Arnold Schwarzenegger
Governor

IMPACTS OF NITROGEN DEPOSITION ON CALIFORNIA ECOSYSTEMS AND BIODIVERSITY

PIER FINAL PROJECT REPORT

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Preface

The Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

The PIER Program, managed by the California Energy Commission (Energy Commission), annually awards up to $62 million to conduct the most promising public interest energy research by partnering with Research, Development, and Demonstration (RD&D) organizations, including individuals, businesses, utilities, and public or private research institutions.
PIER funding efforts are focused on the following RD&D program areas:

- Buildings End-Use Energy Efficiency
- Energy-Related Environmental Research
- Energy Systems Integration
- Environmentally Preferred Advanced Generation
- Industrial/Agricultural/Water End-Use Energy Efficiency
- Renewable Energy Technologies

What follows is the final report for the contract Assessment of Nitrogen Deposition: Modeling and Habitat Assessment, contract number 500-99-013, Work Authorization 61, conducted by the Bren School of Environmental Science and Policy at the University of California Santa Barbara, and the Creekside Center for the Earth Observations. The report is entitled *Impacts of Nitrogen Deposition on California Ecosystems and Biodiversity*. This project contributes to the Energy-Related Environmental Research program.

For more information on the PIER Program, please visit the Energy Commission's website www.energy.ca.gov/pier/ or contract the Energy Commission at (916) 654-5164.

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Abstract

Recognized as a "biodiversity hotspot," California supports numerous endemic taxa with narrow ranges, and that diversity may be threatened by atmospheric nitrogen deposition. This California-wide risk screening included: (1) a 36 x 36 kilometer (km) map of total Nitrogen (N)-deposition for 2002, developed from the Community Multiscale Air Quality Model (CMAQ); (2) identification of sensitive habitats; (3) an overlay of the Forest Resource and Protection (FRAP) vegetation map; (4) an overlay of animal and plant species occurrence data from the California Natural Diversity Data Base (CNDDB); (5) an initial analysis of species life history and habitat; and (6) a discussion of relevance and guidance for assessments of power plant impacts. An area of 55,000 square kilometers (km^2) of California is exposed to more than 5 kilograms of N per hectare per year (kg-N ha^{-1} $year^{-1}$), and 10,000 km^2 are exposed to more than 10 kg-N ha^{-1} $year^{-1}$. Deposition hotspots include: Los Angeles-San Diego, the San Francisco Bay Area, the Central Valley, and the Sierra Nevada foothills. The major documented impact of N-deposition on California terrestrial biodiversity is to increase invasive annual grasses in low biomass ecosystems, resulting in species loss. Of 225 "threatened" and "endangered" plant taxa, 99 are exposed to an average > 5 kg-N ha^{-1} $year^{-1}$. Of 1022 "rare" plant taxa, 290 are exposed to > 5 kg-N ha^{-1} $year^{-1}$. Listed animal species follow similar patterns. This initial screening outlines potential impacts on California's biodiversity and provides targeted guidance for assessing the impacts of power plant and other sources of atmospheric N-deposition.

Keywords: nitrogen deposition, biodiversity, California, annual grasses, invasive species, deserts, grasslands, threatened and endangered species, eutrophication

Executive Summary

Introduction

Atmospheric nitrogen deposition alters the structure and function of terrestrial ecosystems, because nitrogen is often a primary limiting nutrient on overall productivity. These alterations can drive losses of biodiversity, as nitrophilous species increase in abundance and outcompete species adapted to more oligotrophic conditions. California is recognized as a "biodiversity hotspot," with a high fraction of endemic taxa with narrow ranges, and many of those taxa may be at risk from atmospheric nitrogen deposition.

Project Objectives

The California Energy Commission's Public Interest Energy Research (PIER) program funded a project to investigate the potential scope of nitrogen deposition (N-deposition) risks to biodiversity in California. This statewide risk screening includes the following elements: (1) identification of sensitive habitat types, as documented by literature and local expertise; (2) a 36 x 36 kilometer (km) map of total N-deposition for 2002, developed from the Community Multiscale Air Quality Model (CMAQ); (3) an overlay of a statewide Forest Resource and Protection (FRAP) vegetation map; (4) an overlay of animal and plant species occurrence data from the California Natural Diversity Data Base (CNDDB); (5) a compilation of life history and habitat requirements for each species; and (6) a discussion of relevance and guidance for assessments of power plant impacts over which the Energy Commission has regulatory authority.

Project Outcomes

The major documented impact of N-deposition on California terrestrial biodiversity is to increase growth and dominance of invasive annual grasses in low biomass ecosystems such as coastal sage scrub, serpentine grassland, and desert scrub. Lichen communities may be altered. Vernal pools and sand dunes are vulnerable to annual grass invasions that are likely enhanced by N-deposition. Oligotrophic mountain lakes are also vulnerable.

Conclusions

The CMAQ model indicates that an area of 55,000 square kilometers (km^2) (out of California's total area of 405,205 km^2) are exposed to more than 5 kilograms of N per hectare per year (kg-N ha^{-1} $year^{-1}$),[1] and 10,000 km^2 are exposed to more than 10 kg-N ha^{-1} $year^{-1}$. Deposition hotspots include the major urban areas (Los Angeles-San Diego, and the San Francisco Bay Area), agricultural areas of the Central Valley, and portions of the Sierra Nevada foothills. Exposure of 48 different FRAP vegetation types were calculated. For example, 800 km^2 out of a total 6300 km^2 of coastal sage scrub are exposed to more than 10 kg-N ha^{-1} $year^{-1}$, primarily in Southern California.

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[1] Throughout the discussion of N-deposition exposure, a benchmark of 5 kg-N ha^{-1} yr^{-1} is used. This benchmark does not imply that 5 kg-N ha^{-1} yr^{-1} is the critical load for negative impacts for all ecosystems—some may be more sensitive and some may be less sensitive. Data are presented so that any benchmark can be used.

In contrast, many high elevation (> 1500-meter) montane vegetation types are minimally exposed, because they are far from pollution sources, except for localized occurrences in mountains surrounding the Los Angeles Basin. Of 225 federal and state listed "threatened" and "endangered" plant taxa, 101 are exposed to an average greater than 5 kg-N ha^{-1} $year^{-1}$. Of an additional 1022 plant taxa listed as "rare," 288 are exposed to greater than 5 kg-N ha^{-1} $year^{-1}$. Many of these highly exposed taxa are associated with sensitive habitat types and are vulnerable to annual grass invasions. The CNDDB was not of sufficient resolution or completeness to support finer-scale regional analyses. This initial, broad-scale screening indicates that N-deposition poses large potential impacts on California's unique biodiversity.

Recommendations

1. Based on the review and broad-scale screening in this report, nitrogen deposition impacts on ecosystems and species are extensive in California, and should be considered in local environmental assessments.
2. The impacts of N-deposition on California ecosystems are generally cumulative. Establishing critical cumulative loads for particular ecosystems is a research priority.
3. Local environmental assessments should initially focus on low biomass, nutrient poor habitats and the rare species they support, but also consider more general impacts. The state-wide information in this report provides a start, but is not sufficient for local use.
4. Increased invasions by introduced annual grasses and other weeds are the major threat to consider in mitigation. Finding a balance between habitat acquisition, habitat management, and weed management that effectively mitigates the incremental impacts of new power plant sources is a key goal.
5. Establishing reliable bioindicators along N-deposition gradients, such as changes in lichen communities, plant nutrient balances, and degree of weed invasions, will provide better spatial resolution of ecosystem effects.
6. The complexity of N-deposition forces a transdisciplinary approach to any research program.

Benefits to California

Nitrogen deposition is a growing threat to the biodiversity of California. This report is the first statewide analysis of exposure of ecosystems and special-status species to N-deposition, and provides the basis for systematic assessment of threats to specific ecosystems, and development of mitigation and management techniques. Along with an accompanying report on modeling by Tonnesen and Wang, this report provides regulatory guidance for impact assessments of new power plants. The report will provide an impetus for additional research for better understanding this complex phenomenon.

1.0 Introduction

Atmospheric nitrogen deposition has been demonstrated to alter terrestrial and aquatic ecosystem function, structure, and composition in many parts of the world, including Europe, Eastern North America, and Western North America (Galloway, Aber et al. 2003). Emissions, deposition, and N-cycling are highly complex processes and pose many scientific and policy challenges. The major purpose of this report is to examine the known and potential impacts of N-deposition on the varied ecosystems and species in California, using biogeographic data and modeled N-deposition.

Nitrogenous air pollutants have many sources, including transportation, agriculture, industry, electricity generation, wildfire, and emissions from natural and semi-natural ecosystems. Electric power plants in California, primarily fired by natural gas, are major point sources of nitrogen oxides (NO_x) from combustion, and ammonia (NH_3) from selective catalytic reduction (SCR) units used to control NO_x emissions. The California Energy Commission (Energy Commission), in conjunction with other regulatory agencies, is responsible for assessment of environmental impacts from energy-related developments and activities, including siting of new power plants.

Biology staff at the Energy Commission analyzed potential impacts from nitrogen deposition on several power plant licensing cases (Table 1, California Energy Commission 2003, 2001a, 2001b, 1997a, 1997b). These power plants were located in areas where nitrogen deposition impacts to nitrogen-poor, sensitive plant communities are an issue. Such communities are often rare and support many of California's rare and endangered plant and animal species. It is expected that future siting cases may need to review the impact of a power plant emissions on nitrogen-saturated or nitrogen-limited ecosystems. Nitrogen saturation has several detrimental effects, including decreased plant function as a result of leached nutrients (e.g., calcium) from the soil; loss of fine root biomass; decreases in symbiotic mycorrhizal fungi; promotion of exotic invasive species; and, leaching losses of base cations and nitrate into surface waters and ground waters, which increases soil and surface water acidification.

Table 1. California power plant licensing cases

Name	County
Metcalf Energy Center	Santa Clara
Los Esteros Critical Energy Facility	Santa Clara
Gilroy Peaker Plant	Santa Clara
Pico (Donald Von Raesfeld)	Santa Clara
Otay Mesa	San Diego
Sutter	Sutter

The PIER program funded a project to address these issues. The scope of work specifies four broad tasks: (1) a critical review of various air quality models used to determine power plant emissions of nitrogen (nitrogen oxide (NO), nitrogen dioxide (NO_2), and NH_3) concentration, release rate, dispersion, and deposition at ground level; (2) a chemical analysis of power plant plume characteristics including reaction rate from gas to particulate; (3) an assessment of nitrogen-limited habitats that could be at higher risk from further nitrogen deposition, and (4) location of nitrogen-saturated soils/ecosystems in California. Generally, the Energy Commission is interested in assessing impacts to terrestrial ecosystems from nitrogen deposition during power plant commissioning and operation and understanding the validity, strengths and weaknesses of models used to determine this impact. Specifically, the interest is in the short-distance and long-distance nitrogen deposition impacts to nitrogen-limited habitats and species dependent upon those habitats.

The project was awarded to the University of California, Santa Barbara (UCSB) (Dr. Frank Davis P.I.) and the University of California, Riverside (UCR) (CE-CERT, Dr. Gail Tonnesen P.I). This report presents investigations by UCSB into the biotic impacts of N-deposition (topics 3 and 4). Modeling reviews and assessments (topics 1 and 2) are the subject of an accompanying report by the UCR group (Tonnesen and Wang forthcoming).

Apart from this introduction, this biotic impacts report consists of four sections. Section 2 contains a review of existing information and research on N-cycling and the effects of N-deposition on ecosystems in general and California ecosystems in particular. Section 3 describes the spatial distribution of total N-deposition in California at 36 x 36 kilometer (km) scale, using the Community Multiscale Air Quality model (CMAQ) , and the exposure of vegetation types from the Fire and Resource Assessment Program (FRAP) map. Section 4 describes the N-deposition exposure of plant and animal species from the California Natural Diversity Data Base (CNDDB), along with relevant habitat and life history information of those species with higher exposure. Section 5 provides a synthesis and recommendations for further research.

2.0 Review

This review of existing information and research on the effects of nitrogen deposition on sensitive habitats in California draws heavily from a number of edited volumes and review papers regarding multiple aspects of N-deposition (and air pollution in general) in ecosystems (Langran 1999; Bell and Treshow 2002; Bytnerowicz, Arbaugh, et al. 2003), and especially from recent review work of N-deposition and ecological effects in Western North America (Fenn, Baron et al. 2003; Fenn; Haeuber et al. 2003). Interested readers should consult those works for extensive bibliographies of primary research, as there are hundreds of scientific papers dealing with various aspects of N-deposition.

This review will describe key processes in the nitrogen cycle, N-limitations in California terrestrial and aquatic ecosystems, effects of chronic deposition on N-cycling, and mechanisms by which N-deposition can lead to impacts on sensitive species, including direct toxicity, changes in species composition, and enhancement of invasive species. Ecosystems and habitats that are known to be and suspected to be sensitive to N-deposition are listed and specific mechanisms are briefly discussed as background for the biogeographic screening of habitats and species.

2.1. The Nitrogen Cycle

A basic understanding of the nitrogen cycle is essential background for assessing N-deposition impacts on ecosystems. The intricacies of the N-cycle involve diverse plants, animals, fungi, and bacteria interacting in complex aboveground and belowground environments (Schlesinger 1997), and a full discussion is well beyond the scope of this review. Figure 1 outlines key elements of the N-cycle that are relevant to this review.

Nitrogen (N_2) is the most abundant gas in the atmosphere (78%), but the strong triple bond is difficult to break and the gas is relatively inert. Reactive N (N_r) that can be directly used by organisms includes oxidized and reduced inorganic N and numerous forms of organic N. Inputs of N_r to ecosystems include biological N-fixation and atmospheric deposition. Atmospheric N_2 is directly available only to plants with N-fixing symbiotic bacteria. N-fixing plants in California include the Fabaceae (legumes), several genera in the Rosaceae, the genus *Ceanothus* (Rhamnaceae), and alders (Betulaceae). N-fixing cyanolichens are prominent in many ecosystems. Free-living cyanobacteria such as *Nostoc* are present in most ecosystems, and can be abundant in cryptobiotic crusts in deserts. N-fixation can vary from < 1 kg-N ha^{-1} yr^{-1} in habitats that are poor in N-fixers to > 100 kg-N ha^{-1} yr^{-1} in stands of alders, and other N-fixing trees and shrubs.

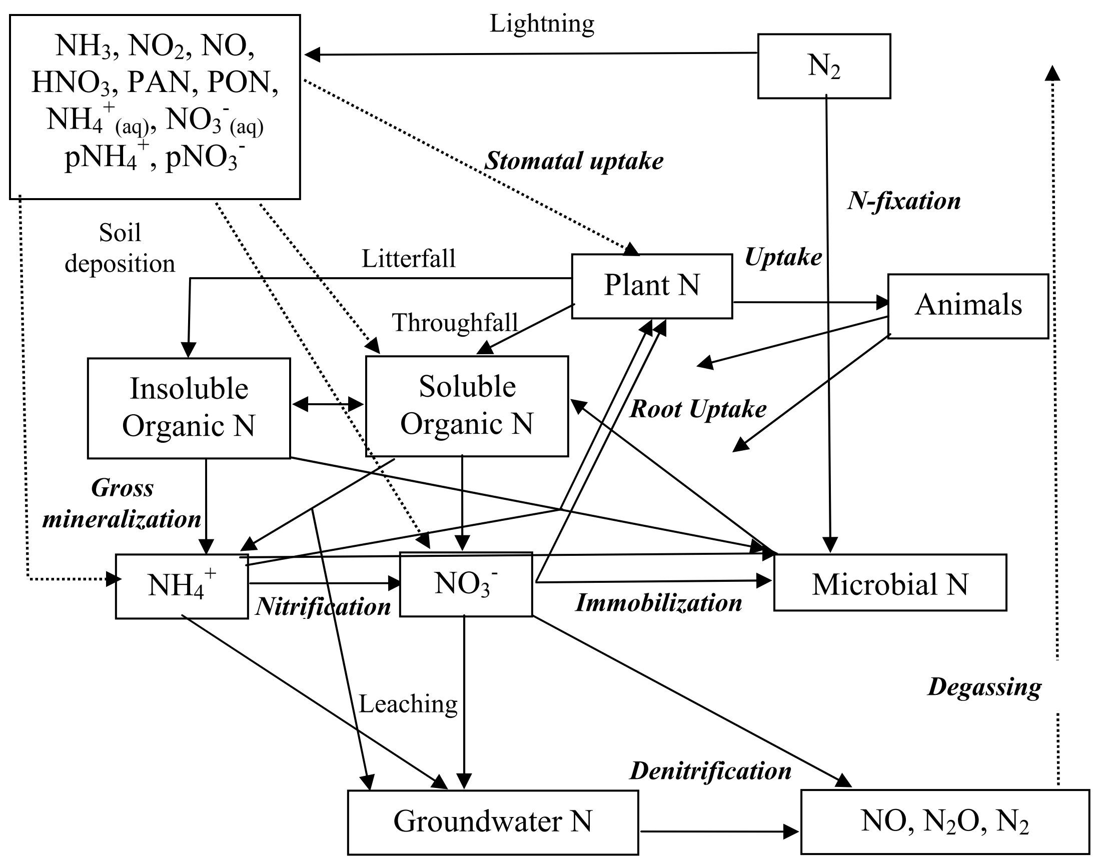


Figure 1. The N-cycle simplified. Biological processes are labeled in bold italics, and the lighter arrows show deposition pathways.

Natural background wet and dry atmospheric deposition originates from NO_x fixed by lightning, marine aerosols, N volatilized by fire, and N_r gases emitted from ecosystems. Large-scale combustion of fossil fuels, fertilizer applications, emissions from livestock, and other sources have greatly increased atmospheric deposition rates. Preindustrial atmospheric deposition in the western United States is estimated at 0.25 kg-N ha^{-1} yr^{-1}; elsewhere, approximate preindustrial background is ~1 kg-N ha^{-1} yr^{-1} (Fenn, Haeuber et al. 2003; Galloway, Aber et al. 2003). Very localized deposition originating from seabird colonies or other animal aggregations may be much higher, but those are exceptional situations. Atmospheric deposition enters ecosystems directly as wet deposition in precipitation and cloudwater, and as dry deposition to surfaces and through plant stomata. The significance of deposition pathways will be discussed below when considering the impacts of elevated deposition.

Most available N in terrestrial ecosystems is provided by decomposition of organic matter, known as *N-mineralization.* Most N is in the soil organic matter pool. Surface litter and larger woody debris decompose in a complex series of steps driven by a diverse array of detritovores (e.g., arthropods, nematodes, and other soil fauna), and ultimately by bacteria and fungi that mineralize organic nitrogen to ammonium (NH_4^+). While microbial biomass may be a small component of soil organic matter, microbial biomass is the key component through which a large portion of N is processed. The depolymerization of proteins into amino acids is a key step in N-availability, and amino acids may be taken up directly by microbes and plants—organic N in soils is difficult to study and relatively poorly understood (J. Schimel, pers. comm.). Turnover of fine roots also contributes to organic matter. Decomposition and mineralization rates generally increase with temperature, and show a hump-shaped relationship with moisture—slow in dry soils, faster up to an optimal moisture level, and slower in waterlogged soils. Either temperature or moisture may be seasonally limiting. The rate of litter decomposition, even under ideal temperature and moisture conditions, is affected by the litter carbon-to-nitrogen (C:N) ratio—high C:N litter generally decomposes more slowly than low C:N litter, although excess N in litter can slow decomposition as well. The coniferous and sclerophyllous evergreen species characteristic of many California ecosystems tend to produce high C:N litter, deciduous trees generally produce lower C:N litter. Many annual grasses produce lower C:N litter. Litter quality provides a major biogeochemical feedback and control over N-cycling, and mediates ecosystem response to increased atmospheric deposition.

The total amount of NH_4^+ released in decomposition is termed *gross mineralization.* Much of the gross mineralization is quickly immobilized as it is incorporated into microbial biomass. The remainder of potentially plant available NH_4^+ is referred to as *net mineralization.* Additions of readily available carbon (sugars, for example) can greatly increase immobilization rates and reduce net mineralization. NH_4^+ is readily adsorbed onto soil cation exchange sites, hence, it is relatively immobile and not prone to leaching. In high pH soils under dry conditions, NH_4^+ can be volatilized into NH_3 gas and lost to the atmosphere.

NH_4^+ is oxidized to nitrate (NO_3^-) by microbes in the process of nitrification. In coarse-textured soils in California, nitrification rates are relatively high and systems tend to be dominated by NO_3^- as opposed to NH_4^+. Nitrification rates are generally reduced by low pH, low O_2, very dry soils or very wet soils, and high litter C:N ratios, but exceptions are known especially under high N-deposition (de Boer and Kowalchuk 2001). NO_3^- is highly soluble in water, and subject to leaching below the root zone. Nitrification also leads to emissions of NO gas, which can be a significant pathway for N-loss back to the atmosphere. Small amounts of N_2O are also produced by nitrification. In most unfertilized ecosystems, N-leaching and NO emissions are minimal, indicating a relatively closed N-cycle. Nitrification provides another critical biogeochemical feedback and control over N-cycling.

Low instantaneous levels of soil NH_4^+ or NO_3^- do not necessarily indicate low N availability over the course of the growing season. Fluxes into and out of these mineral pools integrated over time are a much better indicator of soil N availability. In fact, extended high levels of mineral nitrogen, and leaching of NO_3^- in native ecosystems are symptoms of N-saturation. Similarly, low standing microbial biomass may mask rapid turnover. Measurement of mineralization, nitrification, and microbial dynamics in the field is a complex problem.

Plant roots take up both NO_3^- and NH_4^+ from soil solutions, some species prefer one to the other, but in general, even plants with a nitrogen form preference do better when both are available. Soils adjacent to roots are generally depleted of mineral N and other critical nutrients, indicating high uptake efficiency. NO_3^- is carried by mass flow of soil water to the near-root zone, which increases plant availability; conversely, plants may increase production of fine roots to seek out soil-bound NH_4^+. Cation and anion exchange processes at the root surface during N-uptake affect local soil chemistry.

Mycorrhizal fungi are symbiotic fungi that associate with plant roots and exchange mineral nutrients for plant-derived carbon. Although standing biomass of mycorrhizae may be low compared with plant biomass, the length of fungal filaments can be far greater than plant roots and contribute to N-uptake. Mycorrhizae are known to improve the nutrition of a majority of the macro- and micronutrients required for plant growth, including NH_4, NO_3, and organic N. Mycorrhizae can be sensitive indicators of N status (Egerton-Warburton and Allen 2000), and mutual feedbacks between fungus and plants can mediate ecosystem responses to N-deposition.

Increased N-availability in the soil (during the growing season) leads to either greater plant biomass production or higher tissue N-concentrations, depending on availability of water and other nutrients and the biochemical capabilities of the plants. Increased production and/or N-content leads to an acceleration of parts of the N-cycle (discussed below).

Live plants can emit NH_3 gas back to the atmosphere, especially under high soil N availability in fertilized pastures. Emissions of NH_3 in fertilized systems lead to complications in modeling NH_3 deposition. Plant tissue N (as well as litter) can be volatized through fire as NO_x, NH_3, and particulate-N. Herbivory may also have profound effects on rates of N-cycling. Animals feeding on plants can export N from the system, and redistribute it in relatively concentrated and labile forms. Herbivores are very sensitive to plant-N and selective herbivory can change plant species composition.

NO_3^- is denitrified into N_2O and N_2 under anaerobic conditions (wet soils or oxygen poor microsites). Denitrification is an important pathway for N loss in wetlands, surface water, and in groundwater. Denitrification in coarse, well-drained soils is relatively slow, but anaerobic microsites in soil particles provide some opportunities for denitrification. N_2O emissions are of concern as a greenhouse gas (GHG) and as a destroyer of stratospheric ozone. Denitrification and long-term geologic burial are the only pathways that remove N_r from the biosphere as a whole. Conditions that favor complete denitrification to N_2, with minimal production of N_2O, are the ideal objective of management aimed at removing N_r from ecosystems.

The N-cycle is under strong biotic control, and because of the multiple pathways, processes, and feedbacks that occur in site-specific combinations, it is difficult to generalize about it. Scientific understanding of the N-cycle at many scales is growing, but field measurement of many aspects of the N-cycle and the organisms that drive it continue to challenge ecosystem scientists.

2.2. N-limitations in California Terrestrial Ecosystems

California is recognized worldwide as a biodiversity hotspot, reflecting geographic isolation, strong regional and local climatic gradients, and geologic complexity (Bakker 1984). The mediterranean-type climate of cool wet winters and warm dry summers varies from the wet north to the dry south, from warm lowlands to frigid mountains, and from the maritime coastal zone to more continental inland regions—often over scales of a few kilometers. The complex and often violent geologic history of the state creates diverse edaphic conditions, ranging from shallow infertile serpentine soils and leached sands to deep fertile alluvial soils. California ecosystems span a broad range of physiognomic types, including the world's tallest high biomass evergreen forests, evergreen and deciduous forests, woodlands and shrublands, annual and perennial grasslands, deserts, and localized ecosystems specific to unique edaphic situations. Dramatically different vegetation types are often juxtaposed across abrupt topoclimatic and edaphic gradients, and fires create successional patchiness, creating rich local and regional vegetation mosaics. Aquatic ecosystems are diverse as well, ranging from oligotrophic mountain lakes, eutrophic lakes, seasonal lakes, freshwater and alkaline wetlands, mountain streams, large lowland rivers, and coastal marshes.

According to the Jepson Manual (Hickman 1993), California supports more than 5800 native plant species, of which 1169 are endemic to the California Floristic Province (the strongly mediterranean climate region of the West Coast). There are numerous localized endemic species, subspecies, and varieties that have minuscule ranges corresponding to special edaphic or climatic conditions. Geographic and botanical diversity also have produced a highly diverse fauna, again with many local endemic taxa. Many of these local endemics are listed as rare, threatened, and endangered by the U.S. Fish and Wildlife Service (USFWS) and California Department of Fish and Game (CDF&G) under their respective Endangered Species Acts. The California Native Plant Society (CNPS) maintains a list of rare, threatened, and endangered plants as well (CNPS 2003).

Urban and agricultural development pressures directly threaten habitats—few native species survive paving over and plowing under. Biological invasions, both plant and animal, pose one of the greatest threats to California's biodiversity. California ecosystems have been, and continue to be, heavily invaded by non-native plants—more than 1000 alien species have naturalized, and many have extensively and irrevocably altered millions of acres of California. Native grasslands, in particular, have been heavily altered by annual grasses and forbs from Eurasia, but few ecosystems have completely avoided invasions. Changes in plant composition affect animal communities, especially host-specific herbivores.

Water, temperature, and nutrients all can limit ecosystem productivity in California. The overall physiognomy and productivity of mature vegetation is largely determined by long-term site water balance and the effective length of the growing season. The length of the dry season is particularly important. However, given local water and temperature limitations, additions of nitrogen often produce immediate growth responses, indicating some degree of N-limitation. Phosphorous and other mineral nutrients are generally not limiting in the relatively young soils that dominate California, except in special soil types such as serpentine.

Under the mediterranean climate, seasonal patterns of N-availability, driven by decomposition, N-mineralization, and nitrification, are alternately limited by water and temperature. Most N-cycling occurs in shallow soil layers that contain the majority of organic matter. Soils are dry during the summer, wet with moderate temperatures following the first autumn/winter rainfall, wet but cool in the winter, and warm and wet only in the spring. Decomposition is slow for most of the year, and litter, especially coarse woody debris, tends to accumulate in the absence of fires. Fire is a key process in California ecosystems, and plays a critical role in driving N-deposition impacts (see below, Section 2.6).

Plant uptake and soil-N availability are often out of phase, and California ecosystems may be naturally "leaky," with some seasonal leaching of NO_3. N-mineralization and nitrification spike in autumn after the first soil wetting, but root uptake may lag behind until perennials develop new fine roots and annuals establish root systems. A pulse of NO_3^- can be flushed below the root zone or run off into surface water if early rains are sufficient to cause deep infiltration and runoff. Low plant uptake during the cool winter months can lead to NO_3^- leaching if sufficient rainfall occurs. In cold areas, deposited N accumulates in snowpack, with a large flush during melt. Flushes of NO_3^- following fires and other disturbances are important transient responses.

Specific evidence for N-limitations in a range of California terrestrial ecosystems are discussed in Section 2.4.

2.3. N-limitations in California Aquatic Ecosystems

Aquatic systems range from oligotrophic (i.e., nutrient-poor clear waters, such as Lake Tahoe) to mesotrophic to eutrophic (i.e., nutrient-rich waters with limited visibility, such as Clear Lake). Productivity in aquatic systems can be limited either by N or P, and phytoplankton communities are indicative of limiting nutrients. If N is limiting and P is relatively abundant, N-fixing phytoplankton (cyanobacteria) become more dominant. If P is limiting and N is abundant, then other phytoplankton taxa will dominate. If both N and P are abundant, some other nutrient (silica, for example, in the case of diatoms) may limit productivity. Both N and P enrichment can lead to algal blooms that can decrease water quality, and in extreme cases, decomposition of high algal biomass can deplete oxygen.

Many of the thousands of oligotrophic mountain lakes in the Western United States, including those in the Sierra Nevada, are naturally N-limited. NO_3^- is the major N species in montane lakes, and most N arrives as surface and subsurface flow into lakes and N-inputs depend strongly on the surrounding vegetation and soils. Lake Tahoe, an ultimate example of a naturally oligotrophic system, has changed from N-limitation to P limitation in recent decades (Jassby, Reuter et al. 1994).

Flowing waters are less susceptible to N-eutrophication, but can contain high levels of NO_3^-. NO_3^- is a criteria water quality pollutant. Intermittent streams often exhibit a flush of NO_3^- in high pollution areas, and long-term accumulation of N in watersheds can lead to high NO_3^- in baseflow originating from groundwater. Much N runoff in larger rivers in agricultural regions is associated with agricultural fertilization and livestock emissions, but elevated atmospheric deposition can also play a role.

Wetlands are susceptible to changes in structure and function under elevated N, and atmospheric deposition can encourage the spread of nitrophilous species (Morris 1991). Wetlands can act as filters, both capturing N in high productivity vegetation and in sediments, and perhaps more important, by denitrification in saturated soils (Morris 1991). The loss of riverine wetlands and floodplains greatly reduces basin-wide denitrification (Galloway, Aber et al. 2003).

Coastal bays and nearshore waters may also be N-limited—hypoxia and other water quality problems have been attributed to N-runoff on the East Coast and Gulf of Mexico. Extreme water quality problems in coastal California waters have generally been associated with large point sources, such as sewage outfalls and the mouths of urban creeks. However, recent work has indicated that seepage of polluted groundwater can contribute substantial nutrients to coastal waters (Boehm, Shellenbarger et al. 2004).

2.4. Effects of Chronic Deposition on N-cycling

The fate and impact of deposited N into ecosystems is driven by the response of plants and microbes to increased N-availability, and a series of biogeochemical feedbacks (Langran 1999). This section discusses general ecosystem responses to elevated N-deposition. Dry and wet deposition dynamics are complex and will only be briefly mentioned here, and models and algorithms are reviewed by Tonnesen et al. in an accompanying report (Tonnesen and Wang, forthcoming).

Dry deposition is modeled using atmospheric concentrations and deposition velocities. Deposition velocity is determined by aerodynamic, boundary-layer, and surface resistances (Metcalfe, Fowler et al. 1998). Aerodynamic resistance is driven by atmospheric turbulence, which is a function of surface roughness and wind velocity. There is greater turbulent transport over rougher surfaces, such as forests, than over smooth surfaces, such as grassland. Boundary layer resistance accounts for gaseous diffusion through the thin still layer of air surrounding all surfaces. Surface resistance accounts for the affinity of each particular gas species to different surfaces and moisture regimes. Of the major atmospheric N_r species, HNO_3, and NH_3 have the highest deposition velocities, because they are highly soluble in water, including thin films that remain on apparently dry surfaces. NO_2 is relatively insoluble in water and typically has deposition velocities an order of magnitude lower than HNO_3 and NH_3, and NO hardly dry deposits at all. Extensive reviews of atmospheric chemistry and deposition processes/modeling can be found in Metcalfe, Fowler et al. (1998) and Fowler (2002).

Atmospheric N-deposition enters ecosystems via deposition to plant and soil surfaces and via stomatal uptake into leaf interiors (Metcalfe, Fowler et al. 1998; Fowler 2002). Precipitation contains N_r in various oxidized and reduced forms. *Throughfall* (below canopy wet deposition) includes dry deposition on the surfaces of plant canopies that is washed into soils by precipitation and by fog drip (Collet, Daube, et al. 1990; Fenn, Poth, et al. 2000). Throughfall can also include inorganic and organic N leached from leaves. In California, dry deposition (especially of HNO_3) accumulates over the long summer droughts, and large pulses of accumulated N may be washed into soils with the first rains. Depending on the timing of winter rainfall, similar but smaller spikes of throughfall inputs may occur through the winter. Summer storms can also drive significant throughfall events. The combination of immediate deposition inputs with the initial pulse of mineralization and nitrification as soils are wetted produces a seasonal spike of high mineral N in the autumn. In coarse-textured California upland soils, NH_4^+ inputs—both as NH_3 gas and NH_4^+ particulates—are usually rapidly nitrified. However, the effective differences between reduced and oxidized N species in California are not well known. As mentioned above, NO_3^- leaching may occur following the substantial rainfall events—either summer thunderstorms or winter storms.

Stomatal uptake delivers N directly to the leaf interiors, and stomatal dynamics are essential to deposition models (Fowler 2002). The major deposition pathway for NO_2 is through stomata, as NO_2 is relatively insoluble in water and does not readily deposit to soils and foliage. Nitrogen dioxide is reduced to NH_4^+ in the leaves via nitrite reductase, and NH_4^+ is incorporated into amino acids. Ammonia is also rapidly deposited through stomata, although a high fraction may deposit on wet surfaces and on residual water films. Ammonia input into stomata is directly incorporated as NH_4^+ into amino acids. HNO_3 is also absorbed through stomata, and can also be transported through cuticles into leaf interiors (Marshall and Cadle 1989). Stomatal uptake can provide a substantial fraction of the N requirement of plants, but some plants may have difficulties assimilating NO_2—the ability of plants to tolerate NO_2 depends on antioxidants, nitrite reductase regulation, and other biochemical processes within leaves. Stomatal uptake of NO may not provide a large source of mineral N, but can affect metabolic processes—direct NO effects are an area of uncertainty (Mansfield 2002). NO levels generally decrease with distance from primary source, as it is rapidly oxidized to NO_2.

Once atmospheric N_r is deposited into ecosystems, it has cascading effects as it is assimilated, transformed, and recycled by organisms. The literature of N-fertilization in natural and agricultural systems is large. An extensive review of nitrogen addition experiments in arid, semiarid, and subhumid ecosystems indicates that aboveground net primary production (ANPP) is co-limited by N and water (Hooper and Johnson 1999). Nitrogen and water availability are tightly linked through biogeochemical feedbacks, including changes in litter quality and decomposition rates, microbial community dynamics, allocation patterns within plants, species composition, and other processes. The immediate effects of N and water additions are often additive in arid and semi-arid ecosystems.

Plant productivity typically exhibits a parabolic response to nutrient additions—at low levels, additions of nutrients increases growth, peaking at some intermediate level, and declining at higher levels. The typical immediate response to N-fertilization is a growth increase of existing plants, and such growth responses are taken as evidence of N-limitations. The direct uptake of atmospheric N_r also leads to growth increases in some species. Not all species are capable of large growth increases because of co-limitations from other nutrients or plant life history, architecture, and biochemistry. Plant tissue-N also increases, especially when other nutrients become more limiting; many plants take up available N in excess of demand. Nutrient imbalances can lead to changes in plant allocation, decomposition, herbivory, and other ecosystem processes.

Over longer time scales, increased productivity at the stand level is driven by changes in species composition, as nitrophilous species (adapted to high N conditions) outcompete other species by shading, root competition, selective herbivory, and other mechanisms. Species composition, through differences in foliage quality and phenology, affects N-cycling rates, which further affect species composition and feeds back into N-cycling. Changes in species composition have been extensively documented in Europe and elsewhere under long-term fertilization and N-deposition, and will be discussed below. Species composition changes also involve non-native invasive species, many of which respond strongly to N-fertilization. At ever higher levels of N-availability, productivity may decline as nutrient imbalances disrupt ecosystem processes

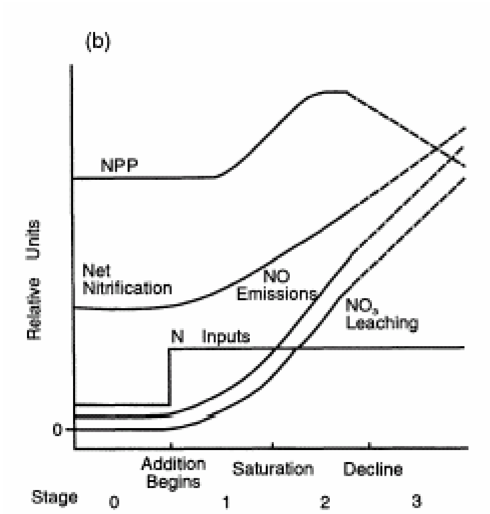
N-deposition can also lead to soil acidification and loss of base cations (e.g., calcium, magnesium, and potassium). Nitric acid (HNO_3) is a strong acid and directly contributes H^+ when it dissociates. Ammonia and NH_4^+ contribute 4 H^+ ions during nitrification, and acidification under high NH_3 deposition is well documented in Europe. Most California soils have high base cation saturation, and appear relatively resilient to acidification, but long-term deposition can reduce base cation saturation and increase acidity.

2.4.1. Nitrogen saturation

N-deposition is a cumulative process, eventually leading to N-saturation. Increased N inputs accelerate N-cycling, as greater litter fall with lower C:N ratios and increase decomposition and mineralization rates, which then stimulate nitrification and production of NO_3^-. Eventually, biotic demand for N (plant uptake and microbial immobilization) is exceeded by supply and N-saturation commences, representing a breakdown of biotic controls over N-cycling and exports.

Nitrogen saturation occurs in several stages in xeric western forests (Figure 2). Stage 0 is the original condition of low deposition, with low NO emissions and NO_3^- leaching—a high fraction of net nitrification is taken up by plants and microbes, and effectively recycled within the system. In Stage 1, incremental N-deposition leads to higher N-availability via increased nitrification and stomatal uptake by plants, leading to increases in net primary productivity (NPP). At saturation (Stage 2), NO emissions and NO_3^- leaching increase as plant uptake and microbial immobilization fall behind nitrification. Decline (Stage 3) is usually the result of multiple stress interactions, including ozone stress, susceptibility to bark beetles, and reduced fine-root biomass (Fenn, Baron, et al. 2003). Nutrient imbalances lead to stress and mortality, decreasing biotic N demand, but also increasing dead biomass inputs. N-saturated watersheds in Southern California have some of the highest levels of NO production and NO_3^- leaching recorded worldwide from non-agricultural ecosystems.

Excess nitrate leaching into surface and groundwater is a major symptom of N-saturation, and poses risks to water quality. A full discussion of water quality impacts is beyond the scope of this report



Source: (Fenn, Poth et al. 1998)

Figure 2. Stages of N-saturation in western xeric forests

The cumulative nature of N-deposition has lead to the concept of *critical loads*, defined as "a quantitative estimate of an exposure to N as NH_x and NO_y below which empirical detectable changes in ecosystem structure and function do not occur according to present knowledge." (Bull 1992; Bull and Sutton 1998) Applicability of critical loads to California ecosystems will be discussed below, but the rigorous identification of critical loads for specific ecosystems is beyond the scope of this report. Critical loads to sensitive European grasslands range as low as 5 kg-N ha^{-1} yr^{-1}, and critical loads for oligotrophic lakes may be even lower (Fenn, Baron et al. 2003). Throughout the comparative discussion of N-deposition exposure, a standard benchmark of 5 kg-N ha^{-1} yr^{-1} is used. This benchmark does not imply that 5 kg-N ha^{-1} yr^{-1} is the critical load for negative impacts for all ecosystems—some may be more sensitive and some may be less sensitive. As better information becomes available, this benchmark number may be modified for particular ecosystems; for this reason, data are graphically presented so that any benchmark can be used.

It is important to realize that the widespread increased atmospheric deposition of oxidized and reduced nitrogen is an unprecedented development—background levels across much of the world are estimated at 0.25-1 kg-N ha^{-1} yr^{-1}. The cumulative and insidious nature of N-deposition effects on ecosystems may be realized only after decades of elevated N inputs, and critical cumulative loads are poorly understood for most California ecosystems.

2.5. Mechanisms by Which N-deposition Can Lead to Impacts on Sensitive Species

2.5.1. Direct toxicity

Potential cases of direct toxicity of N compounds have been reported specifically in California. High ambient levels of HNO_3 in the Los Angeles Basin can approach levels that directly damage conifer foliage, and perhaps other species. High soil N may also be directly toxic—100% of *Artemisia californica* (sagebrush) seedlings died when grown in soils with NO_3^- concentrations similar to field concentrations of high-deposition areas near Riverside. However, these experiment are based on high exposure under artificial conditions. There is some evidence that NO may have direct inhibitory effects on plants at high concentrations (Mansfield 2002). Peroxyacetyl nitrate (PAN) may be toxic as well (Grosjeans and Bytnerowicz 1993).

2.5.2. Changes in species composition among native plants

In Europe, a large body of work has linked N-deposition to changes and losses of biodiversity in bogs, grasslands, heathlands, and forest understory (Bobbink, Hornung et al. 1998; Bobbink and Larners 2002; Stevens, Dise et al. 2004). Increases in nitrophilous grasses, primarily perennials but also some annuals, are a common response in species-rich grasslands on acid soils and calcareous soils, and in heathlands. Acidification from large amounts of NH_3 deposition also contributes to floral changes, but species losses in acid grasslands in the UK are proportional to N-deposition levels and only weakly associated with acidity. Heathlands convert to grasslands when *Calluna vulgaris* (heather) canopies open from herbivory, stress, and disturbance, and nitrophilous grasses quickly establish and dominate. Comprehensive reviews of N-deposition impacts on European ecosystems can be found in several edited compilations (Langran 1999; Bell and Treshow 2002).

Changes in native species composition in California habitats directly attributable to N-deposition have not been explicitly identified, except in the case of invasive species as described below. Air pollution can affect species composition in native dominated habitats—ozone induced mortality in ponderosa and Jeffery pines has led to increases in ozone-resistant species such as incense cedar and white fir in Southern California forests, but the interactions with N-deposition remain an active research arena (Fenn, Poth et al. 2003).

2.5.3. Enhancement of invasive species

Invasive plant species have severely altered numerous California ecosystems. The major documented mechanism of N-deposition impacts on sensitive species is the enhancement of invasions by nonnative species, especially annual grasses. Historical annual grass invasions into richer soils, prior to widespread N-deposition, have restricted many native grassland species to patches of thin soil, or onto naturally nutrient-poor soils such as serpentine. Many, if not most, non-native annual grass species respond strongly to N additions by increasing growth and seed production (e.g. Jones and Evans 1960; Jones 1963; Huenneke, Hamburg, et al. 1990; Yoshida and Allen

2004). Invasive grasses, both annual and perennial, have been documented to alter biodiversity and ecosystem function across the world (D'Antonio and Vitousek 1992). They are highly effective in depleting shallow soil moisture, and provide continuous fine fuels that accelerate fire cycles. Dense buildup of thatch smothers short-statured native plants and suppresses seedling recruitment. Once annual grasses replace shrubs, N-cycling rates increase and continue to favor grasses over shrubs.

Increased fire frequency, driven by annual grass invasions, is hypothesized to drive type conversions in many ecosystems along a biomass gradient. Low biomass shrublands are most sensitive, but chaparral and forests may be vulnerable over longer time-scales (Fenn, Baron et al. 2003). There is some current controversy over the exact role of N-deposition in type conversions of some California shrublands (Keeley, Keeley, and Frothingham 2005), and like any complex ecological problem there may be multiple forcing factors. But, the strong positive response of annual grasses to N-fertilization clearly implicates N-deposition in many of the cases discussed below.

Invasions of many other nonnative weeds are likely enhanced by N-deposition. These plants have high relative growth rates, are effective competitors for water, nutrients, and light, have few herbivores, and respond strongly to N-availability.

2.6. Specific California Ecosystems Known to Be Sensitive

The following accounts are brief summations of documented effects of N-deposition on specific California ecosystems. For a fuller review and extensive literature citations, see (Fenn, Baron et al. 2003).

2.6.1. Conifer forests

Mixed conifer forests of many different sub-types occur across large swaths of California. N-deposition in conifer forests in Southern California leads to high nitrification rates, leaching of NO_3^- into ground and surface waters, and emissions of NO. Impacts of ozone on mixed conifer forests have been extensively documented, and include reductions in photosynthesis and productivity. The combination of high ozone and high N-deposition reduces needle retention, disrupts root growth, increases foliage N, weakens trees, and can leave forests vulnerable to insects. Biomass and litter accumulation increases fuel loads and eventual fire intensity.

2.6.2. Evergreen chaparral

Chaparral ecosystems in the San Gabriel Mountains and Southern Sierra Nevada have experienced N-saturation, as evidenced by high NO_3^- leaching, accumulation of soil NO_3, and high emissions of NO.

In comparison to coastal sage scrub or even Mohave shrublands, chaparral ecosystems are nitrogen-rich. Many of the dominant species are nitrogen fixers, so increases in N-availability is not likely to change the ecosystem function or processes.

Changes in species composition in evergreen chaparral have not been documented. The closed canopy of chaparral can effectively keep out annual grasses in the absence of fires. Following fires, a fire-following herbaceous flora can dominate for several years, until resprouting shrubs and seedling recruitment close the canopy. Post-fire seeding with *Lolium multiflorum* (Italian ryegrass, an annual) and *Lolium perenne* (Perennial ryegrass) for erosion control can suppress the herbaceous phase. *Lolium* responds strongly to N-deposition (see Section 2.6.5). Increased fire frequency can reduce shrub diversity, and eventually eliminate shrubs.

2.6.3. Coastal sage scrub

Coastal sage scrub (CSS) is a primarily deciduous shrubland that occupies relatively dry sites along the coast and further inland. Typical species include *Artemisia californica, Eriogonum* sp., and *Salvia* sp. The relative dominance of species and degree of canopy closure changes along geographic gradients, and these changes are reflected in sub-types of sage scrub—Diegan, Riversidian, Venturan, Central (Lucian), and Northern (Franciscan). Coastal sage scrub in southern California supports a wealth of sensitive species that are at risk from habitat destruction by urban development.

Mature coastal sage has few nitrogen fixers in the mature vegetation stands, thus the ecological processes and functions tend to be more sensitive to changes in nitrogen cycling. Furthermore, in CSS during most years, evapotranspiration exceeds rainfall and no runoff occurs—so any nitrogen that deposits in the ecosystem stays in the ecosystem. Leaching losses may occur only under exceptionally high rainfall events, so soil nitrate tends to accumulate through time.

In high N-deposition areas near Riverside (20–35 kg-N ha^{-1} yr^{-1}), CSS provides a well-studied case of large-scale annual grass invasion converting shrublands to grasslands. N-deposition has been implicated as a major (but not the only) driver of these invasions. (Fenn, Baron et al. 2003). Major invasive grasses include *Bromus madritensis rubens, Avena* sp.¸ and other *Bromus* sp. Dense annual grass can eliminate small native forbs, suppress shrub recruitment, and provide fine continuous fuels that lead to stand-replacing fires. Two successive burns can effectively eliminate shrubs. Mycorrhizal fungal diversity drops with increasing N-deposition (Egerton-Warburton and Allen 2000). Qualitative observations of annual grass invasions in CSS east of San Diego (B. Toone, San Diego Zoological Society, pers. comm. July 2004) indicate that N deposition may be having similar effects there.

The change from shrublands to annual grassland increases the rate of N-cycling in the ecosystem. In annual grasslands, biomass turnover is faster and litter C:N ratio is lower. Shrubs accumulate woody biomass that decomposes slowly, and resorption of leaf N (and other nutrients) reduces litter quality.

Management of annual grasses in CSS poses many difficulties. Restoration to shrublands may be difficult and expensive. Changes in the mycorrhizal community may favor grasses over reestablishment of shrubs. Grazing by cattle, effective for controlling annual grasses in serpentine grassland and vernal pools (see below), may threaten the uninvaded lenses of clay soils that still support cryptobiotic crusts and native forbs. Occasional leaching/flushing events may provide opportunities for shrub re-establishment.

2.6.4. Desert scrub

California desert scrubs vary greatly across elevation climatic gradients, and are characterized by widely spaced shrubs and showy displays of annual wildflowers in wet years. In the Mojave Desert, N-deposition can lead to invasions by annual grasses, including *Bromus madritensis rubens* (red brome), and *Schismus barbatus* (Mediterranean annual split grass) (Brooks 2003). Wet years greatly intensify the grass invasions, and fine continuous fuel loads encourage extensive stand-replacing fires that were not possible prior to the grass invasions. In cooler deserts, *Bromus tectorum* (cheatgrass) has invaded large tracts with similar results, although invasions have occurred in the absence of significant N-additions (D'Antonio and Vitousek 1992).

2.6.5. Bay Area serpentine grassland

In the San Francisco Bay area, serpentine soils support native grasslands with high diversity of annual and perennial wildflowers, and perennial bunchgrasses (right side of fence in Figure 3). Under N-deposition, ungrazed serpentine grasslands (left side of fence in the Figure 3) are invaded by annual grasses primarily *Lolium multiflorum* (Italian ryegrass), *Hordeum murinum leporinum* (wild barley), *Bromus hordaceous* (soft chess), *Bromus madritensis* (red brome), and *Avena* sp. (wild oats) (Weiss 1999). *Lolium* growth strongly responds to N-fertilization and additional water, and rapidly absorbs and assimilates atmospheric NH_3 through stomata (Sommer and Jensen 1991). Nitrogen dioxide may also produce similar responses (Fowler 2002; Mansfield 2002). Concentrations of HNO_3 in south San Jose approach those in polluted parts of the Los Angeles Basin (S.B. Weiss unpublished data). N-deposition effects have been observed along regional pollution gradients and local gradients adjacent to a heavily traveled freeway.

Figure 3. San Francisco Bay Area grasslands in serpentine soils. The area on the left is ungrazed and dominated by non-native grasses. The area on the right is grazed and dominated by native species

Losses of plant diversity are accelerated by accumulation of grass thatch, which smothers small annual forbs. Moderate cattle grazing maintains high plant diversity in these grasslands, because cattle selectively graze N-rich *Lolium*, remove N and biomass from the system, prevent thatch buildup, and provide bare mineral soil for annual forb germination. Cattle also redistribute N and accelerate local N-cycling rates.

Bay Area serpentine grasslands are a biodiversity hotspot, supporting numerous threatened and endangered species, including the Bay Checkerspot butterfly, *Euphydryas editha bayensis* (USFWS 1998). Population extinctions of the butterfly follow grass invasions, because the larval host plant, *Plantago erecta* (dwarf plantain, a short annual forb) is crowded out by grass invasions.

The N-deposition threat to protected species in serpentine grasslands prompted precedent-setting mitigation for power plant emissions from the Metcalf Energy Center in San Jose (and other power plant projects, see Table 1), stimulated specific mitigation for highway projects and industrial developments, and drove the initiation of a Habitat Conservation Plan/Natural Communities Conservation Plan (HCP/NCCP) for Santa Clara County.

2.6.6. Mountain lakes

Primary productivity in Lake Tahoe has increased greatly over the last decades, and has changed from N-limitation to P-limitation (Jassby, Reuter et al. 1994). Atmospheric deposition is a primary source of elevated N in Lake Tahoe, contributing more than half of the N-loading, but the overall N-budget of the Tahoe Basin is still uncertain. Similar changes in phytoplankton communities—a shift from oligotrophic to more mesotrophic species—have been documented in the Southern Sierra Nevada (Fenn, Poth et al. 2003).

2.6.7. Lichen communities

Lichens are common and diverse in many ecosystems, and are sensitive indicators of various air pollutants. Nitrogen-sensitive lichen species have disappeared from high N-deposition areas—more than 50% of the native lichens in parts of the Los Angeles Basin have disappeared. Evidence of affected lichen communities extends across much of the state (Fenn, Baron et al. 2003).

2.7. Other California Ecosystems that May Be Sensitive

2.7.1. Vernal pools

Vernal pools are seasonal wetlands that contain water in the winter rainy season and dry over the summer drought. An impervious subsoil layer (hardpan or claypan) prevents rapid drainage. Vernal pools are characterized by a pronounced mound to pool bottom gradient, where mounds support upland grassland, with progressively longer flooding periods as one descends to the pool bottom. Pool bottoms and intermediate zones are characterized by a unique flora and fauna adapted to seasonal flooding. Many rare, threatened, and endangered species—both plants and animals—are found in vernal pools.

Annual grass invasions in vernal pools have been documented in the Sacramento Valley (Barry 1998; Gerhardt and Collinge 2003). Recent work in the Consumnes Reserve (Marty 2005) has identified annual grasses as a major threat to ungrazed vernal pools (Figure 4). When annual grasses are allowed to grow ungrazed, they evaporate more water from the mound areas, reducing inundation periods in the pools and allowing grasses to further invade deeper portions of the pools. These grass invasions, which occur over 2–3 years, lead to a direct loss of biodiversity of native vernal pool plants through competition and thatch buildup, and the shorter inundation periods lead to losses of invertebrates such as endangered fairy shrimp, and tiger salamander and red-legged frogs. Annual grass invasions, especially by *Lolium multiflorum*, have been noted in vernal pool systems in Sonoma County, with substantial losses of native biodiversity including listed plant species (D. Glusenkamp, Audubon Canyon Ranch, pers. comm.).

Given the well-documented responses of annual grasses to N-additions, and impacts in other California ecosystems, the intensity of annual grass invasions in vernal pools is likely increased by N-deposition and vernal pools can be considered a sensitive ecosystem.

Figure 4. Grassland invasion at a vernal pool

2.7.2. Sand dunes

Annual grass invasions in the Antioch Dunes threaten the endemic flora and fauna of this inland dune system (Steve Edwards, East Bay Regional Park District, pers. comm.). Coastal dune systems are in relatively clean coastal air, but inland sand dune systems may be at risk. Annual grass invasions have been noted in eolian sands in the Arena Plains San Joaquin Valley, where cattle grazing has been a key management practice (Silviera 2000).

2.7.3. California "annual" grassland

Although many California grasslands are dominated by invasive annual grasses and forbs, they can still support local concentrations of native wildflowers and bunchgrasses. Increased annual grass growth stimulated by N-deposition may further restrict native forbs to nutrient-poor thin soils around rock outcrops and on steep slopes.

Coastal grasslands are susceptible to invasion by the native shrub *Baccharis pilularis* (coyote brush) in the absence of fire or grazing. Such invasions occur in clean coastal areas, so N-deposition is likely not the primary driving factor, but the potential contribution of N-deposition to this process is not known.

2.7.4. Oak woodlands

Oak woodlands and savannahs have understory grasslands—formerly dominated by native perennial grasses and annual and perennial forbs, but now dominated by introduced annual grasses—that may be affected by increased annual grass growth as described above. Annual grasses are effective competitors for soil moisture in spring, and have been implicated in suppressing oak seedling recruitment. Grazing removal from oak woodlands in the East Bay regional Park District has led to intensified invasions of annual grasses (S. Edwards. EBRP, pers. comm.), but grazing can also directly affect oak recruitment, and remains a contentious issue in resource management.

2.7.5. Alpine communities

In alpine areas in Colorado, N-deposition has been linked to changes in species composition, with an increase in nitrophilous species and changes in N-cycling. N-inputs may be particularly high and effects substantial in wet meadows where windblown snow accumulates and water limitations are few. Water limitations in rocky fell field communities may restrict growth responses to increased N-deposition. No comparable changes have been explicitly documented in California.

2.7.6. Serpentine soils (other than Bay Area grasslands)

Serpentine soils provide numerous limitations to plant growth, including low calcium, phosphorus, molybdenum, and nitrogen, and high magnesium, nickel, chromium, and other heavy metals. Soils tend to be thin and rocky. The unique and harsh growing conditions on serpentine soils, combined with their island-like distribution have led to the evolution of many serpentine endemic plants. Serpentine soils also provide a refuge for many species crowded off richer soils by invasive species. Serpentine communities range from stunted conifer forests, chaparral, grasslands, and near total barrens.
N-deposition may promote annual grass invasions in serpentine soils. Reports of non-native grasses invading serpentine habitats have been accumulating (Harrison, Inouye et al. 2003). In some cases it appears that some grass species are becoming better adapted to serpentine, but links to N-deposition have not been made explicit. Other serpentine sites where grass invasions have been noted include the Red Hills in Tuolumne County (J.B. Norton, UC Cooperative Extension, pers. comm.).

2.7.7. Alkali sinks

Low-lying areas in deserts and semi deserts accumulate salts and provide habitat for a variety of halophytes. Drier upland soils may be dominated by annual grassland. Dense grass growth and thatch are present in places such as the Springtown Sink near Livermore, covering all but the most saline soils (Figure 5). The potential for N-deposition effects in these habitats has not been explicitly addressed, but alterations similar to those in vernal pools may be expected.

Figure 5. Dense grass growth and thatch in alkali sink near Livermore, California

2.7.8. Salt marshes

Salt marsh productivity is limited by N (Morris 1991). Salt marshes export organic N to adjacent coastal waters, but are also major sites for denitrification. Many salt marshes are locally subjected to elevated N in sewage effluent. The direct impacts of atmospheric N-deposition on California salt marshes have not been assessed. The potential for atmospheric N-deposition to enhance invasion rates by non-native *Spartina* (salt grass) around San Francisco Bay is unknown.

2.7.9. Freshwater marshes

Nitrogen can be limiting to productivity in freshwater marshes (Morris 1991), but the role of atmospheric N-deposition in California freshwater marshes is not known at present.

2.7.10. Other edaphic oddities

California has pockets of unusual soils that support unique ecosystems because of harsh growing conditions. Ione clay is a unique ancient lateritic soil in the foothills of the central Sierra Nevada, supporting several local endemic taxa. Ione clays are heavily leached and very acidic. Impacts of N-deposition are unknown, but annual grasses are present among the endemic shrubs (see Figure 6). Limestone outcrops in the San Bernardino Mountains support a cluster of rare species, as do shallow infertile "pebble-plains" at higher elevations. Gabbro soils in the Sierra foothills also support a cluster of rare species, but no documentation of annual grass invasion or N-deposition impacts has been reported.

Figure 6. Grasses among endemic shrubs (*Arctostaphylos myrtifolia*) in the Ione formation

2.7.11. Surface waters

The leaching of nitrate from N-saturated ecosystems contributes to water quality problems downstream. While nitrate pollution of groundwater and release to surface waters is widely recognized in agricultural areas, there may be atmospheric deposition inputs in other areas, especially in mountain watersheds in the Los Angeles Basin and other high pollution zones. The effects of large nitrate pulses into coastal waters may contribute to near-shore pollution episodes.

3.0 Distribution of N-deposition in California and Ecosystem Exposure

3.1. Distribution of N-deposition at 36 km

The 36 x 36 km CMAQ map of total annual N-deposition identifies levels of exposure across California (Figure 7). Hill-shaded topography and county boundaries are shown to facilitate geographic location. The map is repeated without the topography in following sections. It is extremely important to note that the 36 km scale precludes highly site-specific assessment, and provides a screening tool appropriate to regional-scale analyses. Sharp coastal gradients, in particular, are only approximated at best, and local hotspots within grid squares cannot be resolved. Individual circumstances where greater resolution is needed for assessment accuracy will be identified, but fine-scale analysis will require the completed 4 x 4 km map currently being produced by the UCR group (forthcoming).

Figure 8 presents the overall distribution of N-deposition across California as a cumulative distribution function (CDF). In this presentation format, the proportion of total area below (or above) any selected N-deposition level can be read directly from the graph, and converted to absolute area (in hectares) by multiplying by the total area. For example, approximately 75% of the state (~30,000,000 ha) receives < 5 kg-N ha^{-1} yr^{-1}, or conversely, 25% (or ~10,000,000 ha) receives more. Similarly, approximately 4% (or ~1,600,000 ha) receives > 10 kg-N ha^{-1} yr^{-1}. This graph format will be consistently used for assessing exposure of specific vegetation types from the FRAP map, because it allows the determination for any chosen threshold.

Throughout the discussion of N-deposition exposure, a benchmark of 5 kg-N ha^{-1} yr^{-1} will be used for comparative purposes. If an ecosystem is exposed to substantial areas >10 kg-N ha^{-1} yr^{-1}, that is also noted. Once again, this benchmark does not imply that 5 kg-N ha^{-1} yr^{-1} is the critical load for negative impacts for all ecosystems—the CDF graphs are designed to allow for consideration of all potential thresholds for impacts as they are identified.

The obvious hotspot for N-deposition is the South Coast Air Basin (SoCAB), with a maximum deposition of 21 kg-N ha^{-1} yr^{-1} in the Central Los Angeles Basin, and surrounding cells of 13–16 kg-N ha^{-1} yr^{-1}, dropping off to 8–10 kg-N ha^{-1} yr^{-1} further east and north. Deposition in the Mojave Desert ranges from 6–9 kg-N ha^{-1} yr^{-1} in the west, and decreases to 3–4 kg-N ha^{-1} yr^{-1} in the east.

In the San Diego Air Basin (SDAB), maximum values are 8–9 kg-N ha^{-1} yr^{-1}, just east of San Diego. The coastal areas receive 1–2 kg-N ha^{-1} yr^{-1}. The lightly developed Camp Pendleton gap in Northern San Diego County (5 kg-N ha^{-1} yr^{-1}) is barely resolved at this scale. Deserts in eastern San Diego County receive 6 kg-N ha^{-1} yr^{-1}.

In the San Francisco Bay Area, the maximum deposition is 8–9 kg-N ha^{-1} yr^{-1}. The coastal grid squares such as the San Mateo County Coast have low deposition (1 kg-N ha^{-1} yr^{-1}), and inland areas in the East and South Bay receive 6 kg-N ha^{-1} yr^{-1}.

The deposition hotspot in the San Joaquin Valley is near Modesto (13–14 kg-N ha^{-1} yr^{-1}). The east side of the San Joaquin Valley and lower Sierra foothills receive from 5–9 kg-N ha^{-1} yr^{-1}. The west side of the Valley and adjacent slopes of the Inner Coast Ranges receive 3–4 kg-N ha^{-1} yr^{-1}.

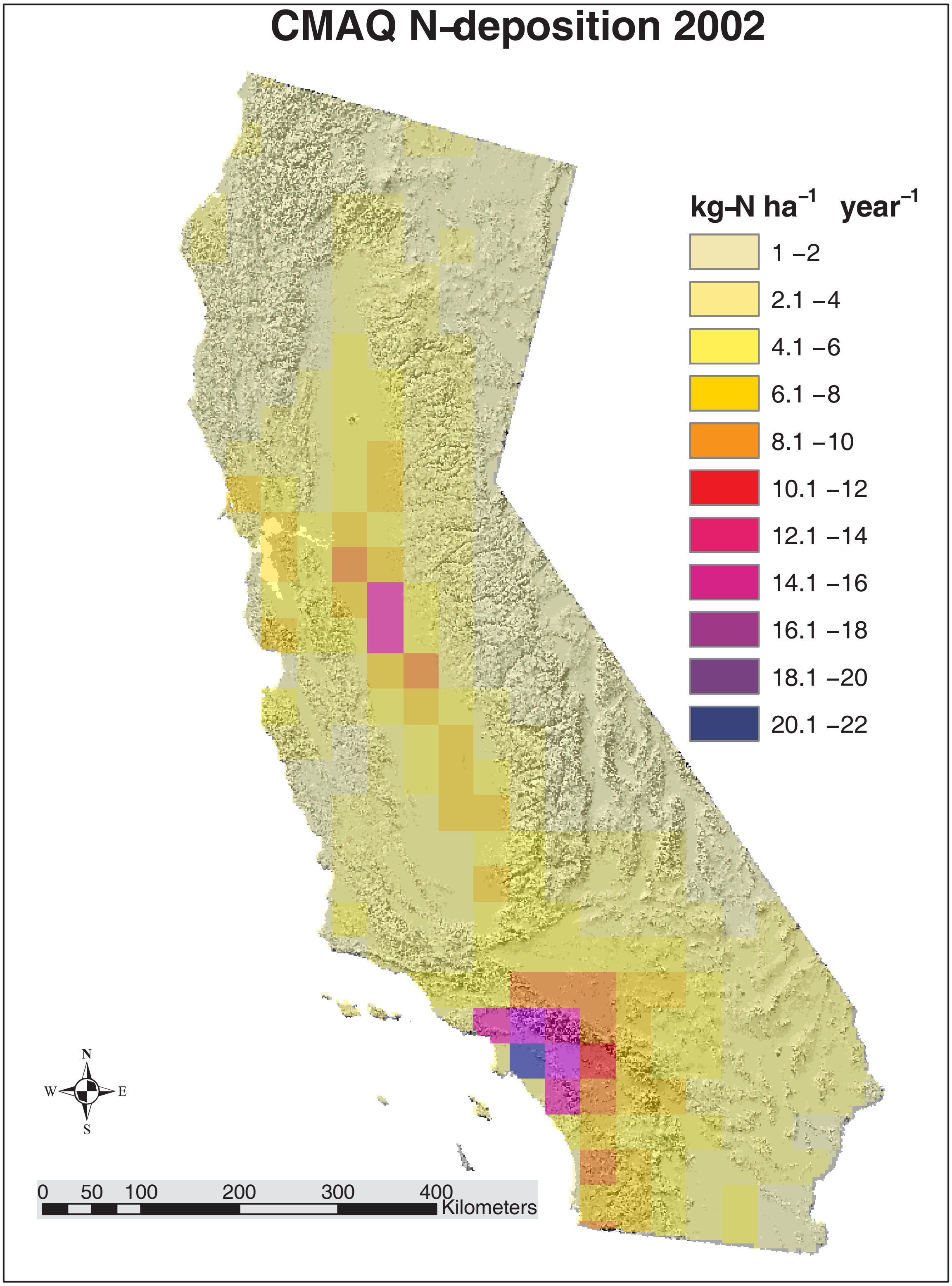


Figure 7. CMAQ 36 km N-deposition

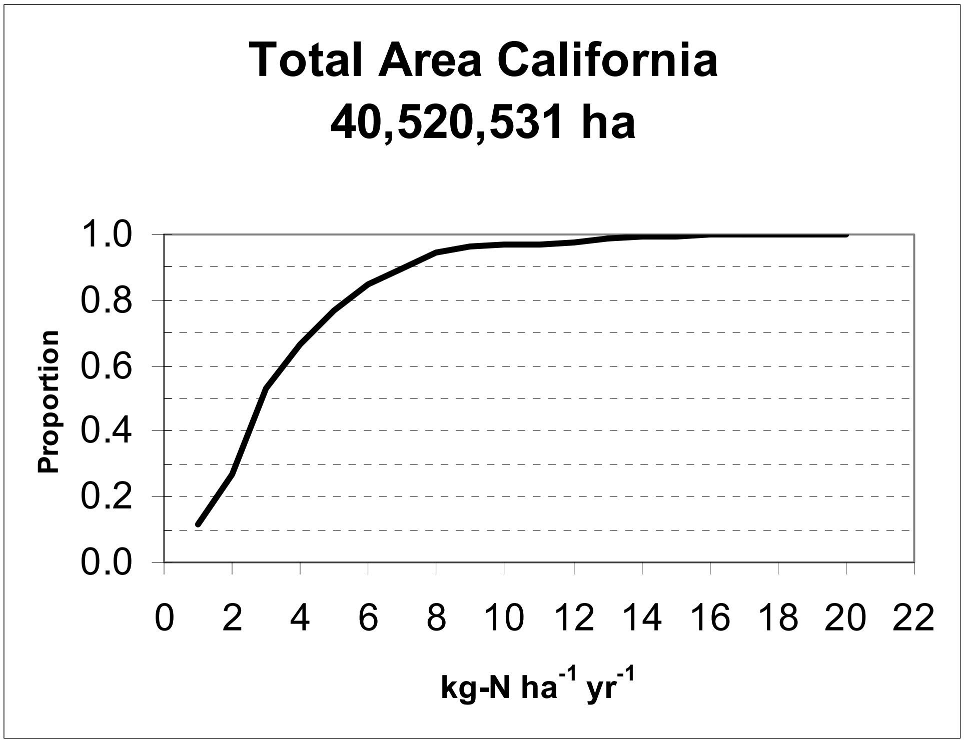


Figure 8. Statewide N-deposition proportion (CDF format)

Maximum values in the Sacramento Valley are 6–8 kg-N ha^{-1} yr^{-1} at the southern end and near Sacramento itself. The Northern Sacramento Valley receives 5–6 kg-N ha^{-1} yr^{-1} along the eastern side, and 3 kg-N ha^{-1} yr^{-1} on the western side.

Coastal areas are generally quite clean. The North Coast has a small area of 4 kg-N ha^{-1} yr^{-1} near Eureka. The Central Coast has two hotspots of 5 kg-N ha^{-1} yr^{-1} near Santa Maria and Monterey, and Ventura County receives 6 kg-N ha^{-1} yr^{-1}.

The Sierra Nevada exhibits a strong gradient away from the Central Valley, with deposition ranging from 4–5 kg-N ha^{-1} yr^{-1} at the lower elevations to 1–2 kg-N ha^{-1} yr^{-1} at the crest. The Eastside has low deposition, similar to the crest. The highest deposition in the Sierra Nevada is in the southern Sierra.

3.2. Ecosystem (Vegetation Type) Exposure

The overlay of the 36 x 36 km CMAQ model with the FRAP map (Figure 9) allows the broad-scale exposure of each vegetation type to N-deposition to be assessed. The complex map does not lend itself to detailed examination at such a small map scale, but is presented to illustrate the complexity of vegetation types in the state. Figure 10 presents the exposure levels to 48 FRAP vegetation types as cumulative distribution functions, as in Figure 8. The CDF graphs are grouped (approximately) by vegetation structure. Appendix A presents maps of the 48 FRAP vegetation types overlaid with the CMAQ 36 km deposition, in the same order as in Figure 10.

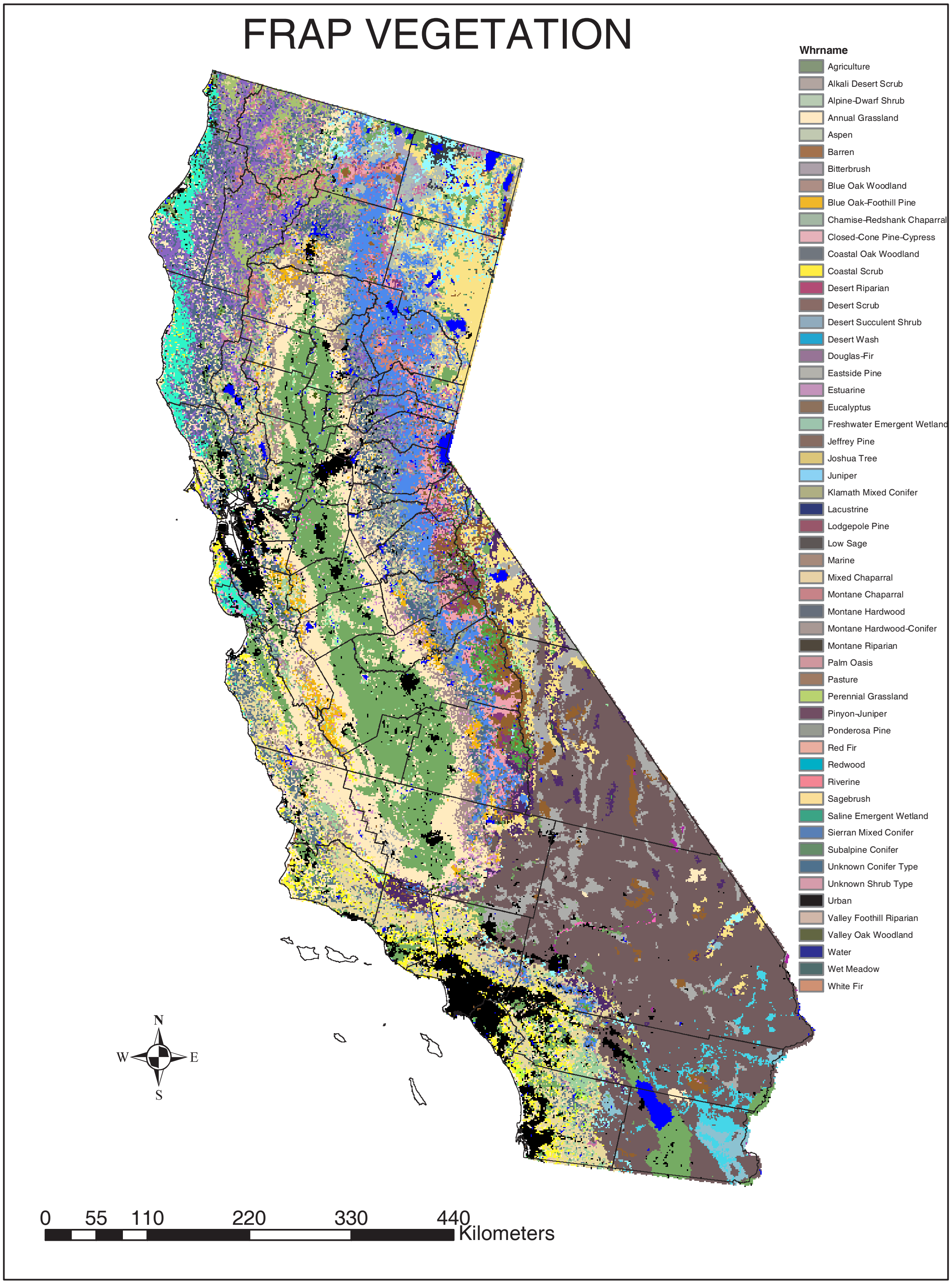


Figure 9. FRAP vegetation

3.2.1. Coastal sage scrub

Approximately 50% of CSS (350,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}. CSS is highly exposed to N-deposition in Southern California—the majority of the ~140,000 ha exposed to > 8 kg-N ha^{-1} yr^{-1} are near Riverside and San Diego. CSS on the central and north coasts is generally exposed to relatively low levels, but there are some hotspots around Santa Maria, Monterey, and the San Francisco Bay Area.

3.2.2. Annual grassland

Annual grassland covers more than 4,300,000 ha of lowland California. About 30% of the annual grassland receives > 5 kg-N ha^{-1} yr^{-1}. The majority of this grassland is on the east side of the Central Valley. These grasslands also support many vernal pools.

3.2.3. Wet meadows

Wet meadows are scattered across the state, and < 5% (~5000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}. These limited hotspots are in the Central Valley and Peninsular Ranges. Meadows in the High Sierra receive low N-deposition.

3.2.4. Perennial grasslands

Perennial grasslands are mapped mostly in San Diego County (especially the Camp Pendleton area), which may reflect a bias in the FRAP map. 90% (~23,000 ha) of mapped perennial grasslands are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.5. Agriculture

Agriculture covers > 4,500,000 ha of land, and is a major source of reactive N, especially NH_3, in the atmosphere. 50% of agricultural land receives > 5 kg-N ha^{-1} yr^{-1}, and 5% (225,000 ha) receives a "fertilizer subsidy" of > 10 kg-N ha^{-1} yr^{-1}.

3.2.6. Urban

Urban areas are the other major source of reactive N, producing NO_x from combustion and vehicles, and NH_3 from catalytic converters on vehicles. Deposition is naturally quite high within and near to urban sources, and 25% of the urban surface area receives > 10 kg-N ha^{-1} yr^{-1}.

3.2.7. Saline emergent wetland (salt and brackish marsh)

The largest remaining areas of salt marsh in California surround the San Francisco Estuary. 30% (~8500 ha) receive > 5 kg-N ha^{-1} yr^{-1}.

3.2.8. Freshwater emergent wetlands

Freshwater emergent wetlands include tule marshes, cattail marshes (both natural and managed) and are most abundant in the Central Valley. 50% (~40,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, and 5% (~4000 ha) are exposed to > 10 kg-N ha^{-1} yr^{-1}, primarily in the northern San Joaquin Valley (Modesto area).

3.2.9. Valley oak woodland

Valley oak woodland has been reduced to scattered remnants across the state, primarily on deep valley floor soils. 20% (11,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}. The grassland understory is likely the most sensitive component in all oak woodlands in the short-term.

3.2.10. Blue oak woodland

Extensive stands of Blue Oak Woodlands surround the Central Valley at elevations just above the annual grassland and extend into the Inner Coast Ranges. 20% (~225,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the Sierra Nevada foothills.

3.2.11. Coastal oak woodland

Coastal Oak Woodlands are dominated by evergreen oak species. 30% (~130,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, much of which in the San Francisco Bay Area. 4% (~17,500 ha) are exposed to > 10 kg-N ha^{-1} yr^{-1}, all in the Los Angeles Basin.

3.2.12. Blue oak-foothill pine woodland

Blue Oak-Foothill Pine Woodland occupies elevations just above the Blue Oak Woodland. 15% (~59,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the Mt. Hamilton Range (southeast of San Jose) and in the Tehachipis.

3.2.13. Montane hardwood-conifer

Montane hardwood-conifer is a closed canopy forest type. 10% (~65,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily east of San Diego and the eastern San Bernardino Mountains. 4% is exposed to > 10 kg-N ha^{-1} yr^{-1}, adjacent to the Los Angeles Basin.

3.2.14. Montane hardwood

10% (~180,000 ha) of montane hardwood forest is exposed to > 5 kg-N ha^{-1} yr^{-1}, including parts of the San Francisco Bay Area, San Diego, and the eastern San Bernardino Mountains. Only 1% is exposed to > 10 kg-N ha^{-1} yr^{-1}, adjacent to the Los Angeles Basin.

3.2.15. Valley foothill riparian

Valley-Foothill Riparian forests have been reduced to scattered remnants across the Central Valley and other inland valleys. 59% (~30,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, and 10% is exposed to > 10 kg-N ha^{-1} yr^{-1}, primarily in the northern San Joaquin Valley near Modesto, with small remnants in the Los Angeles Basin.

3.2.16. Montane riparian

Montane riparian forests occur as narrow strips in canyon bottoms in most mountain ranges in California. 10% (~8500 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the Transverse ranges near Ventura.

3.2.17. Mixed chaparral

Mixed chaparral occurs in numerous mountain ranges across California, and consists of diverse shrub species in various combinations that depend on local factors. 40% (760,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, and 10% (190,000 ha) is exposed to > 10 kg-N ha^{-1} yr^{-1}, with the highest exposure in extensive stands in the mountains around the Los Angeles basin.

3.2.18. Chamise redshank chaparral

Chamise redshank chaparral is dominated by *Adenostoma* sp. and is particularly abundant near the San Diego-Riverside County border. 50% (228,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, and only 2%–3% is exposed to > 10 kg-N ha^{-1} yr^{-1}.

3.2.19. Unknown shrub type

Various stands of difficult-to-characterize shrub stands in the Coast Ranges and Sierra Nevada foothills fall in this category. Twenty percent (41,000 ha) is exposed to > 5 kg-N ha^{-1} yr^{-1}, and very little (< 1%) is exposed to > 10 kg-N ha^{-1} yr^{-1}.

3.2.20. Bitterbrush

Stands of bitterbrush are distributed on the Modoc Plateau and around the Owens Valley, and are in relatively clean air areas. < 1% (1000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.21. Alpine-dwarf shrub

Alpine-dwarf shrub is distributed along the crest of the High Sierra and is minimally exposed to N-deposition.

3.2.22. Sagebrush

Sagebrush is mainly distributed east of the Sierra Nevada and Cascade ranges, with outlying patches in Mojave Desert mountains, Tehachipis, and Transverse Ranges. Less than 2% is exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.23. Montane chaparral

Montane chaparral is distributed at high elevations in the Sierra Nevada, Cascades, and Klamath Mountains. Small patches are found in the high mountains outside Los Angeles. About 5% (30,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily around the Los Angeles Basin.

3.2.24. Low sage

Low sage is distributed on the Modoc Plateau, and around the Owens Valley. None is exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.25. Ponderosa pine

Ponderosa Pine forests are distributed in the Sierra Nevada, Cascades, and Klamath Mountains. About 5% (15,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the southern Sierra Nevada.

3.2.26. Jeffrey pine

Jeffrey Pine forests are distributed in the central, southern and Eastern Sierra Nevada, with outlying stands in the Transverse ranges and Peninsular Ranges. 7% (20,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, and 6,000 ha are exposed to > 10 kg-N ha^{-1} yr^{-1} in the Los Angeles Basin.

3.2.27. Sierran mixed conifer

Sierran mixed conifer forests are distributed along the whole length of the Sierra Nevada and Cascades, with outliers in the Transverse and Peninsular Ranges. 4% (80,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, and 17,000 ha are exposed to > 10 kg-N ha^{-1} yr^{-1} around the Los Angeles Basin.

3.2.28. White fir

White Fir forests are distributed in the Northern Sierra Nevada, Cascades, and Klamath Mountains. Less than 1% are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.29. Lodgepole pine

Lodgepole Pine forests are distributed in the Sierra Nevada and Cascade Ranges. 0.5% (1,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.30. Red fir

Red-fir forests are distributed in the Sierra Nevada and Cascades. 0.5% (2,500 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.31. Subalpine conifer

Subalpine conifer forests are distributed across the High Sierra, Cascades, and Klamath Mountains, with outliers at the highest elevations of the San Gabriel, San Bernardino, and San Jacinto Mountains. 2% (5,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1} around the Los Angeles Basin.

3.2.32. Eastside pine

Eastside pine forests are distributed primarily east of the Cascades, with outliers on the east flanks of the San Gabriel and San Bernardino Mountains. 3% (15,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1} around the Los Angeles Basin.

3.2.33. Redwood

Redwood forests are distributed along the coast from Big Sur north. About 10% (50,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, in the San Francisco Bay Area. This may be an overestimate, because the 36 km CMAQ map does not capture steep coastal deposition gradients in Santa Cruz and Sonoma Counties.

3.2.34. Klamath mixed conifer

Klamath mixed conifer forests are distributed in far northern California, distant from major pollution sources. None are exposed to > 5 kg-N ha^{-1} yr^{-1}, with the highest exposure (4–5 kg-N ha^{-1} yr^{-1}) northeast of the Sacramento Valley.

3.2.35. Unknown conifer type

Coniferous forests of unclassified composition(s) are distributed in the Santa Cruz Mountains and Diablo Range, along with small patches along the west slope of the Sierra Nevada and the Tehachipis. 60% (26,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the southern San Francisco Bay Area.

3.2.36. Juniper

Juniper forests are distributed on the eastern slopes of most major mountain range, including the Peninsular and Transverse Ranges. 15% (60,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1} in Southern California.

3.2.37. Aspen

Aspen forests are distributed in the Central Sierra Nevada, and none are exposed to > 5 kg-N ha^{-1} yr^{-1}. Aspens themselves are present in many mid-high elevation coniferous forest types, including those of the Los Angeles Basin.

3.2.38. Closed-cone pine-cypress

Closed-cone pine-cypress forests are distributed in scattered pockets from the Mexican border to the North Coast Ranges. These forests contain some narrowly distributed conifers such as the Tecate Cypress in San Diego County. 10% (6,200 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.39. Pinyon juniper forests

Pinyon-juniper forests are distributed on the east flanks of most mountain ranges. 13% (60,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily on the east flanks of the Peninsular ranges.

3.2.40. Eucalyptus

Non-native eucalyptus forests were planted in many parts of California, relatively close to urban areas. 50% (2800 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}. Eucalyptus can invade adjacent native habitats, and groves on the immediate coast often support overwintering monarch butterflies.

3.2.41. Desert riparian

Small patches of desert riparian habitats are distributed across the Mojave and Colorado Deserts. 15% (2800 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1} in the western Mojave Desert. Desert riparian zones are susceptible to invasions by non-native tamarisk.

3.2.42. Palm oasis

Small areas of *Washingtonia* palms (total 1250 ha) exist around springs in the SW California deserts. 2.5% (35 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.43. Desert scrub

Desert scrub is distributed across southeastern California. 27% (2,000,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily from the western Mojave Desert south to Eastern San Diego County.

3.2.44. Alkali desert scrub

Alkali desert scrub occupies saline valley bottoms across the Mojave Desert, with outliers in the Southern Inner Coast Range. 15% (270,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the western Mojave Desert.

3.2.45. Barren

Barren land is distributed as high alpine (Sierra Crest and other high mountains) and low desert (Death Valley). 3% (50,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}, primarily in the Mojave Desert.

3.2.46. Joshua tree

Joshua tree woodlands are concentrated in the little San Bernardino Mountains. 50% (16,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}. Joshua trees themselves are much more widely distributed at middle elevations in the Mojave Desert than they are in the map of this vegetation type in Appendix A.

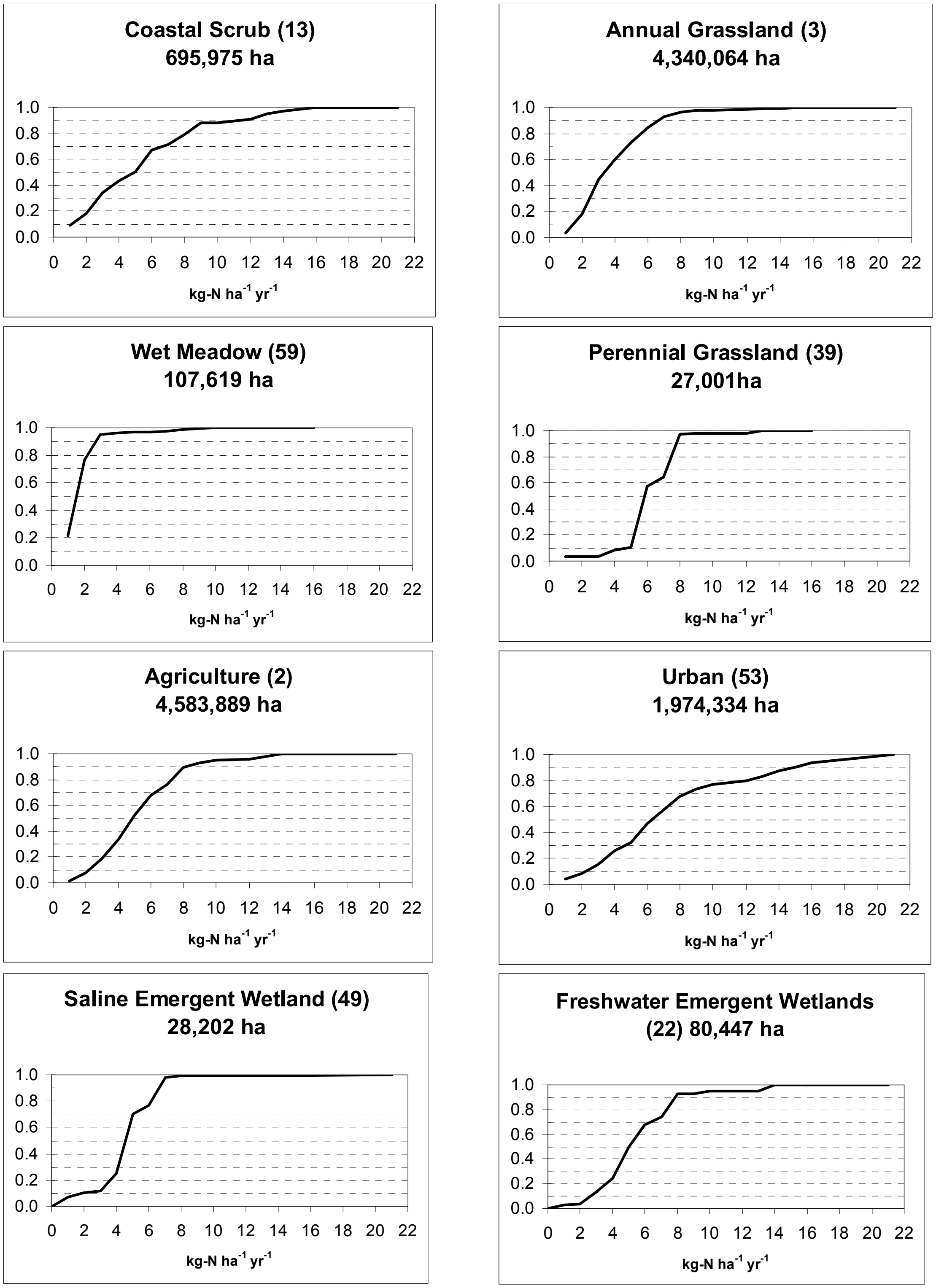
3.2.47. Desert succulent scrub

Desert succulent scrub, with a high proportion of cacti and other fleshy plants, is distributed in low-elevation deserts in San Diego and Imperial Counties. 17% (45,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

3.2.48. Desert wash

Desert washes are distributed in far southeastern California (Colorado Desert). 2.5% (26,000 ha) are exposed to > 5 kg-N ha^{-1} yr^{-1}.

Figure 10. Cumulative distribution functions of N-deposition exposure of FRAP vegetation types. The FRAP code numbers for each vegetation type are in parentheses, followed by total area in hectares so that proportions (Y axis) may be converted to area affected. Maps of each vegetation type are presented in Appendix A, in the same order.



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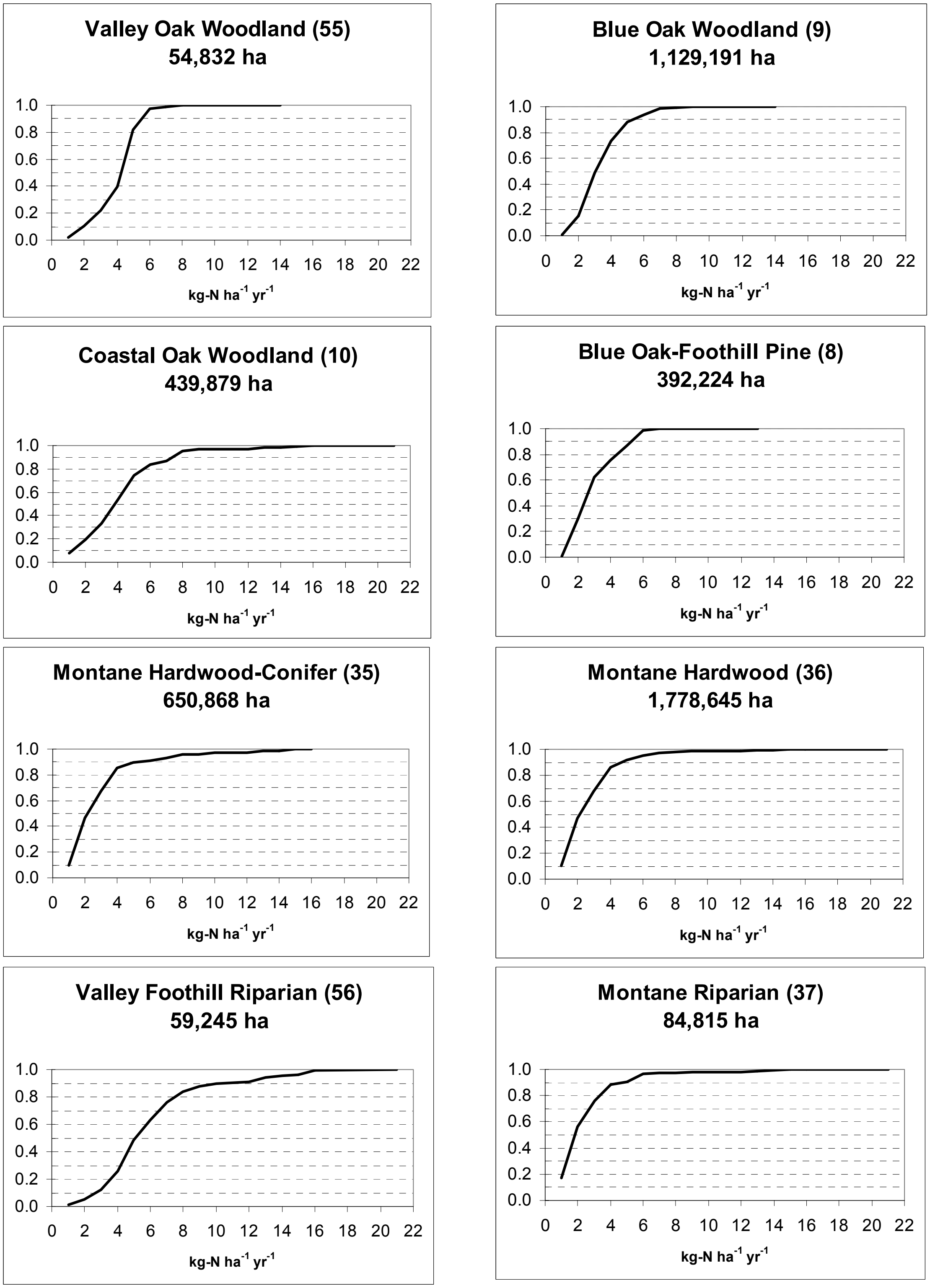

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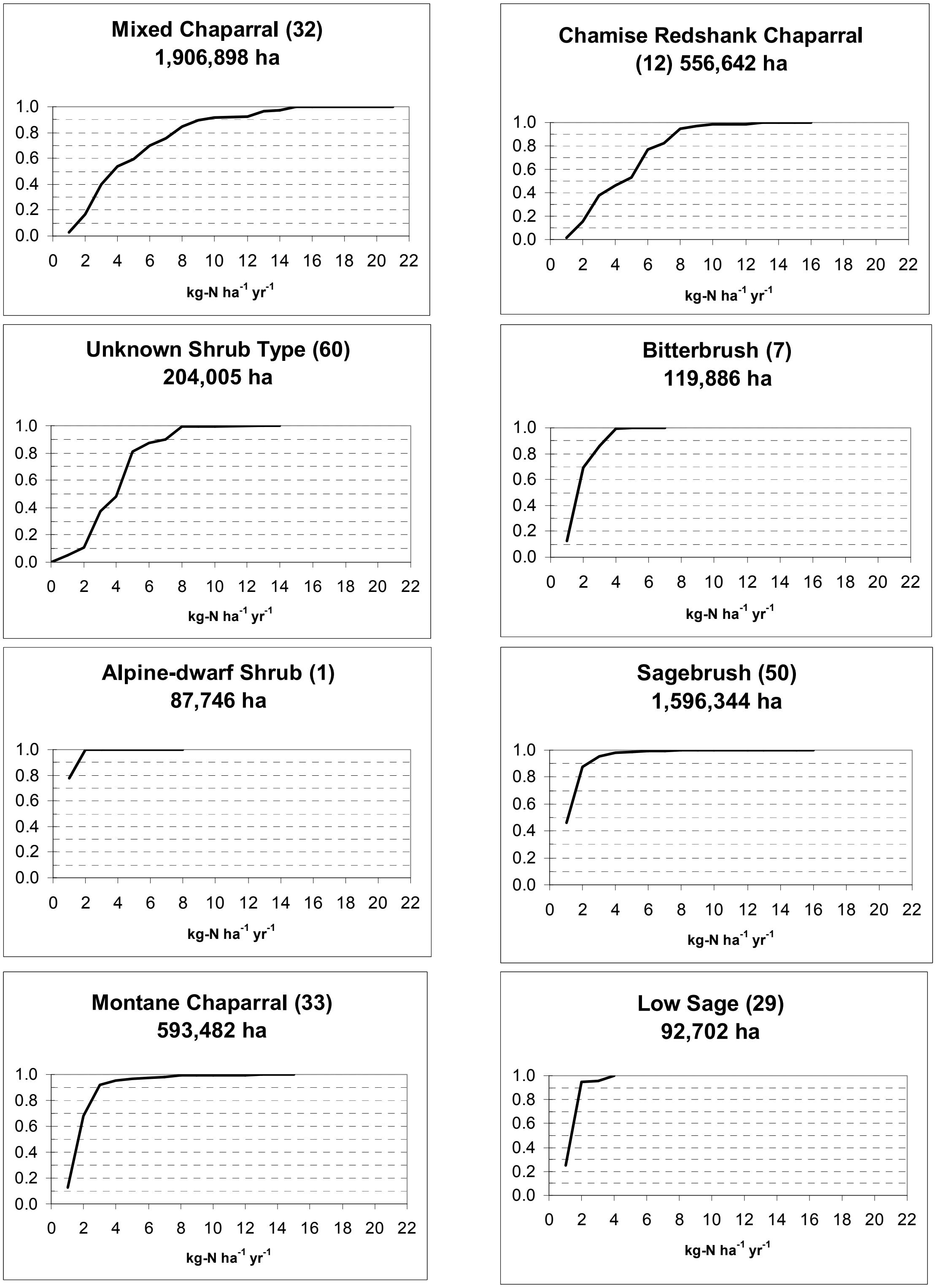

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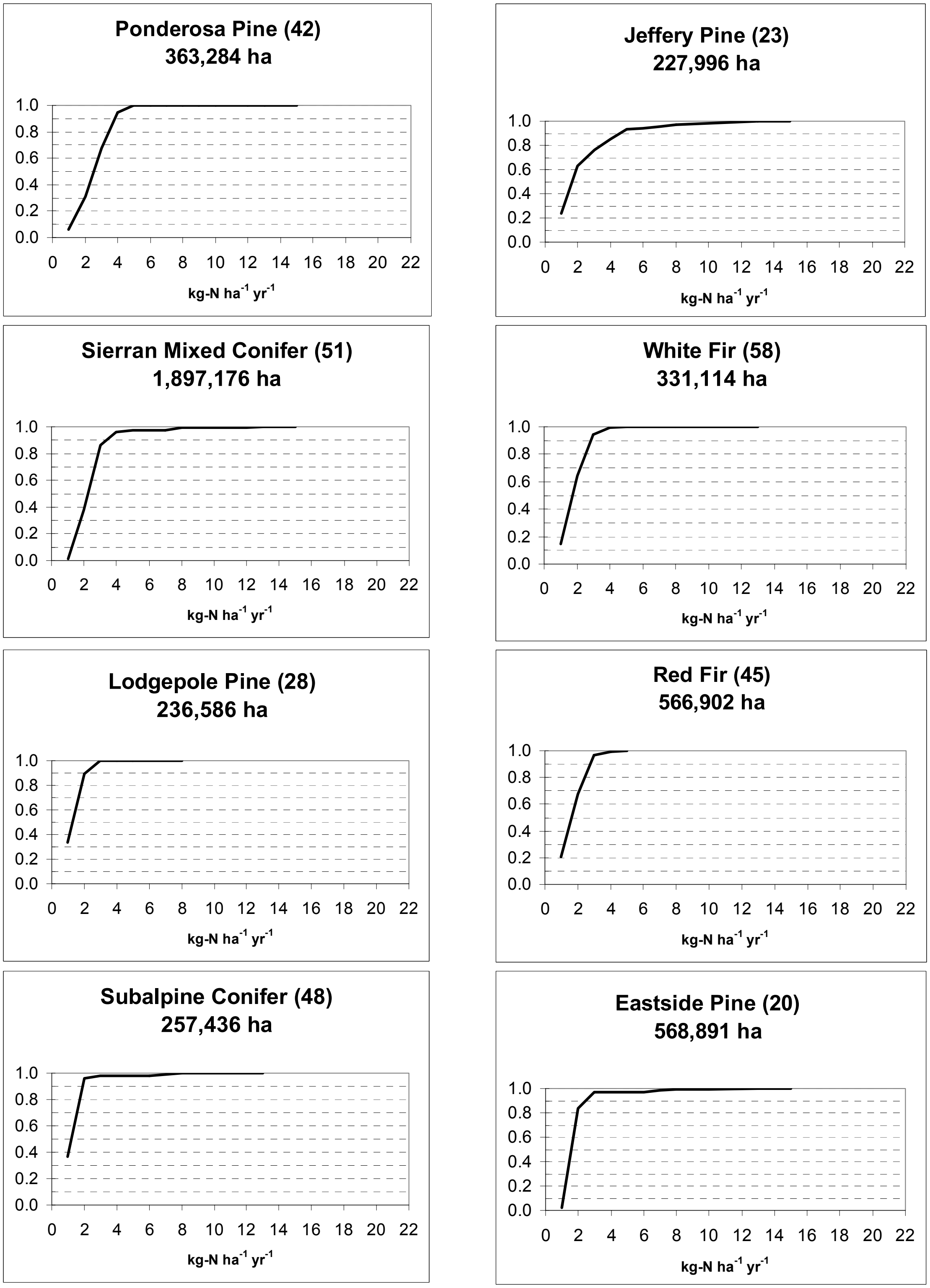

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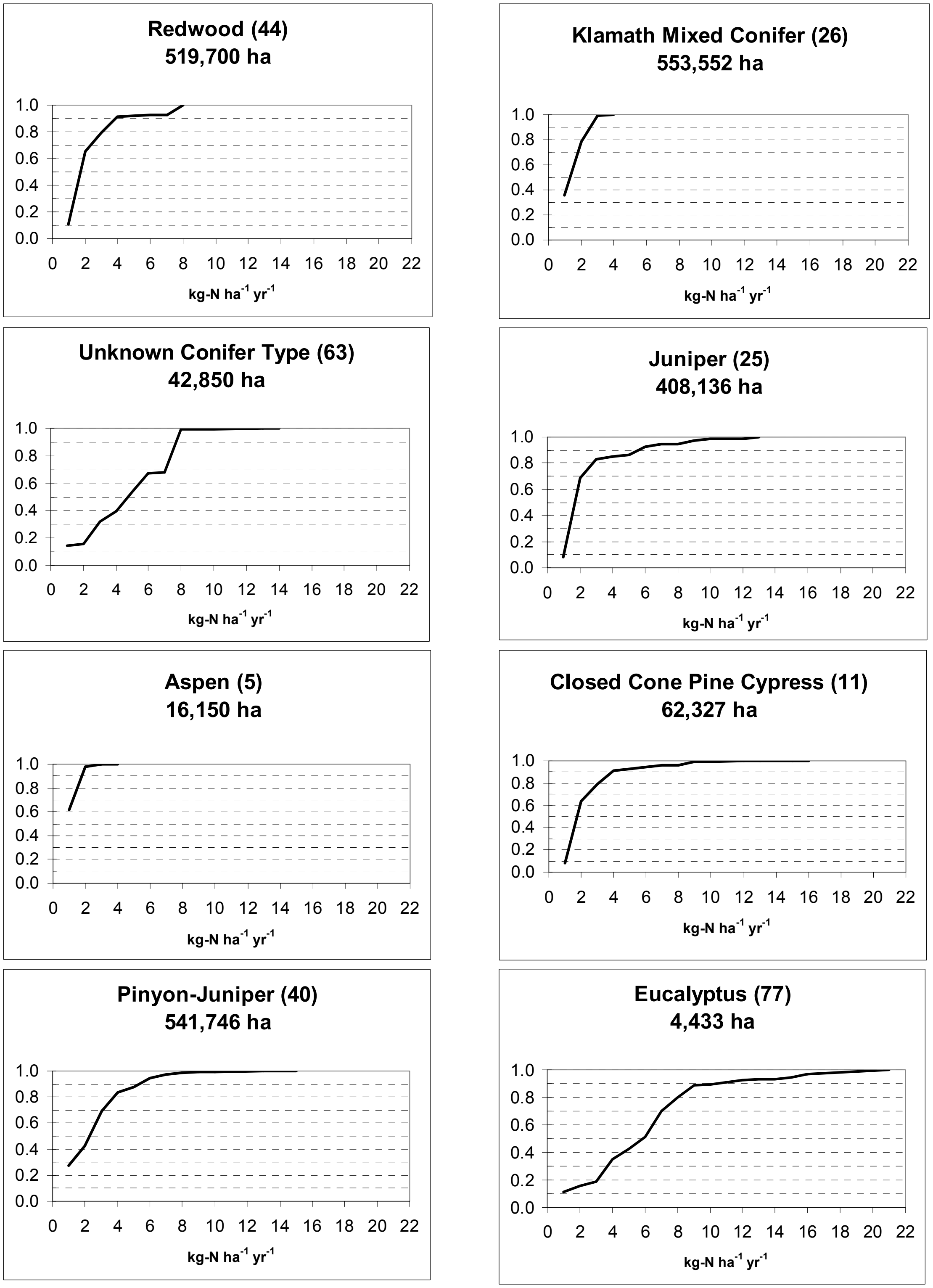

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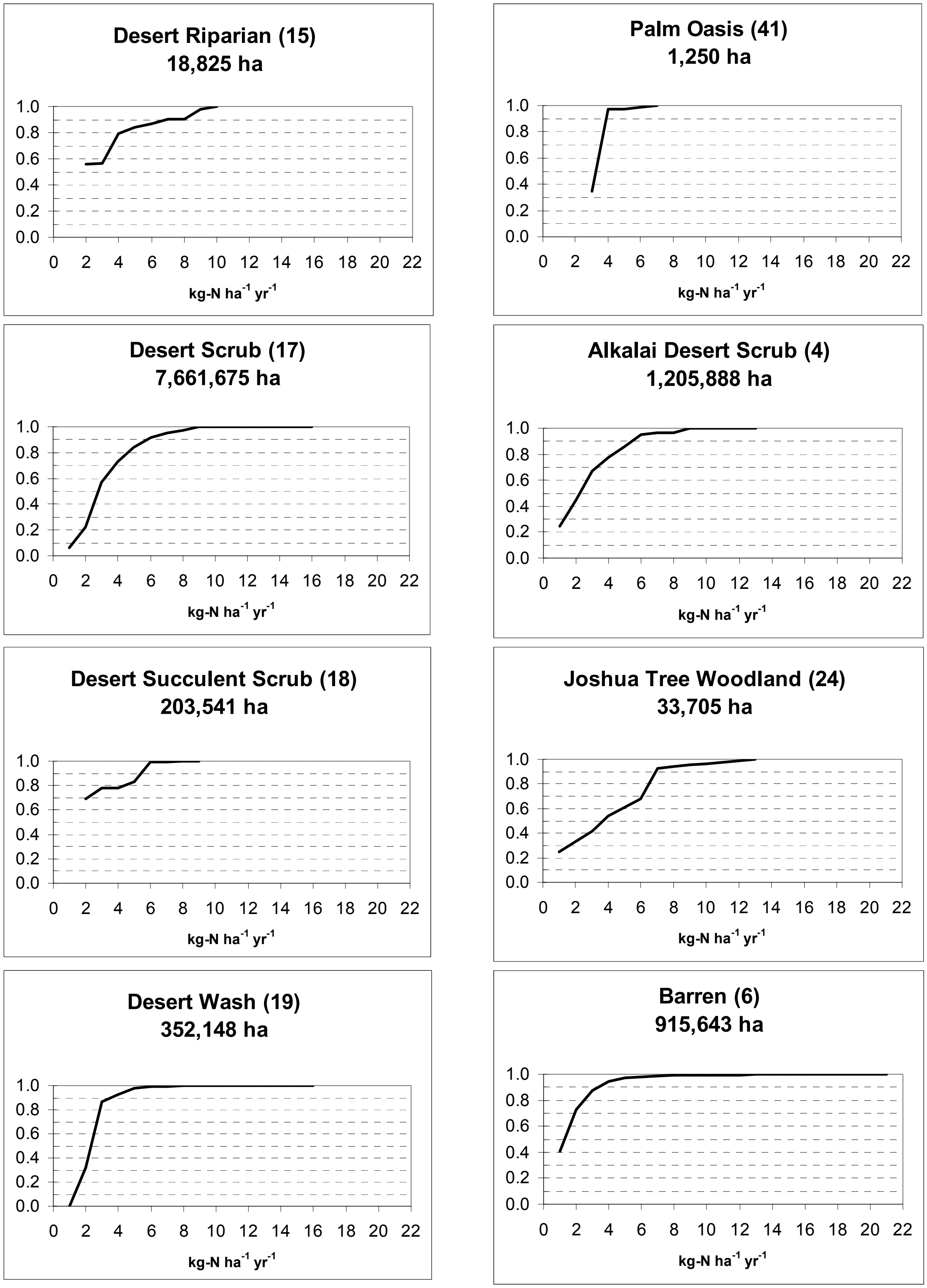


4.0 Exposure and Risks to Endangered, Threatened, and Rare Species

4.1. Methods

This section presents the results of an overlay of the CNDDB and the CMAQ 36 x 36 km map for total N-deposition in 2002. This analysis considers 1242 plant taxa in the CNDDB, including 225 taxa (species, subspecies, and varieties) that are federal- or state-listed as "threatened or endangered." The remaining 1017 taxa are regarded as rare, and include CNPS listed species (CNPS 2003). Mean exposure was calculated using all CNDDB occurrences, so that if a taxon has multiple occurrences in a single CMAQ grid square, all of those occurrences are used to derive the mean exposure. Maximum and minimum exposure across the full range of each taxa were also reported.

The same analysis is also done for the 447 animal taxa in the CNDDB, including 108 taxa (species, subspecies, and varieties) that are federal- or state-listed as "threatened or endangered," and an additional 339 taxa considered rare.

The full results are presented in Appendix B, which is in a spreadsheet format that can be filtered and searched for specific taxa.

Data are presented as CDF graphs of mean exposure and maximum exposure, so that (similar to the vegetation-type analysis) the total number of taxa above and below any given threshold can be obtained readily. The absolute numbers have been used instead of percentages. Note that the orderings of taxa for mean and maximum N-deposition exposure are different.

Note that this analysis is not appropriate for assessing site or region-specific impacts, nor is it sufficient for detailed species-specific assessment. CNDDB-type data are admittedly incomplete and have various degrees of bias, but the overall range of most taxa is at least coarsely accurate. The mean exposure is the prime risk criteria for the present analysis. The maximum exposure analysis can suggest that some part of the species range may be highly exposed, but the 36 km resolution of the CMAQ map makes definitive statements about taxon- and site-specific exposure difficult, until the 4 km CMAQ map becomes available in 2006. The problem is especially acute in near-coastal areas with steep pollution gradients, but local hotspots will undoubtedly be found in nearly many regions of the state.

Information on life history and habitat was compiled for 389 plant taxa with exposure > 5 kg-N ha^{-1} yr^{-1}. This threshold represents the lowest critical loads established for European grasslands (Bobbink and Roelofs 1995), and *serves only as benchmark for coarse screening at present*, and identifies relatively high pollution areas in California according to the 36 km CMAQ map. To reemphasize, this report's authors do not yet know the critical loads for California ecosystems, let alone loads that threaten any individual plant taxa. The data can be reanalyzed for any chosen threshold. Life history and habitat were obtained from Calflora and the online *Jepson Manual*; habitat was identified as best as possible from these descriptions. Identification of special soil types—serpentines, limestones, pebble plains, gabbros, and Ione clays—is included in habitat when noted, so that soil endemics (see Section 2.7.10.) can be mapped out. Habitat and life history factors are presented in tables for selected groups of plants.

4.2. Results

4.2.1. Plant taxa

A substantial fraction of the 225 threatened and endangered (T&E) plant taxa are exposed to elevated N-deposition (Figure 11). There are 126 taxa below the 5 kg-N ha^{-1} yr^{-1} mean benchmark, and 99 above. There are 6 T&E plant taxa above the 10 kg-N ha^{-1} yr^{-1} mean benchmark.

For maximum exposure, 93 taxa are below and 132 taxa are above 5 kg-N ha^{-1} yr^{-1}, and 31 are above 10 kg-N ha^{-1} yr^{-1} (Figure 12). Note again that any benchmark may be chosen on these graphs.

Similar proportions apply to the 1017 listed rare taxa. There are 727 taxa below 5 kg-N ha^{-1} yr^{-1} and 290 are above (Figure 13). There are 24 taxa above 10 kg-N ha^{-1} yr^{-1}. For maximum exposure, 597 taxa are below and 420 taxa are above 5 kg-N ha^{-1} yr^{-1} (Figure 14), and 72 are above 10 kg-N ha^{-1} yr^{-1}.

The map of occurrences of T&E taxa with mean exposure > 5 kg-N ha^{-1} yr^{-1} clearly show concentrations in the high N-deposition regions: Southern California, the floor and east side of the Central Valley, and the San Francisco Bay Area (Figure 15).

It is beyond the scope of this report to discuss individual plant taxa, given the high numbers in the analysis. All CNDDB plant taxa are listed in Appendix B, along with mean, maximum, and minimum N-deposition, initial habitat assignment for the higher exposure plants, federal status, state status, and global and state ranks according to The Nature Conservancy. Note that this list provides only a starting point for regional and local assessments, especially assignments to specific vegetation types.

A breakdown of life form of listed taxa exposed to > 5 kg-N ha^{-1} yr^{-1} (Table 2) shows that most listed taxa are perennial and annual forbs (including several hemiparasitic taxa), followed by shrubs, and then a variety of other life-forms. Annual forbs may be the most immediately vulnerable to annual grass invasions, but in the long run, perennial forbs and shrubs may be at risk from habitat conversion via fire. Assignment of quantitative risk factors based on life history will eventually require a taxon-by-taxon analysis.

A breakdown by habitat (Table 3) shows that 23 T&E plant taxa and 22 rare taxa are vernal pool dependent. Vernal pool taxa are concentrated on the east side of the Central Valley, the Southern California Coast, and the North Bay Area (Figure 16). Assignment of taxa to specific vegetation types will require a regional scale assessment by local experts; available data (CalFlora and Jepson Herbarium) were insufficiently precise for systematic use in this report.

Many other taxa are in low-biomass habitats that are at risk from annual grass invasions, including sandy soils, clay, grasslands, open areas, and meadows, among others. There are sets of taxa that are specialized on particular soils; these soil endemics with mean exposure > 5 kg-N ha^{-1} yr^{-1} include: serpentines in the Bay Area, gabbro; Ione clays, and serpentine in the Sierra Foothills; limestone in the San Bernardino Mountains; and metavolcanics east of San Diego (Figure 17).

As mentioned above, these analyses are constrained by the coarse resolution of the 36 km CMAQ map, especially in coastal areas. Subregional patterns will be resolved with finer resolution N-deposition modeling from the 4 km map. Note also that some highly exposed plant taxa have outliers in low N-deposition regions.

Once again, the results indicate a need for regional and subregional analyses, and Appendix B provides a starting point. Specific treatment of more than a few taxa is beyond the scope of this report.

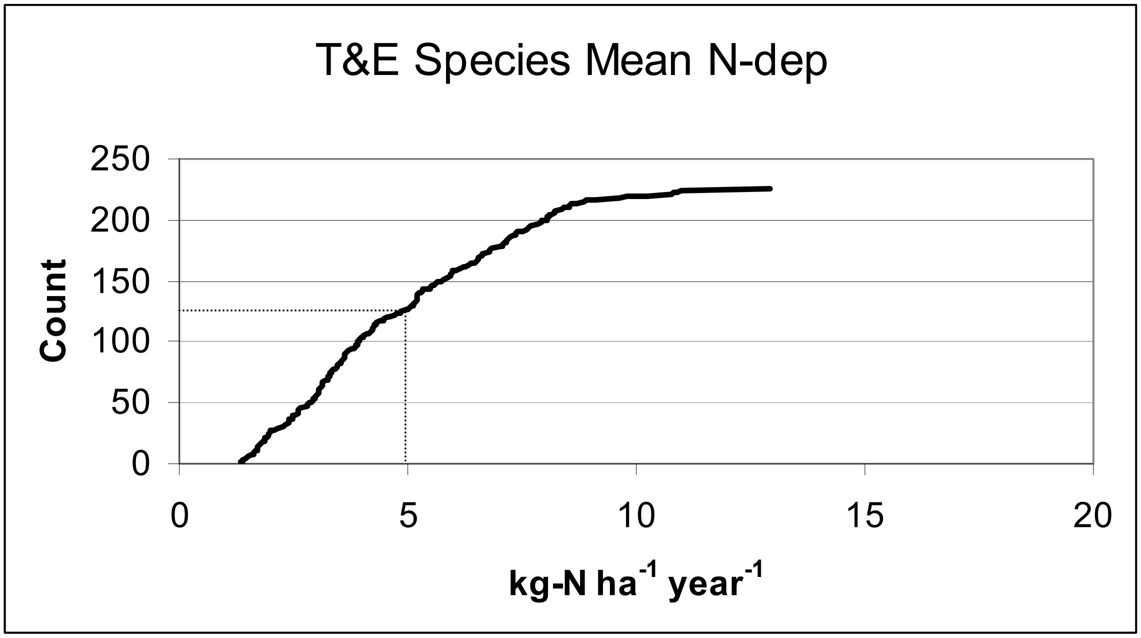


Figure 11. Average N-deposition exposure, state- and federal-listed T&E plant taxa (n = 225)

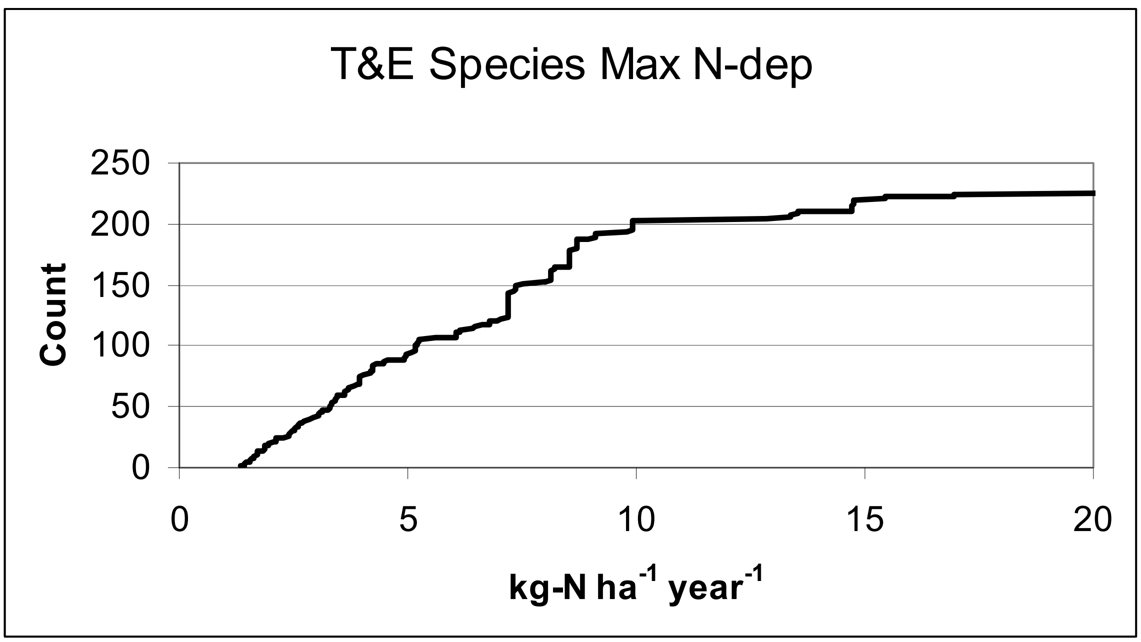


Figure 12. Maximum N-deposition exposure, state- and federal-listed T&E plant taxa (n = 225)

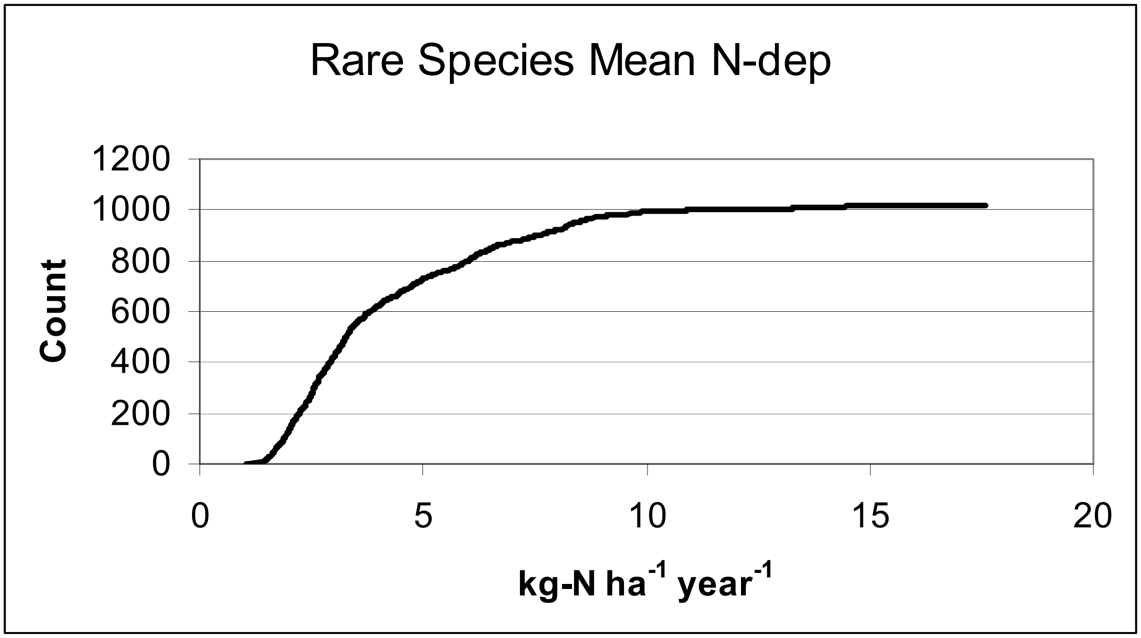


Figure 13. Mean N-deposition exposure, listed rare plant taxa (n = 1017)

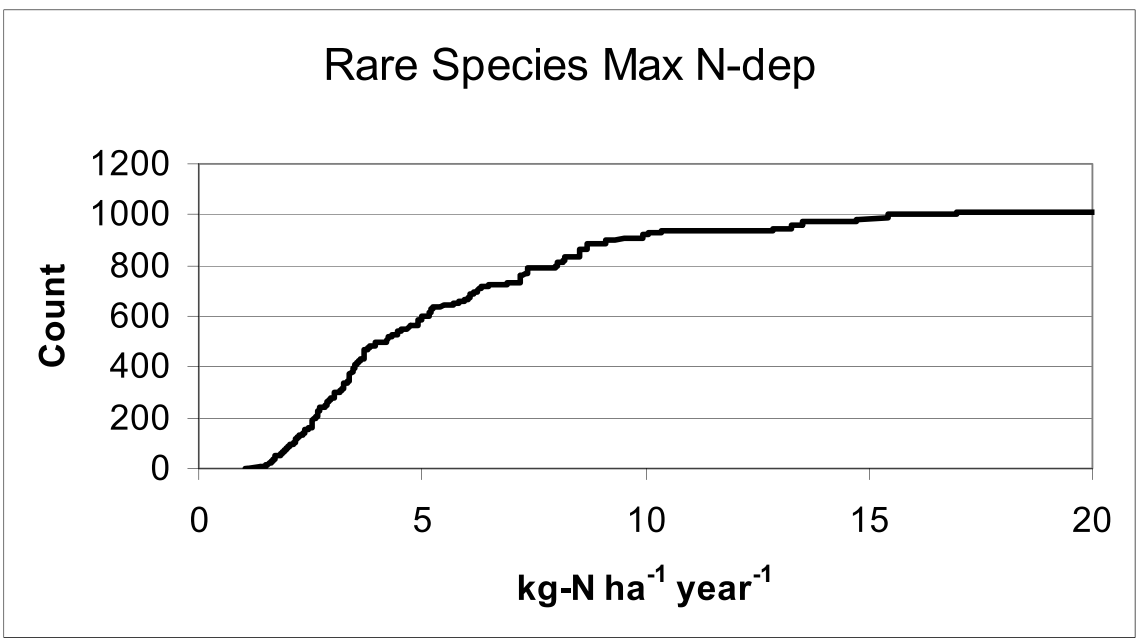


Figure 14. Maximum N-deposition exposure, listed rare plant taxa (n = 1017)

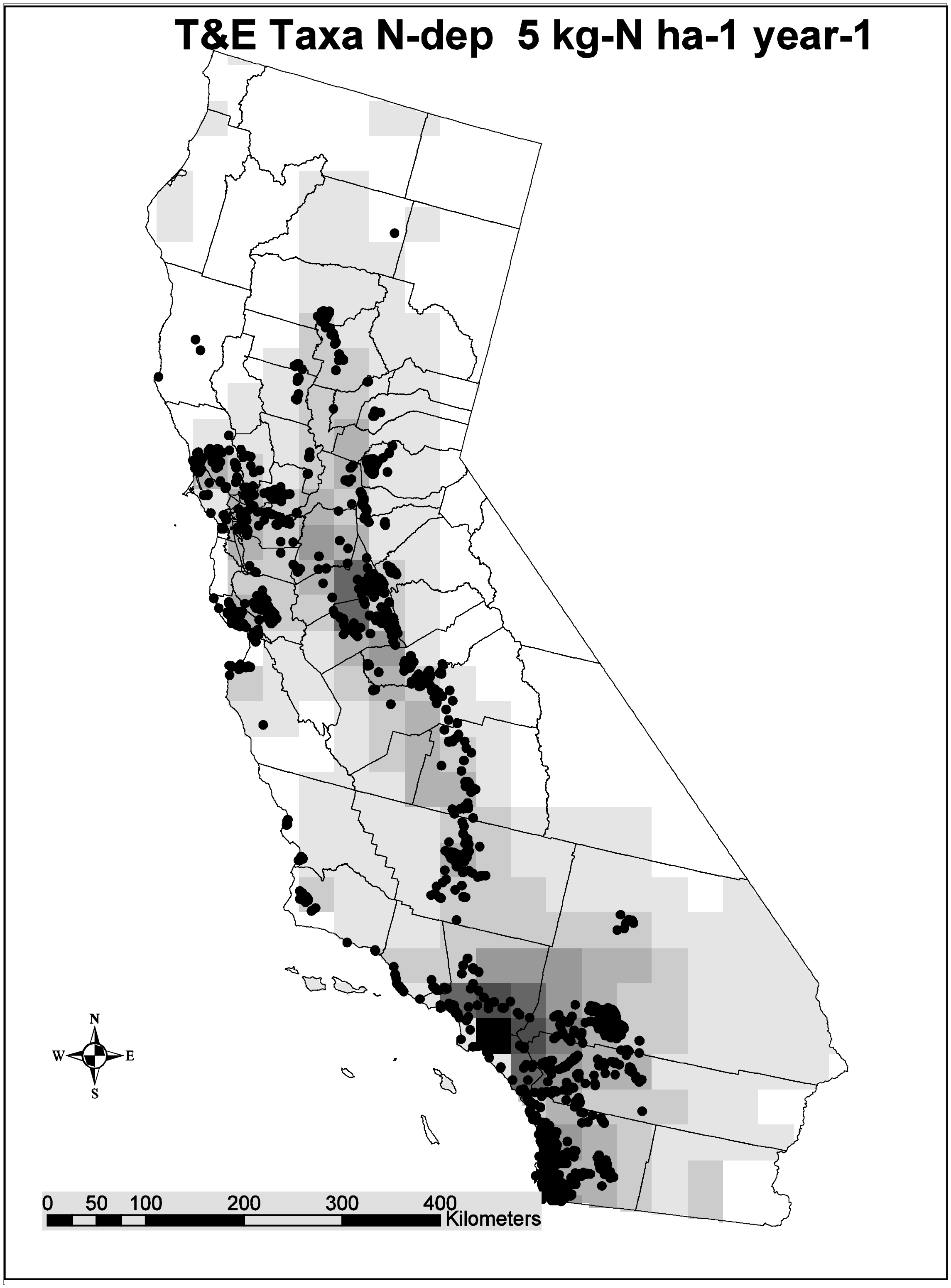


Figure 15. Distribution of federal- and state-listed T&E species exposed to > 5 kg-N ha^{-1} year^{-1}

Table 2. Life history exposure > 5 kg-N ha-1 yr^{-1}

Life Form	T&E	Rare	Total
Perennial forb	38	122	160
Annual forb	35	93	128
Shrub	10	41	51
Annual grass	7	2	9
Annual forb, hemiparasitic	4	4	8
Annual-Perennial forb	3	5	8
Tree	1	6	7
Perennial cactus	1	4	5
Perennial sedge		4	4
Perennial fern		3	3
Perennial Forb parasitic		2	2
Annual rush		1	1
Duckweed		1	1
Perennial grass		1	1
Perennial rush		1	1
Total	**99**	**290**	**389**

Table 3. Habitats of plant taxa exposed to > 5 kg-N ha^{-1} yr^{-1}

Habitat	T&E	Rare	Total
(blank)	17	58	72
Rocky	6	41	47
Vernal pools	23	22	45
Sandy		25	25
Open areas	1	18	19
Serpentine	8	11	19
Meadows	5	13	18
Alkali	1	13	14
Dry soils	1	12	13
Clay	5	7	12
Pebble-plain	2	8	10
Riparian	1	9	10
Dunes	4	4	8
Freshwater-marsh	3	5	8
Washes		8	8
Limestone	3	3	6
Disturbed	1	4	5
Gabbro	3	2	5
Salt marsh	3	2	5
Understory		5	5
Granite soils		4	4
Grassland	2	2	4
Ione clays*	3	1	4
Playas		3	3
Alluvial fans	2		2
Lake-margins	1	1	2
Sandstone	1	1	2
Scrub	2		2
Bogs, seeps	1	3	4
Bluffs		1	1
Exposed sites		1	1
Metavolcanic		1	1
Non-native**		1	1
Ponds		1	1
Grand Total	**99**	**290**	**389**

* See Section 2.7.10
** There is some doubt as to whether this one rare species is native or non-native.

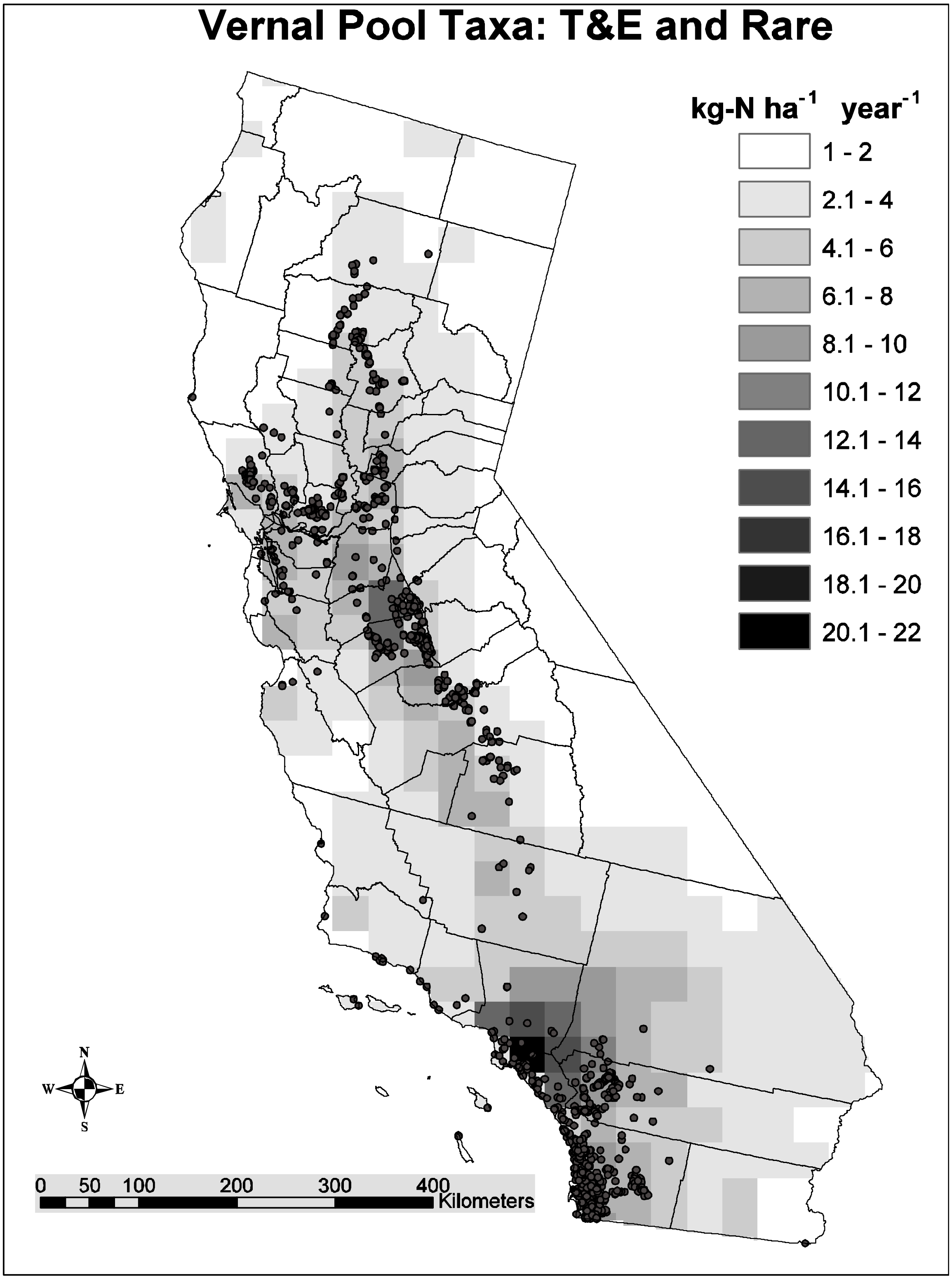


Figure 16. Location of vernal pool taxa exposed to mean > 5 kg-N ha^{-1} yr^{-1}

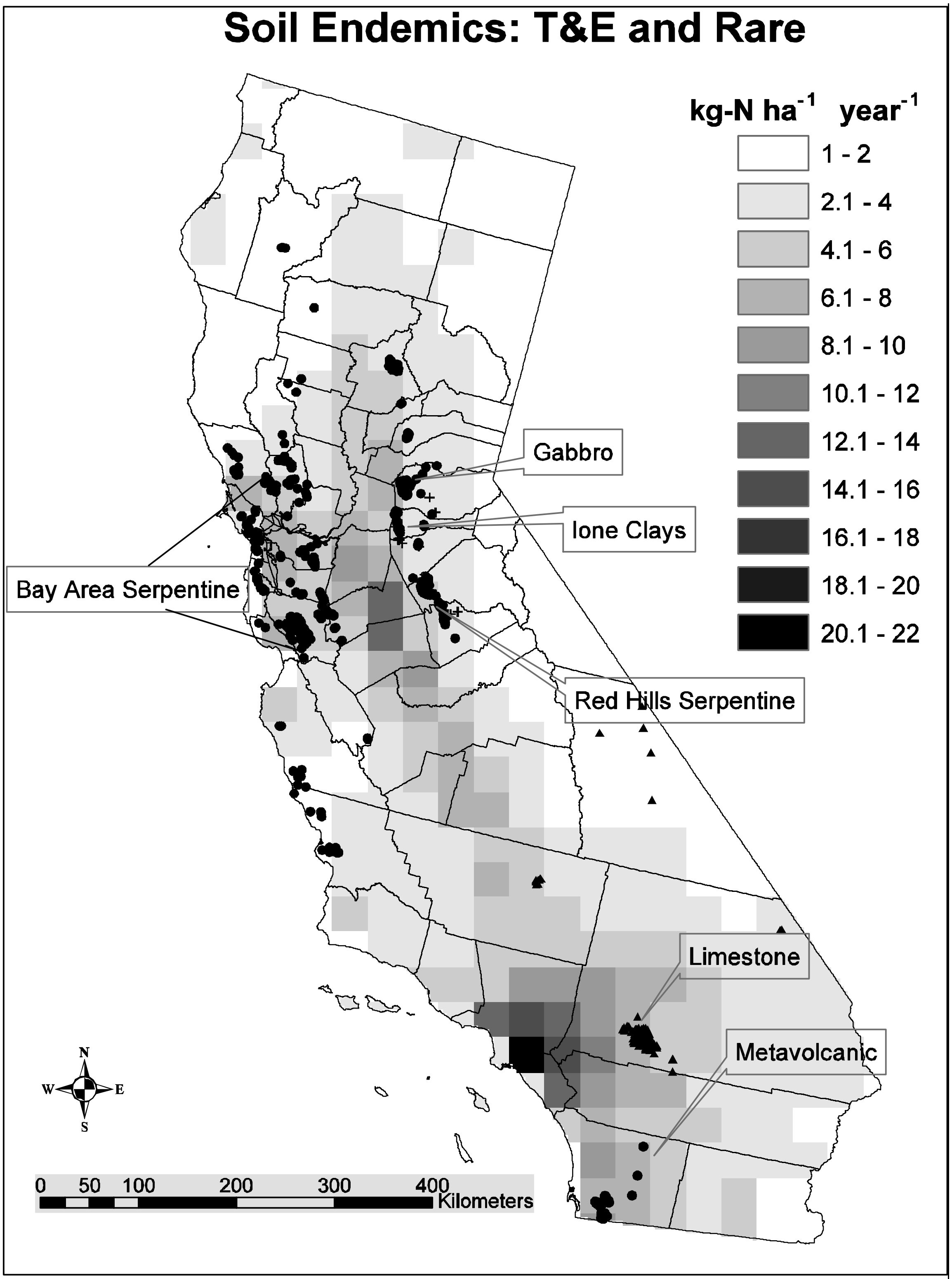


Figure 17. Locations of soil endemic plant taxa exposed to mean > 5 kg-N ha^{-1} yr^{-1}

4.2.2. Animal taxa

The exposure of 108 T&E animal taxa is roughly parallel to that of plants. There are 62 animal taxa below the 5 kg-N ha^{-1} yr^{-1} mean threshold, and 46 above (Figure 18). There are 4 T&E animal taxa above the 10 kg-N ha^{-1} yr^{-1} mean threshold. For maximum exposure, 40 taxa are below and 68 taxa are above 5 kg-N ha^{-1} yr^{-1}, and 28 are above 10 kg-N ha^{-1} yr^{-1} (Figure 19).

The exposure of 339 rare animal taxa is similar (Figure 20). There are 217 rare animal taxa below the 5 kg-N ha^{-1} yr^{-1} mean threshold, and 122 above. There are 5 rare animal taxa above the 10 kg-N ha^{-1} yr^{-1} mean threshold. For maximum exposure, 163 taxa are below and 176 taxa are above 5 kg-N ha^{-1} yr^{-1}, and 61 are above 10 kg-N ha^{-1} yr^{-1} (Figure 21). The geographic distribution of exposed animal taxa is virtually the same as that of the plants, so no map has been prepared.

The CNDDB listed animal species have broad taxonomic representation (Table 4), as do those exposed to > 5 kg-N ha^{-1} yr^{-1}. Species-by-species accounts are beyond the scope of this report.

Vulnerability to N-deposition via grass invasions is most likely in several circumstances. Butterflies and other herbivorous insects are vulnerable to displacement of larval hostplants and nectar sources by annual grasses. These butterflies include: the Bay Checkerspot (*Euphydryas editha bayensis*), in serpentine grassland with mean N-deposition exposure of 5.1 kg-N ha^{-1} yr^{-1}; the Quino Checkerspot (*E. editha quino*), in coastal sage scrub and grassland with mean N-deposition exposure of 6.9 kg-N ha^{-1} yr^{-1}; and Lange's metalmark (*Apodemia mormo langei)* in the Antioch Dunes with mean exposure of 5.2 kg-N ha^{-1} yr^{-1}. The Delhi Sands flower-loving fly (*Rhaphiomidas terminatus abdominalis*) is the most highly exposed animal with mean exposure of 13.7 kg-N ha^{-1} yr^{-1}.

Highly exposed vernal pool invertebrates include various taxa of fairy shrimp; Riverside fairy shrimp (*Streptocephalus woottoni*, mean 9 kg-N ha^{-1} yr^{-1}), San Diego fairy shrimp (*Branchinecta sandiegonensis*, mean 8.2 kg-N ha^{-1} yr^{-1}), Conservancy fairy shrimp (*Branchinecta conservatio*, mean 7.7 kg-N ha^{-1} yr^{-1}), vernal pool tadpole shrimp (*Lepidurus packardi*, mean 7 kg-N ha^{-1} yr^{-1}), Longhorn fairy shrimp (*Branchinecta longiantenna*, mean 6.5 kg-N ha^{-1} yr^{-1}), and vernal pool fairy shrimp (*Branchinecta lynchi*, mean 6.0 kg-N ha^{-1} yr^{-1}) are all vulnerable to grass invasions that shorten the inundation periods of pools (Marty 2005). California red-legged frogs *(Rana aurora draytonii*, mean 5 kg-N ha^{-1} yr^{-1}) and Tiger salamanders (*Ambystoma californiense*, mean 6.1 kg-N ha^{-1} yr^{-1}) often breed in vernal pools and are also highly susceptible to shortened inundation periods.

Animal species dependent on coastal sage scrub, such as the coastal California gnatcatcher (*Polioptila californica californica*, mean 8.7 kg-N ha^{-1} yr^{-1}) are vulnerable to habitat conversion to annual grassland. Animal species dependent on desert scrub may also be vulnerable to habitat conversion.

Threatened and endangered animal taxa and mean, maximum, and minimum N-deposition exposure are listed in Appendix B.

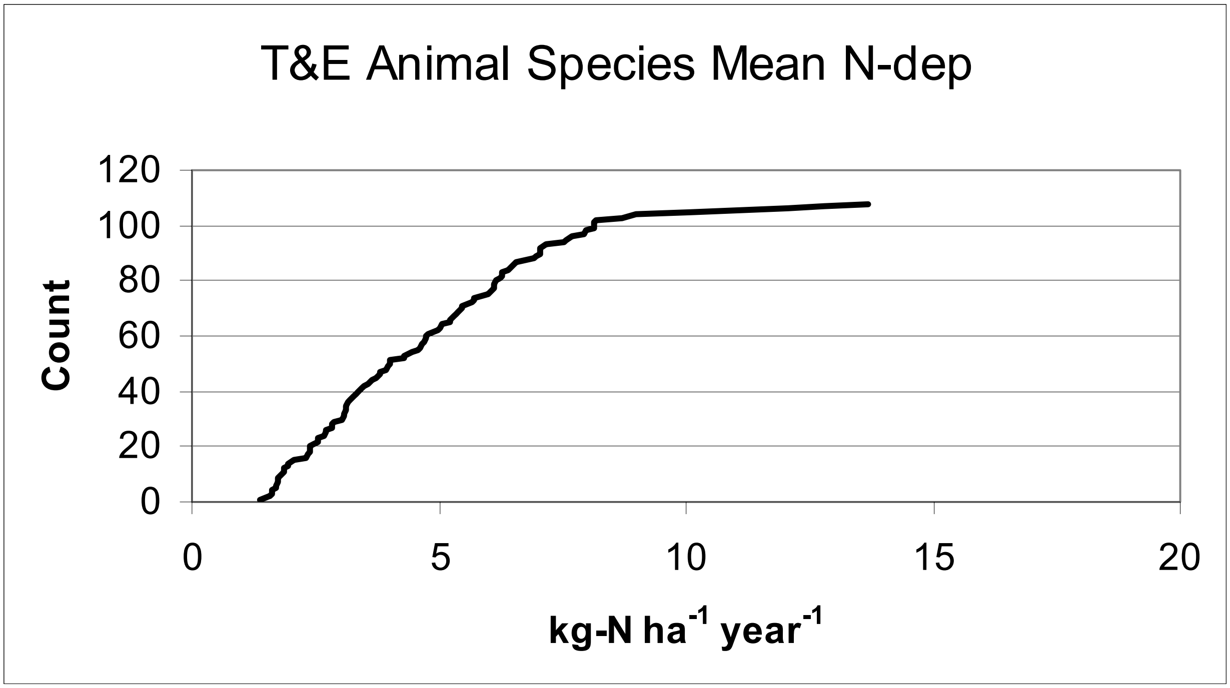


Figure 18. Average N-deposition exposure, state- and federal-listed T&E animal taxa (n = 108)

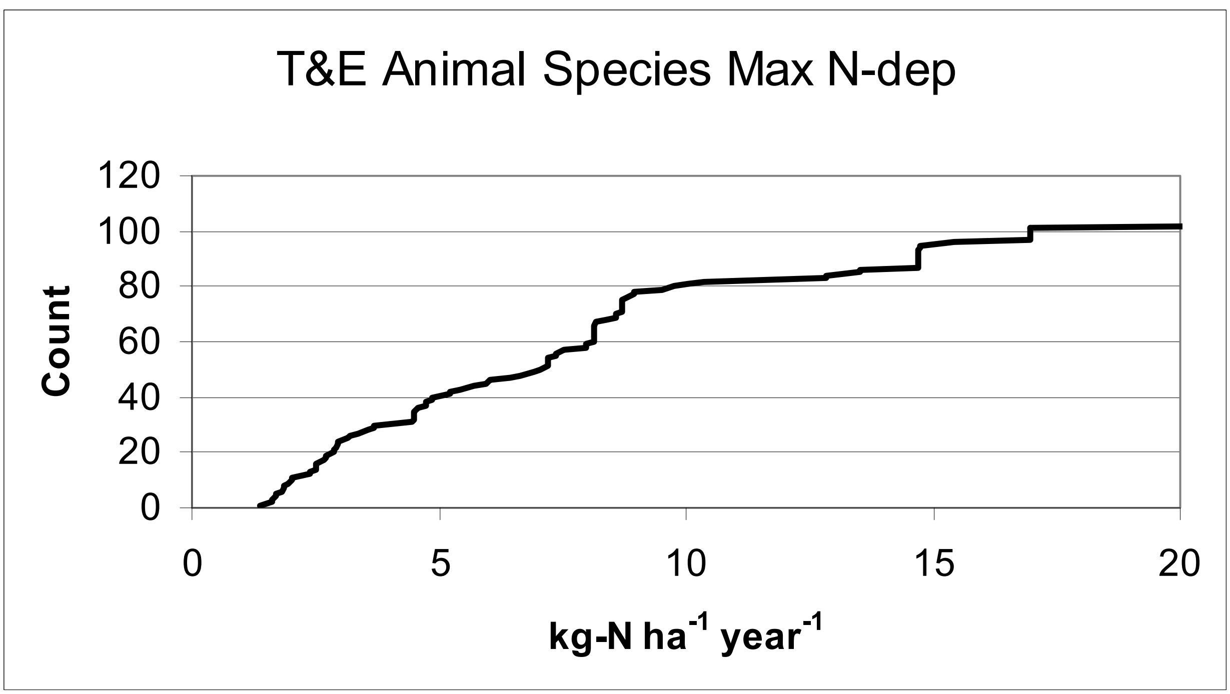


Figure 19. Maximum N-deposition exposure, state- and federal-listed T&E animal taxa (n = 108)

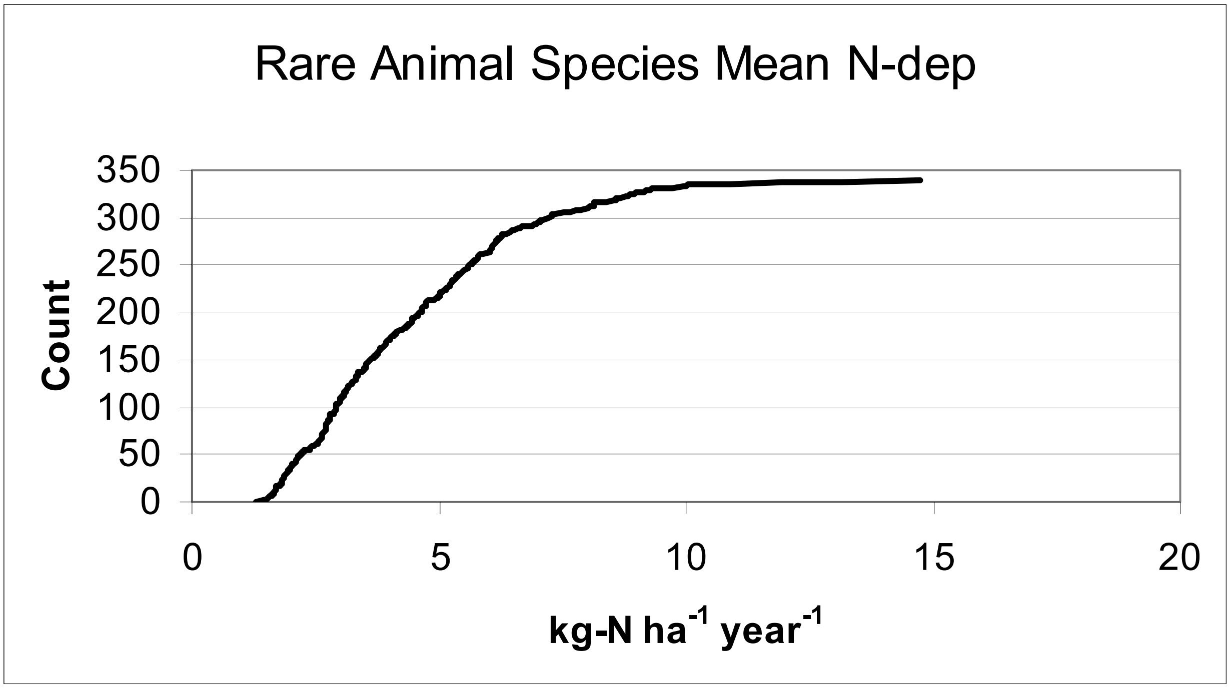


Figure 20. Mean N-deposition exposure, state- and federal-listed rare animal taxa (n = 339)

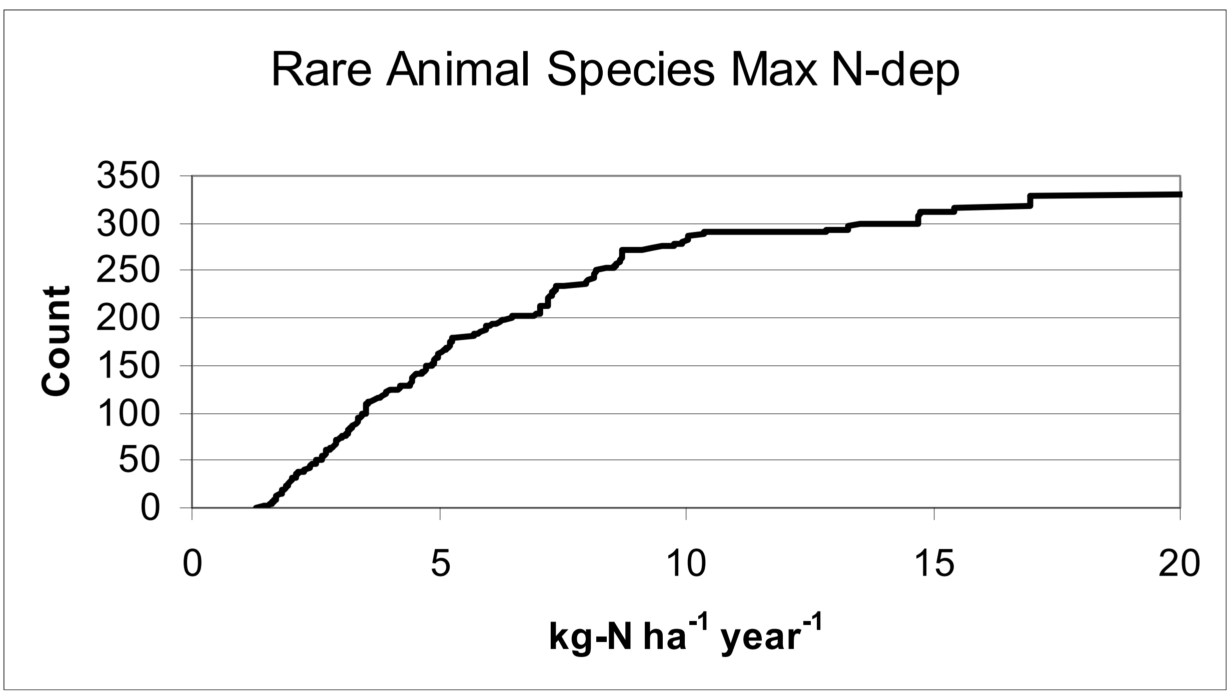


Figure 21. Maximum N-deposition exposure, state- and federal-listed rare animal taxa (n = 339)

Table 4. Taxonomic composition of T&E and rare animals

Life Form	T&E All	Rare All	T&E > 5 kg-N	Rare > 5 kg-N
Fish	26	35	6	6
Bird	25	65	8	28
Insect	19	59	9	22
Mammal	17	62	9	27
Invertebrate	9	60	7	10
Reptile	7	25	3	19
Amphibian	5	32	4	10
Grand Total	**108**	**339**	46	122

5.0 Policy Implications

There is broad scientific consensus that atmospheric nitrogen deposition profoundly changes functioning of ecosystems, which can lead to losses of biological diversity in both terrestrial and aquatic ecosystems (Vitousek 1994; Vitousek, Aber et al. 1997; Fenn, Poth et al. 1998; Galloway, Cowling et al. 2002; Matson, Lohse et al. 2002; Galloway, Aber et al. 2003). A recent synthesis of N-deposition effects in the Western United States (Fenn, Baron et al. 2003; Fenn, Haeuber et al. 2003) documents impacts on numerous California ecosystems. Large areas of California are exposed to highly elevated N-deposition, and the 36 km CMAQ map captures the geographic distribution at a regional level. In this report, the broad-scale overlays of 36 km CMAQ N-deposition with vegetation-types and special status species illustrate the broad threat that N-deposition poses to biodiversity across much of California.

The best documented mechanism for biodiversity impacts is the enhanced invasion of introduced annual grasses, which directly crowd out native species, shorten the fire cycle, and alter hydrology, microclimate, and nutrient cycling (D'Antonio and Vitousek 1992). These effects have been documented and explicitly linked to N-deposition in coastal sage scrub, serpentine grassland, and desert scrub (Fenn, Baron et al. 2003). Annual grass invasions also threaten vernal pools (Marty 2005), and are likely enhanced by N-deposition. Species that may be at risk include many narrowly distributed endemic plants that inhabit nutrient-poor soil types or microsites. Animals that depend on specific plants, hydrologic regimes, or vegetation structure are at risk in the sensitive habitat types. While annual grass invasions are well-documented, N-deposition may be enhancing the spread of numerous other weeds.

There are two routes toward minimizing and mitigating N-deposition impacts on California biodiversity: (1) decreasing N_r emissions into the atmosphere, and (2) preserving and managing sensitive habitats.

5.1. Minimizing N-deposition Impacts Via Emissions Controls

Despite the complexities of N-deposition as a process extending from initial emissions through atmospheric transport and chemical transformations; dry-and wet-deposition; changes in ecosystem function, structure, and biodiversity; and cascading "downstream" effects, the ultimate solution is to greatly decrease emissions. Some of the nitrogenous pollutants of concern are primary pollutants (NH_3, NO_x, and N_2O). Others are secondary pollutants (HNO_3, NO_3- particulates, and NH_{4+} particulates). Policy and regulatory strategies can differ depending on the source and mechanisms of synthesis.

Ongoing efforts to control NO_x emissions from vehicles and industrial sources have somewhat decreased atmospheric concentrations of NO_x in many regions of California, even in the face of population growth (Alexis, Delao et al. 2001). However, emissions of NH_3 are unregulated, although increasing attention is being paid to NH_3 because of its importance as a particulate matter ($PM_{2.5}$) precursor. On a statewide basis, power plants are a relatively minor component of emissions (Alexis, Delao et al. 2001), but nonetheless add both NO_x and NH_3 that will eventually deposit somewhere downwind.

Specific to mitigating power plant sources, the application of Best Available Control Technology (BACT) and purchase of pollution credits have been implemented to meet local air quality regulations (CARB 2000). Pollution credits are primarily aimed at ozone precursors (NO_x and ROG), and direct emissions of PM_{10}. The effectiveness of BACT and emissions credits in minimizing N-deposition is complicated by two factors. First, both NO_x and ROG credits may be purchased to offset ozone precursors, so that the total NO_x emissions may not be covered by emission offsets. Second, selective catalytic reduction (SCR) is recognized as the BACT, but SCR units emit NH_3 (known as *ammonia slip*), especially as catalysts age. There are no emissions credits for NH_3, nor is the additional N-deposition taken into account for NO_x credits. Ammonia emissions from the Metcalf Energy Center (MEC) project (see Table 1) were regulated to a maximum of 10 ppm, which was used in the assessment of N-deposition impacts on adjacent and downwind serpentine grassland habitats. The actual NH_3 emissions from SCR units may be substantially less than the regulated cap.

Determining the best modeling approach for site-specific deposition estimates from new power plants is the subject of the accompanying report by Tonnesen and Wang (forthcoming).

5.2. Mitigating N-deposition Impacts: Habitat Acquisition

Given current levels of N-deposition and the premise that source controls will at best lead to gradual decreases in deposition, the only feasible immediate actions for mitigation are habitat preservation, management, and research.

Identification of sensitive habitats and plant/animal taxa at risk can begin with the analyses presented in this report. The listing of taxa in the tabular data in Appendix B provides an initial start for assessment purposes. An independent search of the CNDDB should provide a relevant list of local special-status taxa. Local knowledge of habitat requirements can place each taxon into a habitat-type, and sensitivity to grass and other weed invasions and other impacts may be assessed. The increased N-deposition exposure of specific habitats can be estimated from modeling.

Preserving habitats through acquisition of fee title or easements is a standard mitigation practice. However, given that even a large power plant will only incrementally increase deposition in the polluted areas where species are at risk, the actual area of habitat protected in such a manner may be small relative to the extent of the target ecosystem. For example, mitigation for the MEC project included 47 ha (131 acres) of serpentine grassland habitat, in a 116 acre parcel adjacent to the power plant; and 6 ha (15 acres) several kilometers away, out of several thousand hectares of serpentine grassland. While transfer of any amount of land into protected status is a positive step, it was the *qualitative* impact of this mitigation—establishing a precedent that could be applied to highway construction, commercial/residential developments, and other power plants—that has provided the impetus for ongoing purchases of hundreds of hectares and the development of a Habitat Conservation Plan/Natural Communities Conservation Plan (HCP/NCCP) for Santa Clara County.

5.3. Monitoring, Adaptive Management, and Treatments

Monitoring and adaptive management of protected land is absolutely necessary, and can extend beyond land directly protected by purchase or easements. Numerous management treatments, including hand labor, targeted herbicides, soil/landscape disturbance, and fire are all worth exploring in one or more of the threatened ecosystems. The key is monitoring and using the monitoring data to inform the next round of treatment options—adaptive management is explicitly experimental and empirical.

For example, in serpentine grassland and vernal pools, moderate well-managed cattle grazing is effective in curbing annual grass invasions and maintaining native biodiversity and T&E/rare species. Grazing management was an explicit component of the MEC mitigation, along with adaptive management of grazing levels based on detailed monitoring of grassland composition.

Many conservation organizations, including The Nature Conservancy, California State Parks, East Bay Regional Park District, and the CNPS, are rethinking attitudes toward grazing management, because of empirical experience with negative impacts of *removing* grazing—primarily enhanced annual grass invasions that reduce native forb and grass cover. Management options may be limited, though. Grazing may be problematic in other ecosystems, such as coastal sage scrub, where the remnants of native forb cover may be on cryptobiotic crusts on clayey soils that are easily disturbed by cattle. Or, the invading grasses may be relatively unpalatable (red brome in deserts, for example).

There are relatively few options for managing annual grasses, besides livestock grazing. Fire may be useful in grasslands, but proper seasonal timing is essential and institutional barriers (air quality concerns, safety, and availability of trained personnel) can limit opportunities. Fire in grass-invaded shrublands is likely to exacerbate the problem and lead to habitat conversion unless restoration measures can be developed. Mowing can be effective if timed correctly, but may have a high cost/acre. Targeted, grass-specific herbicides can be used on fine scales, but broad applications are problematic because of cost, effectiveness, and regulatory concerns. Broadleaf weeds can be controlled by any number of approaches, as well.

Weed management is a regional-scale issue and contributions to Weed Management Areas and other organizations for long-term management of weed invasions may be effective mitigation for the dispersed impacts of N-deposition. Such contributions, in the form of a long-term endowment, may be preferable to buying small, expensive, and difficult to manage mitigation parcels, but these decisions need to be made on a case-by-case basis.

5.4. Research

Research can provide a basis for understanding the complexities of N-deposition impacts, and can guide management decisions. *Adaptive management* views management decisions as experiments that require ongoing evaluation. Monitoring the results of management activities is essential and drastic changes in management need careful consideration and perhaps should be implemented as small-scale experiments.

The complexities of the N-cycle at global, regional, and local scales are widely recognized in the scientific community. Examples include the First, Second, and Third International Nitrogen conferences, multiple sessions at major conferences (e.g., the American Geophysical Union, Ecological Society of America, and others), and specific symposia (e.g., Atmospheric Ammonia Workshop, N-eutrophication Symposium). Many efforts are underway to define long-term research goals for N-science, and the complete research agenda is well beyond the ability of any one agency to fully fund.— Research needs are similar in scale to the carbon-cycle science that has developed over the last decade. The research recommendations below are a small subset of the potential questions and topics that are of interest to California and the Energy Commission in particular.

5.4.1. Estimates of N-deposition

Research all along the pathway of emissions/transport/chemical transformations/ deposition is necessary to better quantify the flux of various N-species to ecosystems.

Emissions: Emission inventories are the most uncertain input into models such as CMAQ, and need continual improvement and adaptation to new circumstances. Emissions from power plants are monitored under AQ regulations, but the progression of NH_3 slip over several years under actual operating conditions is an uncertainty that could be reduced by compilation and analysis of emission records from existing SCR units in California and elsewhere, or by collecting new data. A 1-year pilot study could assess existing data and recommend if a multi-year monitoring program (3 years, at a series of power plants) would be necessary.

Modeling: The modeling research needs are dealt with in the accompanying report by Tonnesen and Wang (forthcoming). Ready availability of the 4 km model results—in monthly time steps and by N-species—for regional assessments and validation studies will greatly enhance the capacity to study N-deposition in California.

Measurements: Atmospheric concentrations of N_r species are first-order drivers of N-deposition, and can be measured at various time-intervals. Passive sampling systems economically measure time-averaged concentrations (days to weeks/months) of NO_2, NO, HNO_3, NH_3, and O_3, and can supplement existing AQ networks (Bytnerowicz, Arbaugh et al. 2003). Standardized measurement of NH_3 and HNO_3 concentrations are lacking in current AQ networks. A 1-year scoping study and pilot project on the design and implementation of regional and local passive monitoring networks in California would establish costs and protocols for an optimized network that could answer key N-deposition questions and be used to calibrate AQ models. The 4 km CMAQ output provides a first hypothesis on regional gradients to test with passive samplers.

Throughfall measurements, using ion exchange resins, is a passive method of estimating N-deposition to forests and shrublands but may not capture stomatal uptake and direct deposition to soil surfaces (Fenn and Poth 2004).

Passive flux monitors are a relatively new development (Fritz and Pisano 2002) that allows for directional sampling of total flux (wind speed x concentration) of the same gaseous species as passive samplers. Deployment of a network around a power plant, and relative to other local sources, would deconvolute sources and allow for estimation of the power plant contribution to local concentrations and deposition.

Direct measurement of atmospheric deposition of multiple N-species to various surfaces is one of the most technically challenging fields of science. Eddy-flux systems can be adapted for NH_3 and NO_y, and in conjunction with measurements of CO_2 and H_2O fluxes can establish key deposition parameters such as surface resistances and stomatal conductance under varying conditions and calibrate deposition models to specific ecosystems.

Recent advances in analyses of stable isotopes and radiocarbon provide opportunities to trace emissions sources, deposition rates, and biogeochemical processing (e.g. Kendall and McDonnell 1998). Nitrogen, oxygen, and carbon isotopes provide multivariate information to constrain and deconvolute N-budgets along the N-cascade.

The development of cost-effective biomonitors will be critical for realistic integrated measurements of N-deposition. Field deployable lysimeters—small pots with standardized species composition, soil, and isotopic composition—can potentially measure N-accumulation, isotopic composition, and effects on growth among growing seasons and across local and regional deposition gradients. It may be a challenge to separate out the effects of co-occurring pollutants, especially ozone, but careful consideration of initial lysimeter conditions, local pollution sources, and deployment patterns may overcome these limitations.

5.4.2. Ecosystem impacts

Further studies of all aspects of N-cycling and budgets in California ecosystems are critical. Such research will necessarily be complex, and include field surveys along local and regional gradients, site-specific experiments, modeling, and development of N-deposition indicators in an array of local ecosystems. These studies are more process oriented, and complement targeted surveys of annual grass and other weed impacts in high deposition areas.

Among the key questions to be addressed in an integrated manner are the following:

- How much N_r in various forms is deposited in particular ecosystems, and what are the effective differences between oxidized and reduced N forms? How does direct stomatal uptake effect plant performance compared with throughfall and root uptake?

- How is N-deposition accumulated, stored, cycled, and lost from various ecosystem components through time, especially in low-biomass systems? Key loss processes include: leaching, volatilization, trace gas emissions, denitrification, and fire. Key accumulation processes are plant uptake and storage, litter, and soil organic matter accumulation. The focus on semi-arid California ecosystems would include field measurements and applications of appropriate ecosystem models.

- What is the N-saturation status of California ecosystems? Assessment will require development of ecosystem indicators—N-content of vegetation and soils, readily measured processes that indicate enhanced N-cycling rates, repeatable changes in species composition—and application to known and suspected sensitive ecosystems.

- What are critical loads for particular ecosystems and habitats, and how do we account for the cumulative nature of N-deposition impacts? What are the broad implications for water quality as more ecosystems begin to export nitrate in surface and groundwater?

- How does N-deposition drive weed invasions? Which weed species are particularly advantaged under N-deposition, and how do weeds affect biogeochemical processes, and reduce native biodiversity? Mechanistic studies of differences in response between native species and introduced species could untangle the roles of herbivory, mycorrhizal status, and other ecological interactions in determining the likelihood of N-deposition impacts.

- What are the management and restoration options for mitigating N-deposition impacts? Local studies using good experimental designs should be part of any adaptive management program mandated by mitigation requirements. Other activities include: surveys of existing management activities—grazing and prescribed fire, especially—in a variety of ecosystems and establishment of exclosures.

5.4.3. Education and public awareness

The disruption of the N-cycle is a profound change that is relatively unknown among land managers, regulators, conservation groups, elected officials, and the public at large. A concerted effort to develop appropriate educational materials, both printed and web-based, to raise awareness of the magnitude and severity of the problem among the various groups is a key step in moving toward solutions.

5.5. Benefits to California

This research provides a systematic study of known and potential threats of N-deposition to California's biodiversity. The benefits to the state include the following:

- Recognition that N-deposition is a serious threat to biodiversity across much of the state is the first step in dealing with the problem. This report provides technical background material and an entry to the large worldwide N-deposition literature.

- The geographic analyses provide a basis for regional and local studies to further understand the problem. Understanding N-deposition as a driving force behind intensified annual grass invasions and potential intensification of other weed invasions, provides land managers with key information that can inform site-specific management to protect sensitive species and habitats.

- An outline of regulatory guidance (Section 5.6 below) provides a basis for more efficiently establishing mitigation requirements and options to meet those requirements.

- The research recommendations highlight promising and necessary steps to greater understanding of the N-deposition phenomenon and impacts, and can help make California a pioneer in addressing the issues.

5.6. Regulatory Guidance Outline

Based on the procedure followed for the Metcalf Energy Center (Section 5) and other power plant projects (Table 1) the following outline presents a synthesis of key questions to ask and possible avenues for effective mitigation measures. Many of the steps are already routine in an environmental assessment and can be applied to developing impact analysis and mitigation for N deposition.

I. Estimate additional N-deposition generated by a power plant
    A. Use maximum allowable emissions under AQ regulations for the specific plant
        1. May overestimate the actual emissions (especially SCR ammonia slip), but parallels AQ analysis
    B. Estimate spatial distribution of deposition
        1. Model choice and implementation are covered in Tonnesen and Wang (forthcoming)
        2. Background levels for 2002 will soon be available in 4 x 4 km map from Tonnesen et al.
        3. The 36 km map is not suitable for local analysis, except to identify high deposition regions

II. Assess potential impacts on local ecosystems and species
    A. Develop local list of habitat types, rank into qualitative sensitivity classes according to available data
        1. The discussion in this report provides the preliminary list, but local knowledge and expertise are essential.
        2. Consider weed threats to these habitats, especially from annual grass, but also from annual and perennial forbs and shrubs.
    B. Develop a local list of Endangered, Threatened, and Listed Species, along with habitat associations, and rank into potential sensitivity classes according to available data
        1. CNDDB inquiry for local listed species is standard in environmental review. The list of species from the CNDDB in Appendix B of this report

provides an initial screening for species-specific range-wide N-deposition exposure.

2. Finer-scale local data sources and experts should be consulted when available for habitat associations of listed species.

3. Sensitivity of particular species needs to be considered on a local scale. The criteria outlined here—overall exposure statewide from Appendix A, habitat type, life form, and rarity—can be used to rank risks in a local context.

4. Conduct initial surveys to identify potential weed threats to habitats and species.

C. Assess exposure of sensitive elements

1. Choose the most appropriate local/regional habitat maps with explicit connections between sensitive species and habitat types and set target areas.

2. Overlay local map of sensitive habitats with N-deposition exposure from model.

3. If detailed species distributions data are available, also calculate species-specific exposure.

4. Calculate a histogram of annual increment of deposition increase on habitat within areas receiving an increment greater than 0.005 kg-N ha^{-1} $year^{-1}$, the Deposition Analysis Threshold value for Class 1 areas (NPS 2001, www2.nature.nps.gov/air/permits/flag/NSDATGuidance.htm).

5. Calculate the impact as a proportional increase over background levels multiplied by the habitat area affected. However, proportional impacts will be lower in high pollution zones where impacts may already be acute, and higher in low pollution areas. This point needs careful consideration, perhaps in the framework of Prevention of Significant Deterioration (PSD).

6. Apply a mitigation ratio (U.S. Fish & Wildlife Service has used 3:1) to the impact. Mitigation ratios are commonly used for off-site mitigation—if for example, the impact is estimated to be 1 hectare, then 3 hectares of mitigation land need to be secured.

III. Evaluate mitigation options

A. Land purchases

1. If suitable examples of impacted habitat-types of sensitive species are available, then attempt to buy sufficient habitat to meet mitigation ratio.

a) Areas close to the power plant site that are predicted to have higher deposition increments are preferable to those farther away.

b) The uncertainties of the real estate market, availability of appropriate habitat, and potentially small size of mitigation parcels are complicating factors, and alternatives to purchase (section III-B) could be considered.

B. Contribution to monitoring, management , restoration, and weed control in local reserves

1. Many established local reserves are in need of targeted management money for short- and long-term weed control. The provision of endowment money specifically for this purpose so that weed control can

be implemented over areas equal to or greater than the mitigation requirement.

2. Funding for restoration of habitats sufficient to cover the mitigation requirements may be considered.

C. Contribute to research on N-deposition effects and mitigation options in the region.

1. N-deposition is a complex process, and funding for targeted research (see research priorities, Section 5.4) may be lacking. Developing methods for monitoring N-deposition, effects on ecosystems, changes in biodiversity, and restoration of degraded habitats can add to capacity for mitigating impacts.

IV. Fund and institutionalize implementation

A. Develop a Property Analysis Report (PAR) for purchased land, establish an Inventory and Capital Phase, and set aside an endowment sufficient to implement long-term monitoring and adaptive management of target species and habitat.

1. Monitoring should adhere to high scientific standards, and adaptive management should include experimental scale evaluation of options.

B. If management monies are used for weed control and management on existing reserve lands, implement monitoring and documentation of the efforts that adhere to high scientific standards.

C. Require an annual report and meeting of stakeholders.

1. Field tours during the appropriate season are important to firsthand understanding of issues.

2. When possible, coordination with other local and regional conservation entities, and adjacent landowners should be pursued.

6.0 References

Alexis, A., A. Delao, et al. (2001). *The 2001 California Almanac of Emissions and Air Quality.* Sacramento, California, California Air Resources Board.

Bakker, E. (1984). *An Island Called California.* Berkeley: University of California Press.

Barry, S. J. (1998). Managing the Sacramento Valley vernal pool landscape to sustain the native flora, in *Ecology, Conservation, and Management of Vernal Pool Ecosystems – Proceedings from a 1996 Conference.* C. W. Witham, E. T. Bauder, D. Belk, W. R. Ferren Jr., and R. Ornduff, eds. Sacramento, California. 1998. California Native Plant Society**:** 236–240.

Bell, J. N. B., and M. Treshow, eds. (2002). *Air Pollution and Plant Life.* Chichester, UK: John Wiley and Sons.

Bobbink, R., M. Hornung, et al. (1998). "The Effects of Air-Borne Nitrogen Pollutants on Species Diversity in Natural and Semi-Natural European Vegetation." *Journal of Ecology* **86**(5): 717–738.

Bobbink, R., and L. P. M. Larners (2002). Effects of increased nitrogen deposition. In *Air Pollution and Plant Life.* J. N. B. Bell, and M. Treshow, eds. Chichester, UK: John Wiley and Sons**:** 201–236.

Bobbink, R., and J. G. M. Roelofs (1995). "Critical loads for natural and semi-natural ecosystems: The empirical approach." *Water, Air, & Soil Pollution* **85**: 2413–2418.

Boehm, A. B., G. G. Shellenbarger, et al. (2004). "Groundwater Discharge: Potential Association with Fecal Indicator Bacteria in the Surf Zone." *Environmental Science and Technology* **38**: 3558–3566.

Brooks, M. L. (2003). "Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert." *Journal of Applied Ecology* **40**: 344–353.

Bull, K. R. (1992). "An introduction to critical loads." *Environmental Pollution* **77**(2-3): 173–176.

Bull, K. R., and M. A. Sutton (1998). "Critical loads and the relevance of ammonia to an effects-based nitrogen protocol." *Atmospheric Environment* **32**(3): 565–572.

Bytnerowicz, A., M. J. Arbaugh, et al., eds. (2003). *Ozone Air Pollution in the Sierra Nevada: Distribution and Effects on Forests.* Developments in Environmental Science. Oxford, UK.

California Energy Commission. (1997a). Commission decision: Application for Certification for the Sutter Power Plant Project (97-AFC-2). CEC Publication Number: P800-99-010.

California Energy Commission. (1997b). Commission decision: Application for Certification for the Los Esteros Critical Energy Facility Project (01-AFC-12). CEC Publication Number: P800-02-005.

California Energy Commission. (2001a). Commission decision: the Metcalf Energy Center: Application for Certification (99-AFC-3), Santa Clara County. CEC Publication Number: P800-01-023. 569 pp.

California Energy Commission. (2001b). Commission decision - Application for Certification - Otay Mesa Generating Project (PG&E National Energy Group) (99-AFC-5). CEC Publication Number: P800-01-014

California Energy Commission. (2003). Commission decision: Application for Certification for the PICO Power Project, Santa Clara County (02-AFC-3). CEC Publication Number: P800-03-014.

CARB (2000). Offset Packages for Licensed Power Plant Projects. Sacramento, California**:** www.arb.ca.gov/energy/powerpl/guidedoc/offsets.pdf.

CNPS (2003). *CNPS Inventory of Rare and Endangered Plants.* Sacramento, California, California Native Plant Society.

Collett, J. L., B. C. Daube, et al. (1990). "The chemical composition of intercepeted cloudwater in the Sierra Nevada." *Atmospheric Environment* 24A:959-972.

D'Antonio, C., and P. M. Vitousek (1992). "Biological invasions by exotic grasses, the grass/fire cycle, and global change." *Annual Review of Ecology and Systematics* **3**: 63–87.

De Boer, W., and G A. Kowalchuk. (2001). "Nitrification in acid soils: Microorganisms and mechanisms." *Soil Biology and Biochemistry* 33: 853–866.

Egerton-Warburton, L. M., and E. B. Allen (2000). "Shifts in arbuscular mycorrhizal communities along an anthropogenic nitrogen deposition gradient." *Ecological Applications* **10**: 484–496

Fenn, M. E., M. A. Poth, et al. (1998). "Nitrogen Excess in North American Ecosystems: Predisposing Factors, Ecosystem Responses, and Management Strategies." *Ecological Applications* **8**(3): 706–733.

Fenn, M. E., M. A. Poth, et al. (2000). "Throughfall and fog deposition of nitrogen and sulfur at an N-limited and N-sautirate site in the San Bernardino Mountains, Southern California." *Canadian Journal of Forest Research* 30:1476–1488.

Fenn, M., M. A. Poth, et al. (2003). Effects of ozone, nitrogen deposition, and other stressors on montane ecosystems in the Sierra Nevada. In *Ozone Air Pollution in the Sierra Nevada: Distribution and Effects on Forests.* A. Bytnerowicz, M. J. Arbaugh, and R. Alonso, eds. Oxford, UK: ElsevierScience, Ltd. **2:** 111–155.

Fenn, M. E., and M. A Poth (2004). "Monitoring nitrogen deposition in throughfall using ion exchange resin columns: A field test in the San Bernardino Mountains." *Journal of Environmental Quality* 33:2007–2014.

Fenn, M. E., J. S. Baron, et al. (2003). "Ecological Effects of Nitrogen Deposition in the Western United States." *Bioscience* **53**: 404–420.

Fenn, M. E., R. Haeuber, et al. (2003). "Nitrogen Emissions, Deposition, and Monitoring in the Western United States." *BioScience* **53**(4): 391–403.

Fowler, D. (2002). Pollutant deposition and uptake by vegetation. *Air Pollution and Plant Life.* J. N. B. Bell and M. Treshow, eds. Chichester, UK, John Wiley and Sons**:** 43–68.

Fritz, D., and J. Pisano (2002). *Flux Measurements of Ammonia to Estimate Emission Factors for Area Sources.* Sacramento, California. California Air Resources Board.

Galloway, J. N., J. D. Aber, et al. (2003). "The nitrogen cascade." *Bioscience* **53**: 341–356.

Galloway, J. N., E. B. Cowling, et al. (2002). "Reactive Nitrogen: Too Much of a Good Thing?" *AMBIO: A Journal of the Human Environment* **31**(2): 60–63.

Gerhardt, F., and S. K. Collinge (2003). "Exotic plant invasions of vernal pools in the Central Valley of California, USA." *Journal of Biogeography* **30**: 1043–1052.

Grosjean, D. and A. Bytnerowicz. 1993. "Nitrogenous air pollutants at a southern California mountain forest smog receptor site." *Atmospheric Environment* 27A: 483-492.

Huenneke, L, S. P. Hamburg, R. Koide, H. A. Mooney, and P. M. Vitousek. (1990). "Effects of soil resources on plant invasion and community structure in Californian serpentine grassland." *Ecology* 71: 478-491.

Harrison, S., B. D. Inouye, et al. (2003). "Ecological Heterogeneity in the Effects of Grazing and Fire on Grassland Diversity." *Conservation Biology* **17**(3): 837–845.

Hickman, J. C., ed. (1993). *The Jepson Manual: Higher Plants of California.* Berkeley, California, University of California Press.

Hooper, D. U., and L. Johnson (1999). "Nitrogen limitation in dryland ecosystems: Responses to geographical and temporal variation in precipitation." *Biogeochemistry* **46**(1-3): 247–293.

Jassby, A. D., J. E. Reuter, et al. (1994). "Atmospheric deposition of nitrogen and phosphorous in the annual nutrient load of Lake Tahoe (California-Nevada)." *Water Resources Research* **30**: 2217–2236.

Jones, M. B. (1963) "Yield, percent nitrogen, and total nitrogen uptake of various California annual grassland species fertilized with increasing rates of nitrogen." *Agronomy Journal* 55:254-257.

Jones, M. B., and R .A. Evans. (1960). "Botanical composition changes in annual grassland as affected by fertilization and grazing." *Agronomy Journal* 52:459–461.

Keeley, J. E., M. B. Keeley., and C. J. Fotheringhamb. 2005. "Alien plant dynamics following fire in Mediterranean-climate California shrublands." *Ecological Applications* 15:2109–2125.

Kendall, C., and McDonnell, J. J., (eds), (1998). *Isotope Tracers in Catchment Hydrology*, ElsevierScience Publishers, 839 p.

Jovan, S. and McCune, B. 2005. Air-quality bioindication in the greater Central Valley of California, with epiphytic macrolichen communities. *Ecological Applications* 15; 1712-1726.

Langran, S. J., Ed. (1999). *The Impact of Nitrogen Deposition on Natural and Semi-natural Ecosystems.* Environmental Pollution. Dordrecht, Netherlands, Kluwer Academic Publishers.

Mansfield, T. A. (2002). Nitrogen oxides: Old problems and new challenges. *Air Pollution and Plant Life.* J. N. B. Bell and M. Treshow, eds. Chichester, UK: John Wiley and Sons**.** 119–133.

(Marshall, J.D. and S.H. Cadle, (1989). "Evidence for trans-cuticular uptake of HNO_3 vapor by. foliage of eastern white pine (Pinus strobus L.)." *Environmental Pollution* 60:12-28).

Marty, J. (2005). "Cattle promote diversity in ephemeral wetlands." *Conservation Biology* 19:1626-1632.

Matson, P., K. A. Lohse, et al. (2002). "The Globalization of Nitrogen Deposition: Consequences for Terrestrial Ecosystems." *AMBIO: A Journal of the Human Environment* **31**(2): 113–119.

Metcalfe, S. E., D. Fowler, et al. (1998). Spatial and temporal aspects of nitrogen deposition. In *The Impact of Nitrogen Deposition on Natural and Semi-natural*

Ecosystems. S. J. Langran. Dordrecht, Netherlands: Kluwer Academic Publishers**.** 15–50.
Morris, J. T. (1991). "Effects of Nitrogen Loading on Wetland Ecosystems with Particular Reference to Atmospheric Deposition." *Annual Review of Ecology and Systematics* **22**: 257–279.
Schlesinger, W. H. (1997). *Biogeochemistry: An Analysis of Global Change.* San Diego, California: Academic Press, Inc.
Silviera, J. G. (2000). "Vernal pools and relict duneland at Arena Plains." *Fremontia* **27-28**: 38–47.
Sommer, S. G., and E. S. Jensen (1991). "Foliar absorption of atmospheric ammonia by ryegrass in the field." *Journal of Environmental Quality* **20**: 153–156.
Stevens, C. J., N. B. Dise, et al. (2004). "Impact of Nitrogen Deposition on the Species Richness of Grasslands." *Science* **303**: 1876–1879.
Tonnesen, G., and Z. Wang. Forthcoming. *Assessment of Nitrogen Deposition: Modeling and Habitat Assessment.* California Energy Commission, PIER Energy-Related Environmental Research.
USFWS (1998). Recovery Plan for Serpentine Soil Species of the San Francisco Bay Area. Portland, Oregon**:** 330+.
Vitousek, P. M. (1994). "Beyond Global Warming: Ecology and Global Change." *Ecology* **75**(7): 1861–1876.
Vitousek, P. M., J. D. Aber, et al. (1997). "Technical Report: Human Alteration of the Global Nitrogen Cycle: Sources and Consequences." *Ecological Applications* **7**(3): 737–750.
Weiss, S. B. (1999). "Cars, Cows, and Checkerspot Butterflies: Nitrogen Deposition and Management of Nutrient -Poor Grasslands for a Threatened Species." *Conservation Biology* **13**(6): 1476–1486.
Yoshida, L. C., and E. B. Allen. (2004). 15N uptake by mycorrhizal Artemisia californica and the invasive Bromus madritensis of a N-eutrophied shrubland. *Biology and Fertility of Soils* 39:243–248.

<u>Personal Communications</u>

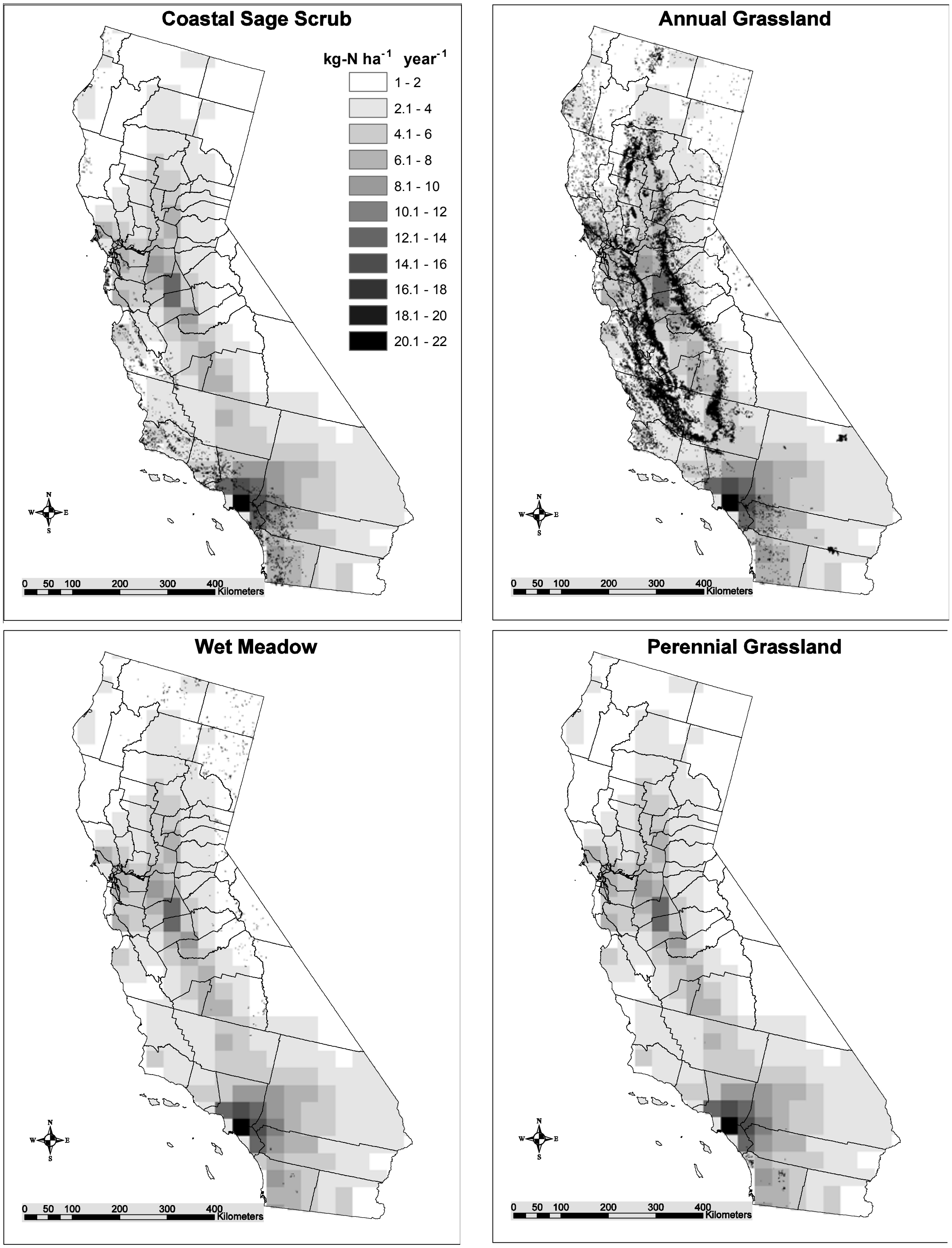
Steve Edwards, East Bay Regional Park District, April 2004
J. B. Norton, UC Cooperative Extension, December 2004
Joshua Schimel, UC Santa Barbara, May 2004
Bill Toone, San Diego Zoological Society, July 2004

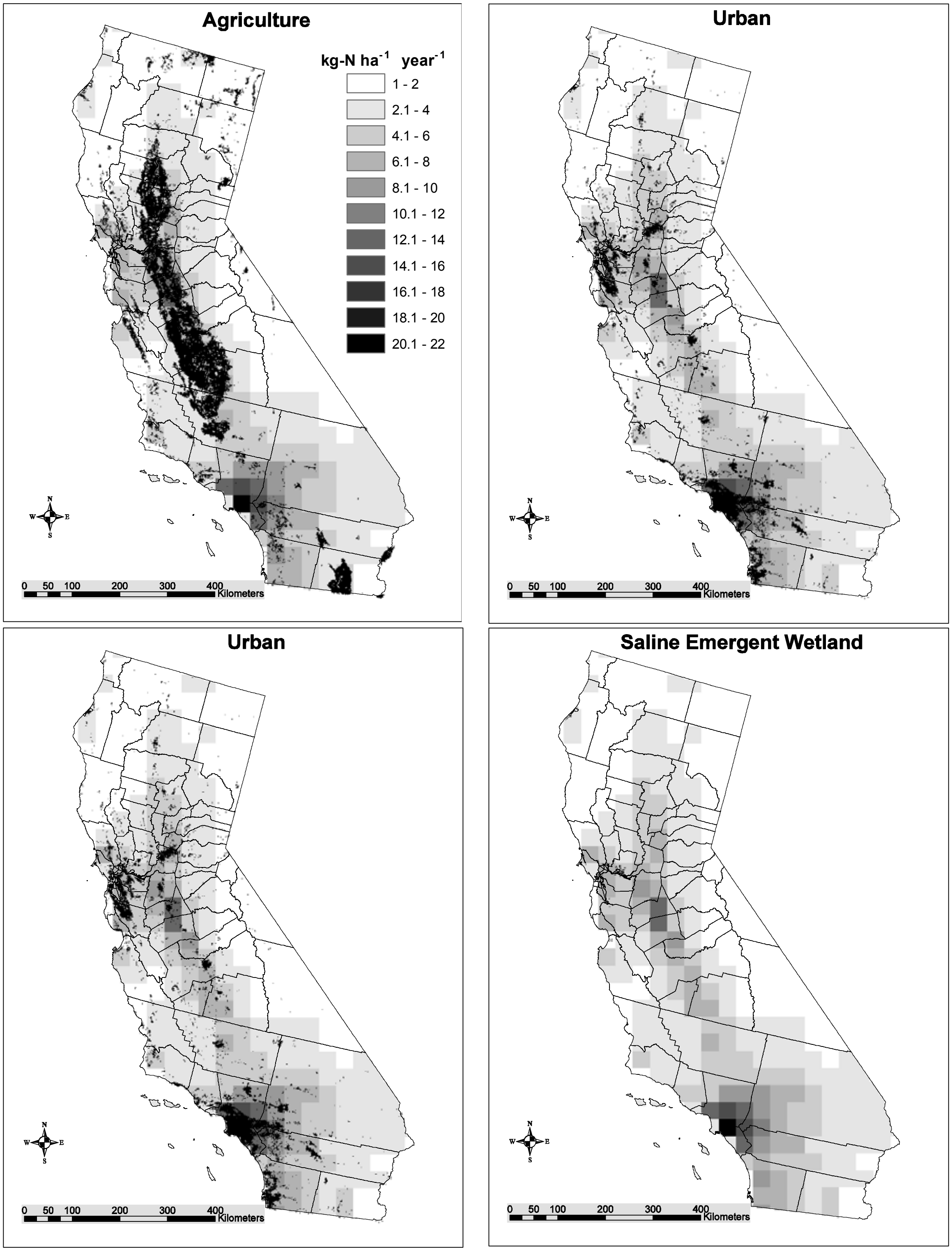
Glossary

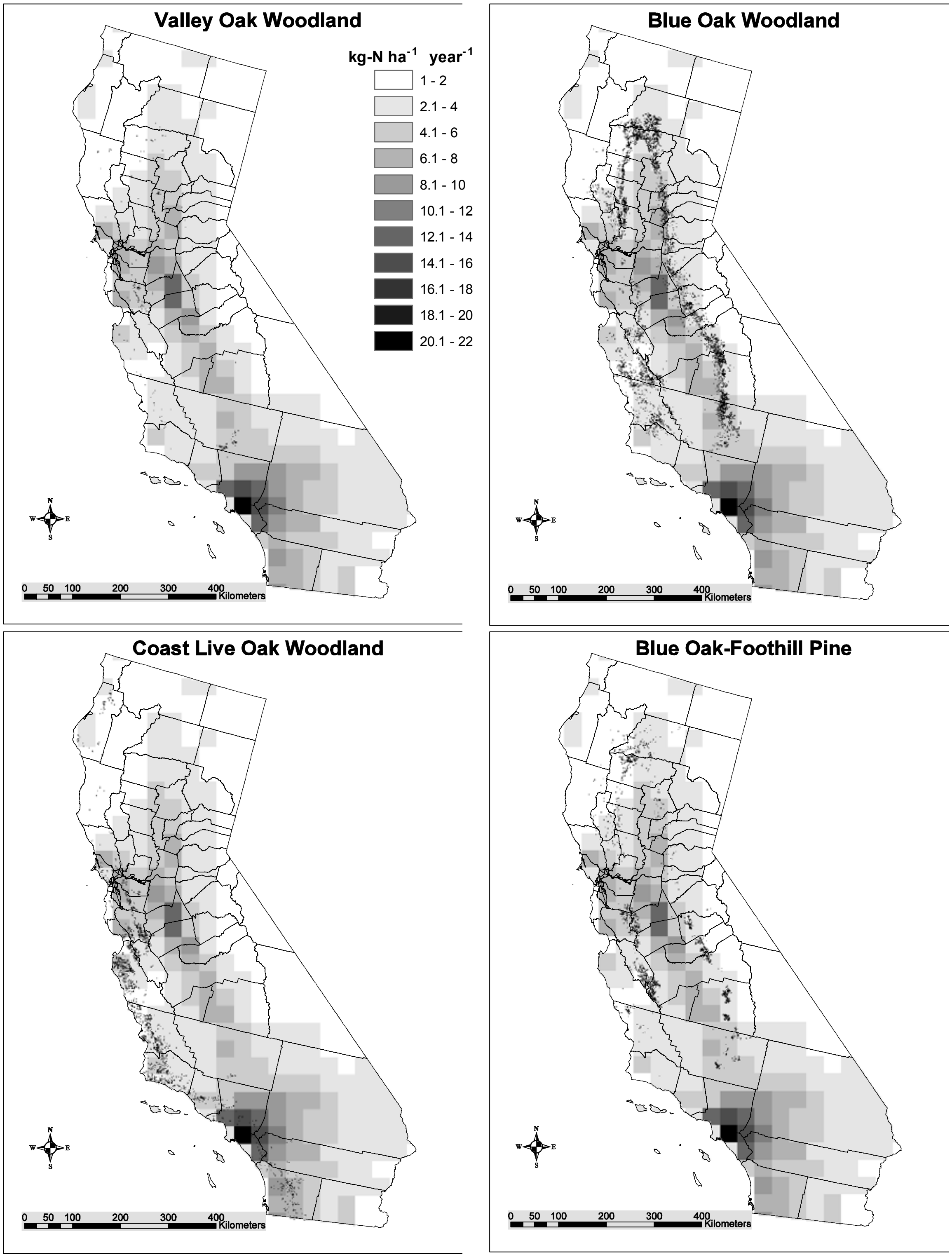
BACT	best available control technology
CDF	cumulative distribution function
cryptobiotic	soil containing microbes that hold together the soil and reduce erosion
depolymerization	the breakdown of proteins into amino acids
edaphic	affected by the soil
eutrophic	nutrient-rich water bodies
forb	a non-woody, broadleaved wild plant, such as many wildflowers
gabbro	coarse-grained igneous rock
halophytes	plants that can live in a saline environment
HCP	Habitat Conservation Plan
herbivory	the process of animals eating plants
HNO_3	nitric acid
hypoxia	a low oxygen supply
lateritic	leached, clay rich soils
mycorrhizal fungi	symbiotic fungi attached to plant roots
N_2	Nitrogen
NCCP	Natural Communities Conservation Plan
net mineralization	the amount of NH_4^+ released from breakdown of organic matter
NH_3	ammonia
NH_4^+	ammonium
nitrophilous	rich in nitrogen
nitrogen-fixing	the ability of a plant to fix atmospheric nitrogen into itself
NO	nitrogen oxide
NO_2	nitrogen dioxide
NO_3^-	nitrate
N_2O	nitrous oxide
oligotrophic	water bodies that have low nutrient levels
PAN	peroxyacetyl nitrate
$PM_{2.5}$	particulate matter ≤ than 2.5 microns
PM_{10}	particulate matter ≤ than 10 microns
pNH_4^+	particulate ammonium
pNO_3^-	particulate nitrate
PON	particulate organic nitrogen
ppm	parts per million
reductase	an enzyme that reduces the substrate
sclerophyllous	tough evergreen leaves
SCR	selective catalytic reduction
SoCAB	South Coast Air Basin
stomata	pores on the underside of leaves
taxa	groups of organisms under comparison
T&E	threatened and endangered
xeric	characterized by a dry habitat

Appendix A

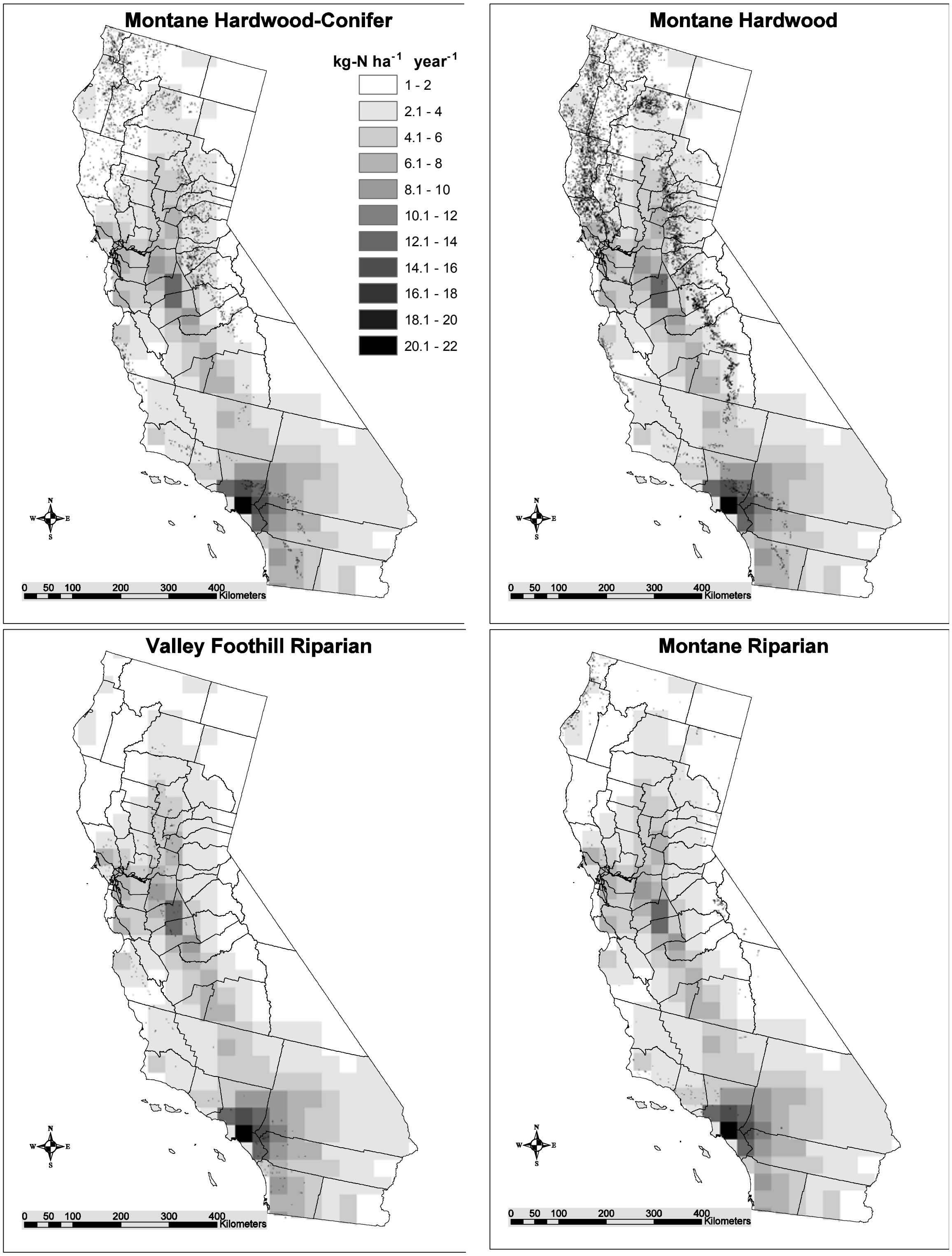
Maps of the 48 FRAP Vegetation Types Overlaid with the CMAQ 36 km Deposition Maps

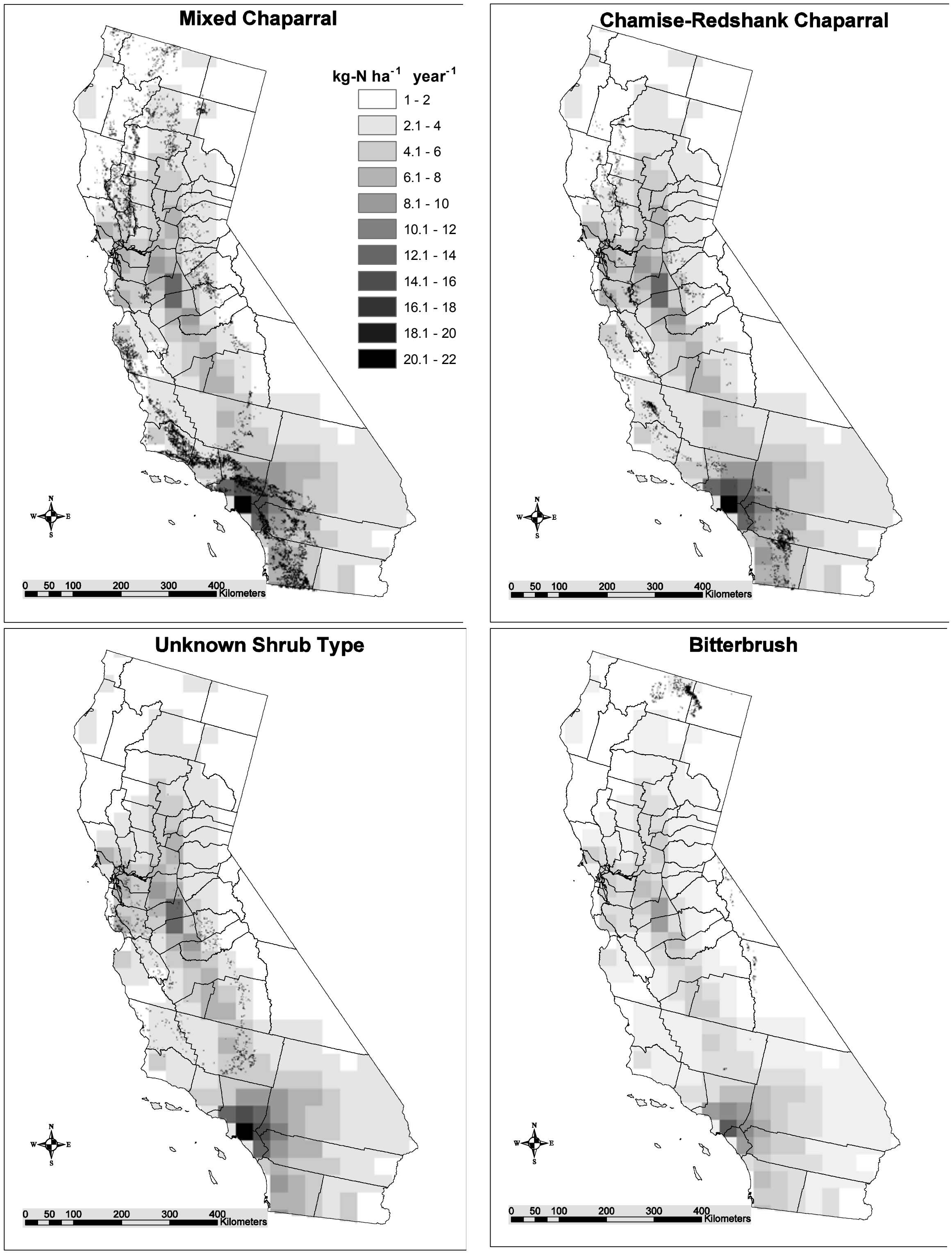

Coastal Sage Scrub
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400
Kilometers
Annual Grassland
Wet Meadow
Perennial Grassland

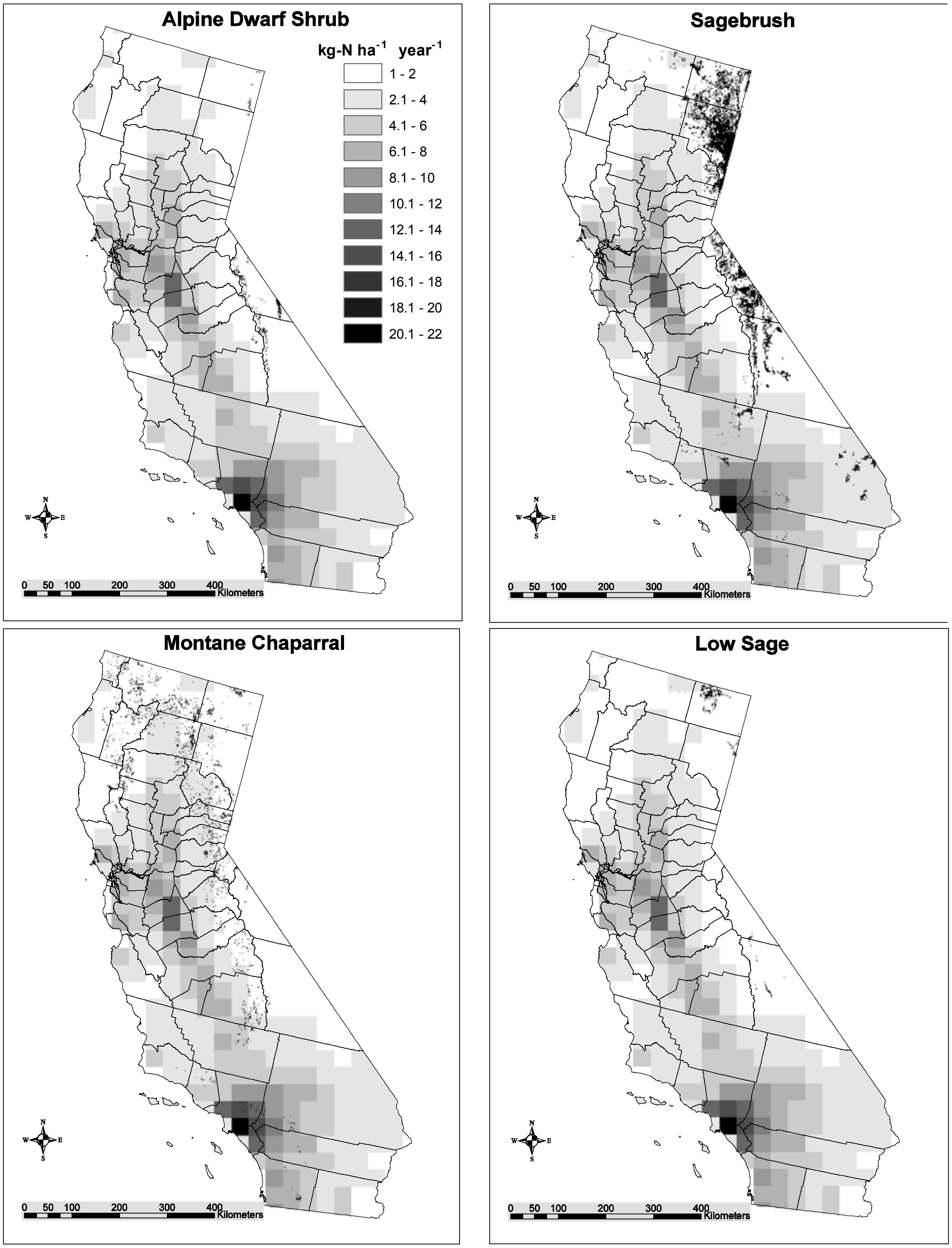

Agriculture
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
0 50 100 200 300 400 Kilometers
Urban
0 50 100 200 300 400 Kilometers
Urban
0 50 100 200 300 400 Kilometers
Saline Emergent Wetland
0 50 100 200 300 400 Kilometers

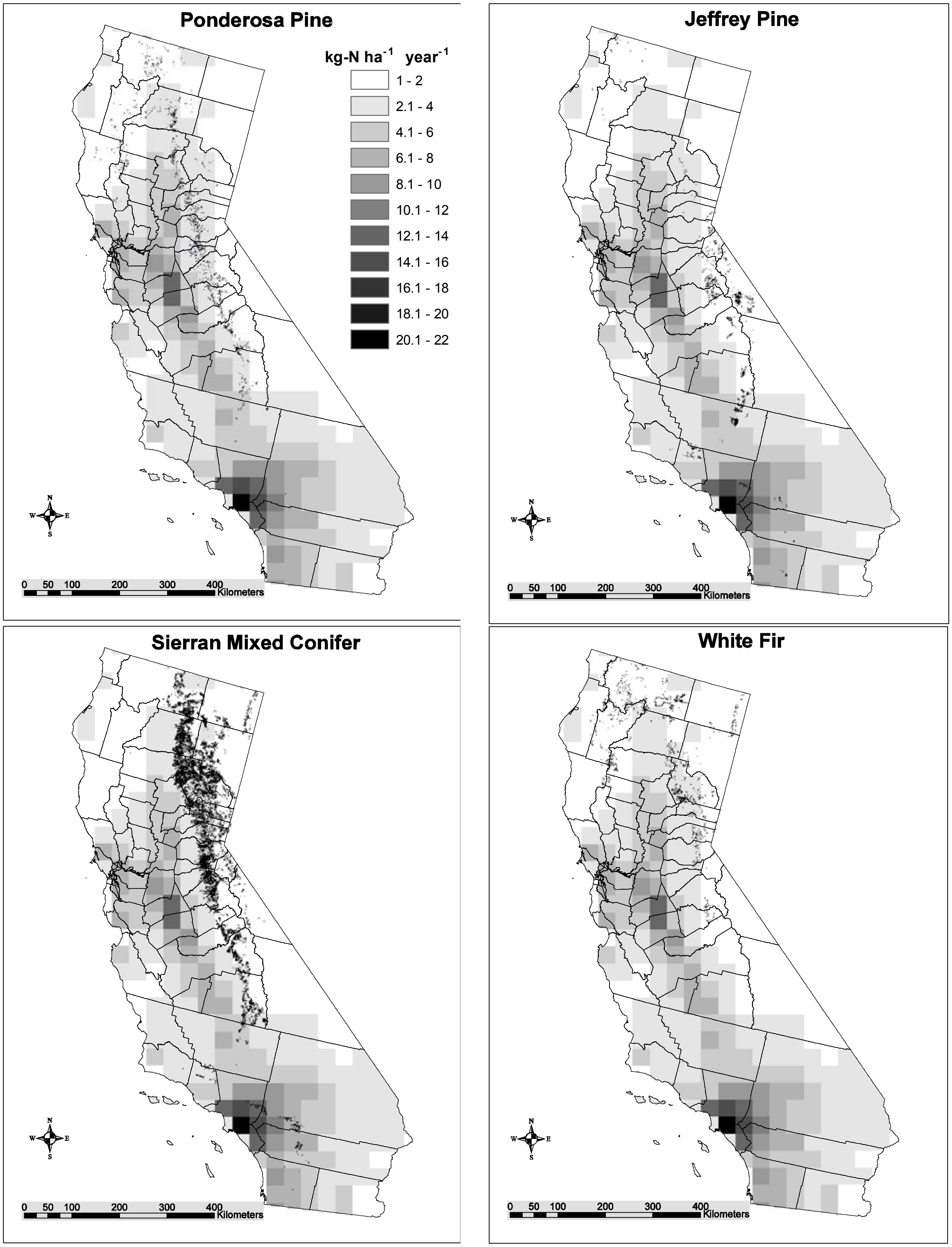


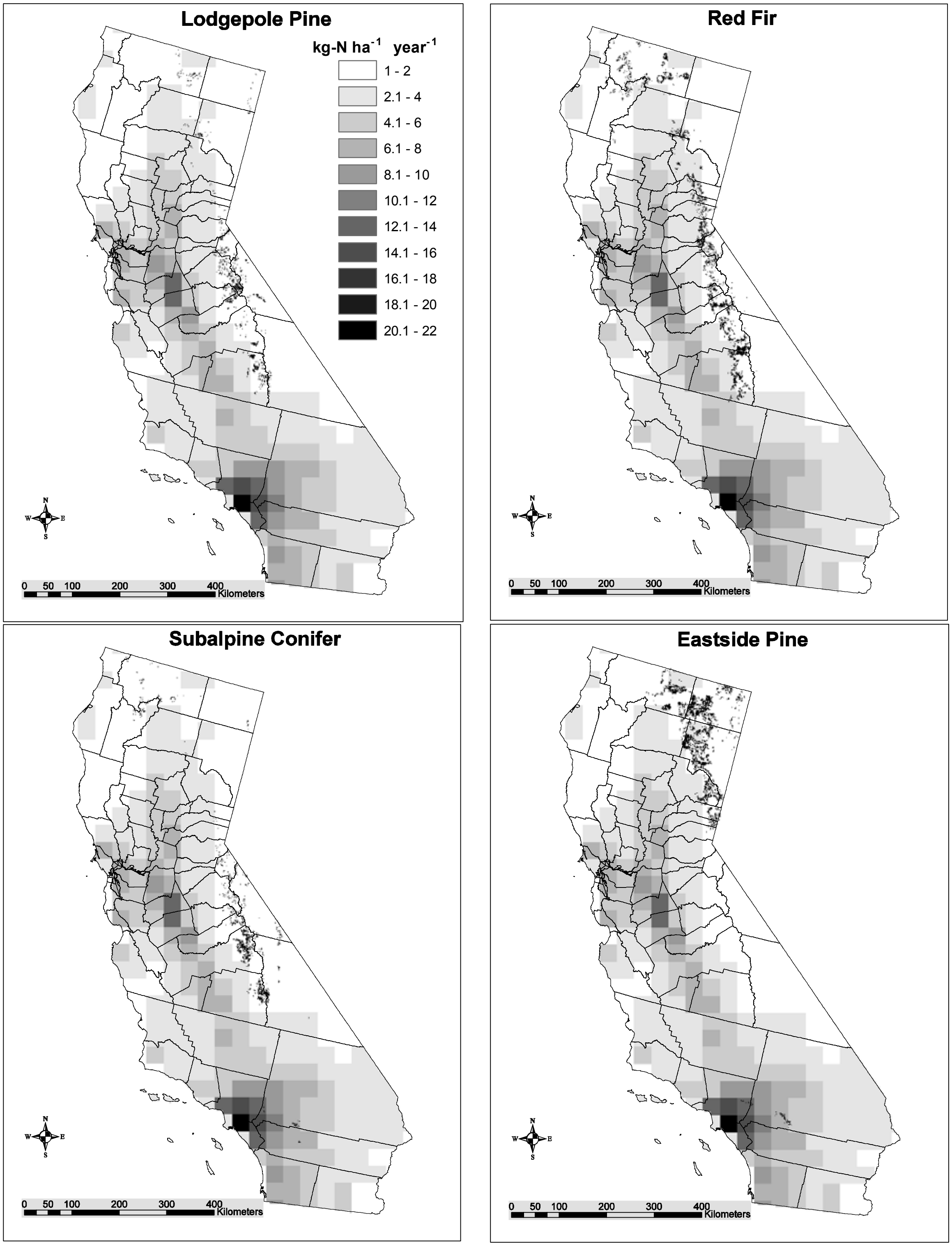
Valley Oak Woodland
Blue Oak Woodland
Coast Live Oak Woodland
Blue Oak-Foothill Pine
kg-N ha^{-1} year^{-1}
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400
Kilometers

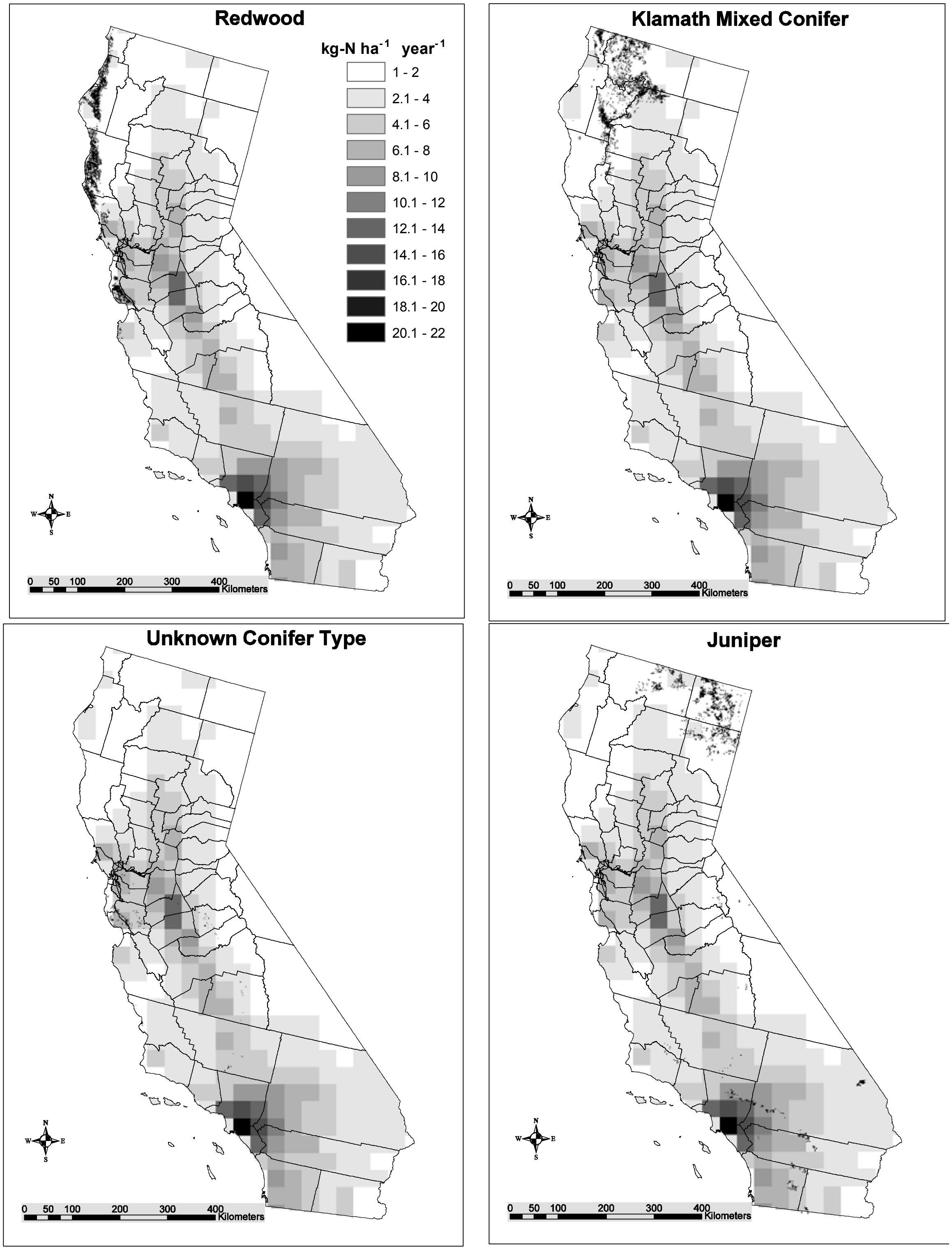

Montane Hardwood-Conifer
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
Montane Hardwood
Valley Foothill Riparian
Montane Riparian
0 50 100 200 300 400
Kilometers
N
S
E
W

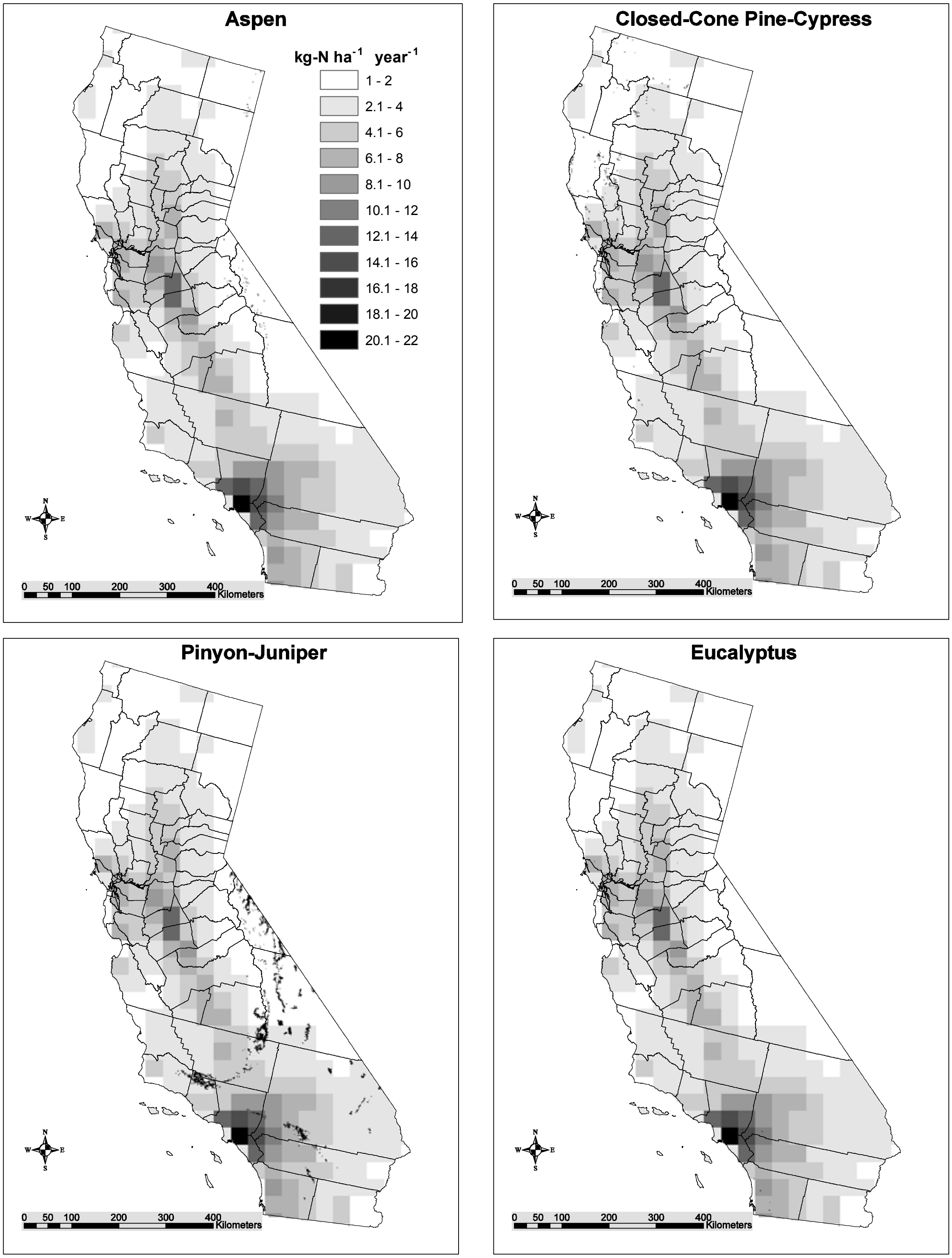

Mixed Chaparral
kg-N ha^{-1} year^{-1}
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
0 50 100 200 300 400 Kilometers
Chamise-Redshank Chaparral
0 50 100 200 300 400 Kilometers
Unknown Shrub Type
0 50 100 200 300 400 Kilometers
Bitterbrush
0 50 100 200 300 400 Kilometers


Alpine Dwarf Shrub
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
0 50 100 200 300 400 Kilometers
Sagebrush
0 50 100 200 300 400 Kilometers
Montane Chaparral
0 50 100 200 300 400 Kilometers
Low Sage
0 50 100 200 300 400 Kilometers

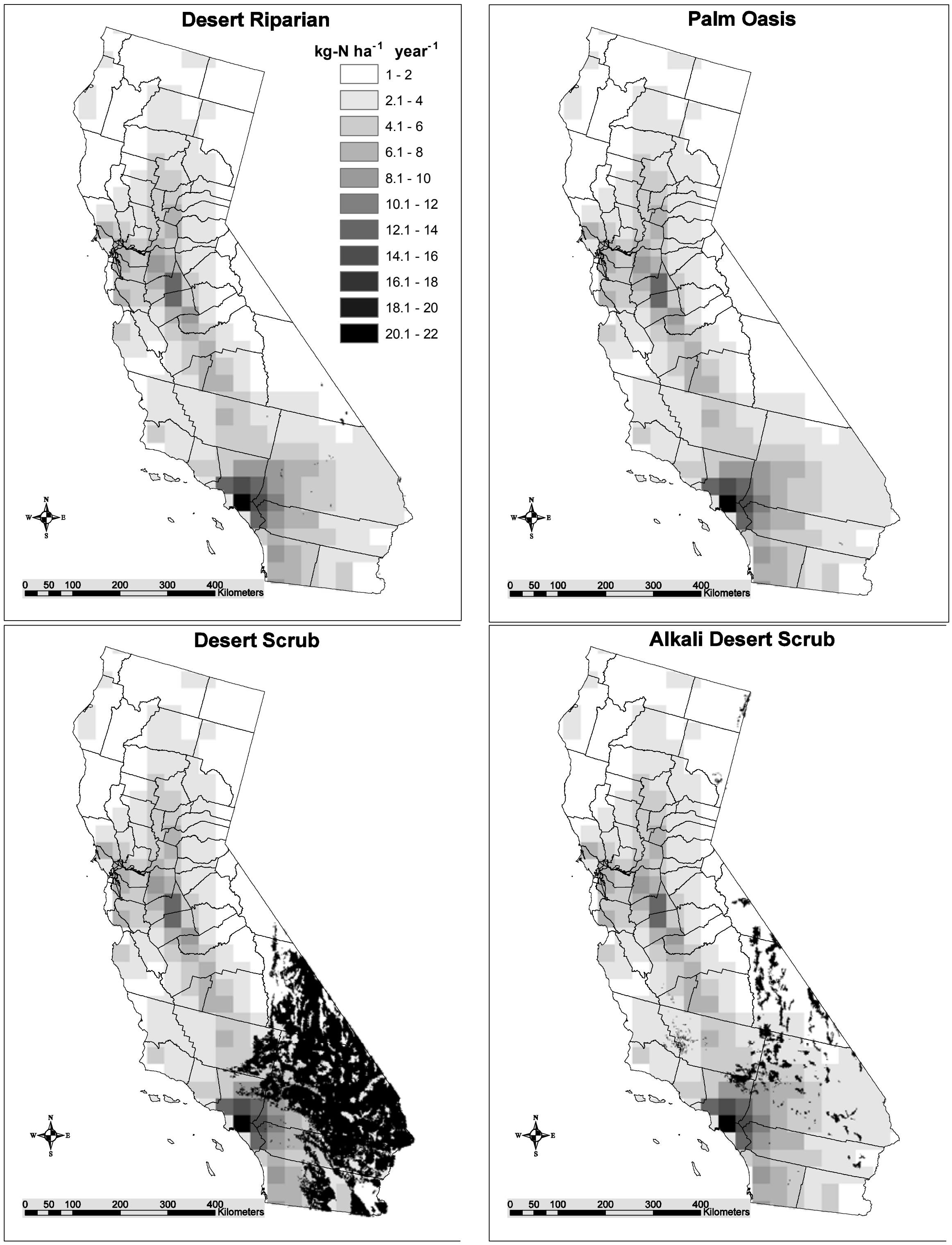

Ponderosa Pine
Jeffrey Pine
Sierran Mixed Conifer
White Fir
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400
Kilometers

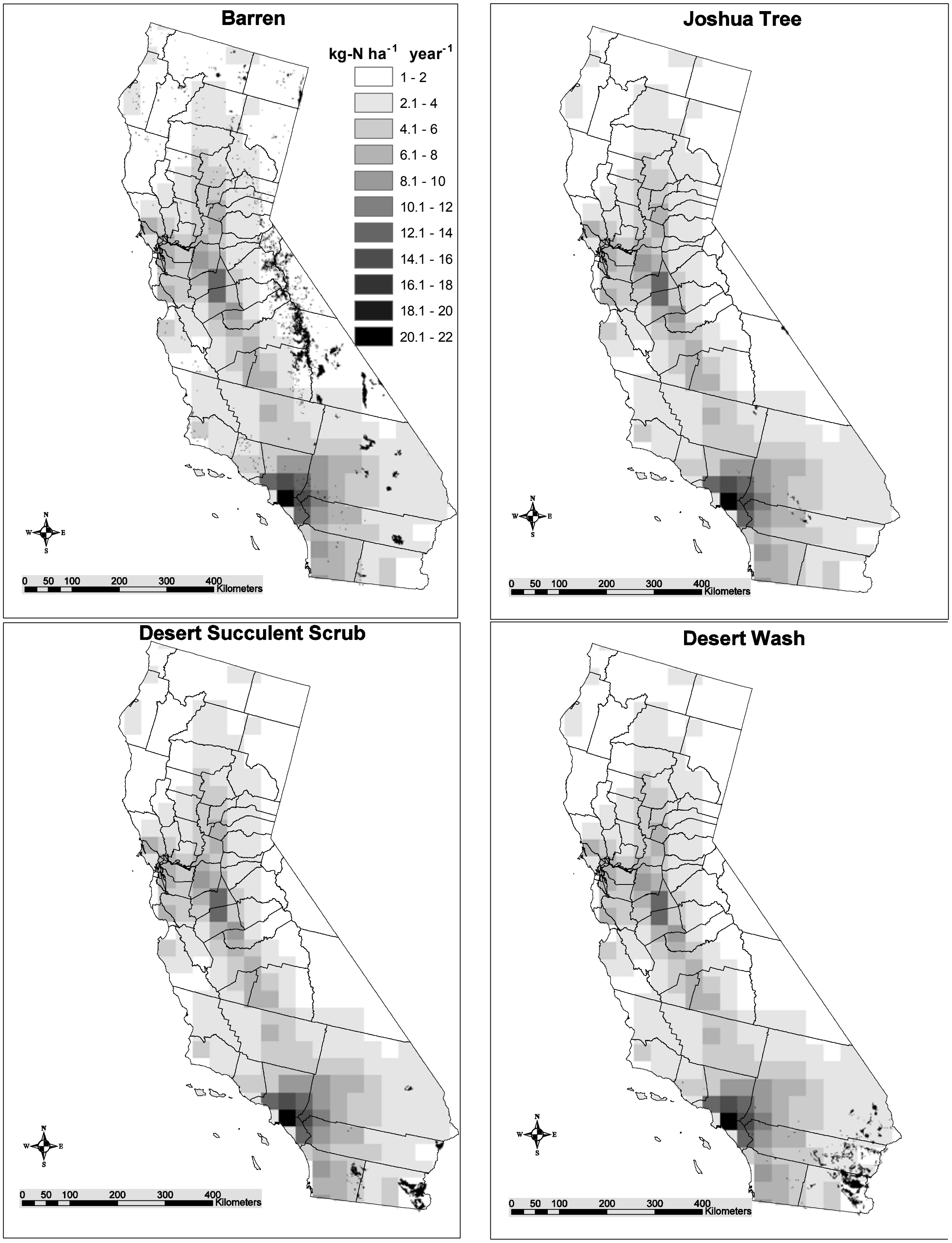

Lodgepole Pine
kg-N ha^{-1} year^{-1}
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400 Kilometers
Red Fir
0 50 100 200 300 400 Kilometers
Subalpine Conifer
0 50 100 200 300 400 Kilometers
Eastside Pine
0 50 100 200 300 400 Kilometers


Redwood
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400
Kilometers
Klamath Mixed Conifer
0 50 100 200 300 400
Kilometers
Unknown Conifer Type
0 50 100 200 300 400
Kilometers
Juniper
0 50 100 200 300 400
Kilometers



Aspen
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
N
W
E
S
0 50 100 200 300 400
Kilometers
Closed-Cone Pine-Cypress
Pinyon-Juniper
Eucalyptus


Desert Riparian
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
0 50 100 200 300 400 Kilometers
Palm Oasis
0 50 100 200 300 400 Kilometers
Desert Scrub
0 50 100 200 300 400 Kilometers
Alkali Desert Scrub
0 50 100 200 300 400 Kilometers


Barren
kg-N ha-1 year-1
1 - 2
2.1 - 4
4.1 - 6
6.1 - 8
8.1 - 10
10.1 - 12
12.1 - 14
14.1 - 16
16.1 - 18
18.1 - 20
20.1 - 22
0 50 100 200 300 400 Kilometers
Joshua Tree
0 50 100 200 300 400 Kilometers
Desert Succulent Scrub
0 50 100 200 300 400 Kilometers
Desert Wash
0 50 100 200 300 400 Kilometers

Appendix B

California Natural Diversity Data Base (CNDDB) Plant and Animal Taxa List with N-deposition exposure

This Excel spreadsheet contains information from the California Natural Diversity Data Base (CNDDB) and the 36 km CMAQ map. The codes for Fedlist and Statelist (columns G and H) are 1 = Endangered, 2 = Threatened, and 3 or more = Rare. Global and State rankings (columns N and O) are The Nature Conservancy classifications of status, and definitions can be found at the CNDDB site. Nitrogen deposition exposure is in kg-N ha^{-1} yr^{-1} (columns I [Mean], J [Max], and K [Min]). Threatened and Endangered status (column V) is inclusive of both state and federal lists.