 2008). Interagency Modeling of Protected Visual Elements (IMPROVE) data show that Hells Canyon is a regional hot spot for ammonium (NH<sub>4</sub><sup>+</sup>) and nitrate (NO<sub>3</sub><sup>-</sup>) deposition from the Snake River valley and possibly as far away as California. Other pollutants—SO<sub>2</sub>, NO<sub>x</sub>, O<sub>3</sub>, hydrogen sulfide (H<sub>2</sub>S), and fine particulate ammonium sulfate ((NH<sub>4</sub>)<sub>2</sub>SO<sub>4</sub>)—were low and comparable to other remote areas. Mean annual deposition of total N from CMAQ in this area between 1990 and 1999 was 2.6 to 3.0 kg N ha<sup>-1</sup> yr<sup>-1</sup> (Porter 2007), with some uncertainty surrounding the reliability of the deposition estimate. Significantly higher thallus N concentrations and eutroph cover occurred within 0.2 km of the Snake River. Ammonia (NH<sub>3</sub>) emissions were especially high at these sites in spring and summer (5 to 19 ppb biweekly averages), presumably volatilizing from the fertilizer-laden Snake River. Average annual NH<sub>3</sub> concentrations at five year-long monitoring sites along the Snake River ranged from 1.4 to 4.7 ppb. In a review of European studies, Cape et al. (2009) suggested a long-term mean annual critical level for ammonia of 1 µg m<sup>-3</sup> (1.4 ppb), based on lichen and bryophyte responses; local data support that estimate.

Biotic soil crusts are well developed in cold deserts and, in pristine areas, can account for as much surface cover as vascular plants (Davidson et al. 2002). Consisting of cyanobacteria-dominated mats of intricately enmeshed bacteria, green algae, microfungi, and crustose lichens, they play vital ecological roles by improving topsoil stabilization, soil fertility, surface water holding capacity, and seedling survival (Harper and Belnap 2001). Biotic soil crusts are responsive to N deposition, but because component species are physically inextricable from each other, measurements reflect the response of the

combined biota, along with climate, soil chemistry, and other environmental variables. Some examples:

- 1) An aqueous solution of 0.5 g N L<sup>-1</sup> was added to biotic crusts from Canyon Lands National Park in the Colorado Plateau (Belnap et al. 2008). Reductions in quantum yield, b-carotene, nitrogenase activity, scytonemin pigments, and UV protective xanthophyll pigments due to N alone were observed in early- and mid- but not late-successional crusts. Higher temperatures, more UV, and more frequent precipitation had adverse effects on all crust types. In these cases, N positively moderated some adverse effects of UV, but only in the late successional crusts.
- 2) In a gradient study in the Sonoran Desert near Phoenix (McCracken et al. 2008), microbes of wetted soils between plants responded (increased carbon dioxide (CO<sub>2</sub>) flux) readily to N deposition, whereas microbes of wetted soils under plants did not. The authors postulated that anthropogenic N could essentially decouple microbes from reliance on shrubs for nutrients, resulting in increased activity in open spaces between plants. In other words, whereas native desert microbial activity is primarily confined to zones influenced by plant root exudates, if adequate water and anthropogenic N are available, microbial activity can occur in soils outside these zones. Major shifts in microbial metabolism, community composition, and biomass in extensive bare soil surface areas are implied effects.

## 12.4 Critical Loads Estimates

Fertilization studies in the Sevilleta grasslands of the Chihuahuan Desert (Baez et al. 2007) indicate a critical load greater than 2.45 and less than 22 kg N ha<sup>-1</sup> yr<sup>-1</sup>, based on alteration in species composition and biomass observed at 22 kg N ha<sup>-1</sup> yr<sup>-1</sup> after 1 year and no changes under ambient deposition (2.45 kg N ha<sup>-1</sup> yr<sup>-1</sup>). Fertilization studies by Schwinning et al. (2005) and Brooks (2003) were not used to set the

**Table 12.3—Empirical critical loads of nutrient N for North American Desert ecoregion.**  
**Reliability rating: ## reliable; # fairly reliable; (#) expert judgment**

Ecosystem component	Critical load for N deposition <i>kg N ha<sup>-1</sup> yr<sup>-1</sup></i>	Reliability	Response	Comments	Study
Lichens	3	(#)	Lichen community shifts, increase in thallus N concentration	Uncertainty regarding modeled N estimate	Geiser et al. 2008, Porter 2007
Shrubland, woodland, and desert grassland	3-8.4	#	Vegetation response, community change. Increased biomass of invasive grasses; decrease of native forbs		Allen et al. 2009, Inouye 2006, Rao et al. 2010

critical load, as the high levels of N inputs over both short-term experiments could not accurately determine the critical load for long-term N deposition. At Joshua Tree National Park in the Mojave Desert, Allen et al. (2009) found that vegetation response was determined by a combination of factors, including soil properties and precipitation. These experimental studies indicated that a critical load for invasive grass biomass response may be as low as 8.4 kg N ha<sup>-1</sup> yr<sup>-1</sup> (3.4 kg N ha<sup>-1</sup> yr<sup>-1</sup> deposited, plus 5 kg N ha<sup>-1</sup> yr<sup>-1</sup> fertilized), as measured in the cleanest site (Allen et al. 2009). Their results suggest a critical load of 8.4 kg N ha<sup>-1</sup> yr<sup>-1</sup> in wet years to prevent an increase of invasive grasses in the desert ecosystem (Allen et al. 2009). The results from the biogeochemical DayCent model compared well with the experimental fertilization studies, and suggested a critical load of 3.1 to 8.2 kg N ha<sup>-1</sup> yr<sup>-1</sup> for lower elevation desert dominated by the invasive common Mediterranean grass (*Schismus barbatus*), and a critical load of 3.0 to 5.7 kg N ha<sup>-1</sup> yr<sup>-1</sup> in higher elevations with red brome (*Bromus rubens*) as the invasive grass (Rao et al. 2010). The model shows that at levels higher than 5.7 to 8.1 kg N ha<sup>-1</sup> yr<sup>-1</sup> the fire risk is controlled by precipitation, i.e., these values exceed the critical load for N. The critical load was determined by the amount of N modeled to produce 1 T ha<sup>-1</sup> of fine fuel to carry a fire, as these are regions subject to increasing grass-fire cycle impacts (Brooks and Matchett 2006). The critical load for vegetation response to N deposition is set at 3.0 to 8.4 kg ha<sup>-1</sup> yr<sup>-1</sup>, based on the research by Allen et al. (2009) and Rao et al. (2010), and supported by evidence (Table 12.3) of soil moisture depletion in that range (Inouye 2006). This critical load is considered

fairly reliable; the empirical study (Allen et al. 2009) is supported by a biogeochemical model (Rao et al. 2010). Further research in various desert study locations would increase the reliability of this critical load for the North American Deserts ecoregion. No critical load specific to arbuscular mycorrhizal fungi can be estimated currently. A tentative estimate of 3.0 kg N ha<sup>-1</sup> yr<sup>-1</sup> based on data from Geiser et al. (2008) and Porter (2007) is suggested for lichen thallus N concentration and lichen community shifts in the Columbia Plateau. This value is considered expert judgment, as there was uncertainty associated with the deposition estimate.

## 12.5 Comparison to Critical Loads for Other Regions

No critical loads have been calculated for Europe, as desert regions do not occur in Europe. We are not aware of critical loads estimated for other continents with desert systems. The European critical level for lichen and bryophyte response to NH<sub>3</sub> (1.0 µg m<sup>-3</sup> or 1.4 ppb; Cape et al. 2009) appears to be consistent with increases in cover of eutrophic lichens and elevated N in lichen thalli observed in the Hells Canyon National Recreation Area in the Columbia Plateau, where annual average NH<sub>3</sub> was 1.4 to 4.7 ppb (Geiser et al. 2008).

## 12.6 Future Research Directions and Gaps in Data

More research is needed to determine the range of critical loads in the North American Deserts ecoregion. Finer resolution fertilization studies could more closely pinpoint critical loads, as could research along



N deposition gradients. In addition, current research does not encompass the full range of diversity in North American desert ecosystems or flora. Only the far western section of the Sonoran Desert ecosystem downwind of Los Angeles has been studied. This is a region that contains great biological diversity and which is also subject to high rates of increased N deposition around the urban area of Phoenix. In addition, future research should examine the effects of N deposition on ecosystem components with longer generation times, including tree and cactus species. Long-term observations are under way in some of the fertilizer studies cited (Allen et al 2009).

### 12.6.1 Lichens and biotic crusts

Establishing critical loads is challenging for epiphytic lichens, biotic crusts, and other cryptogamic communities of the North American Deserts compared to other ecoregions, due to slow growth rates, the complex ecology of biotic crusts, and interactions of N deposition with changing climate and soil chemistry. Further research is needed to link accurately measured or modeled estimates of deposition with physiological data for indicator species, threshold N concentrations, or with lichen community responses. Studies that measure physiological responses to fertilization may hold the most promise.

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