# NITROGEN DEPOSITION, GRASSLAND CONVERSION AND REHABILITATION

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## ABSTRACT

Nitrogen deposition from anthropogenic sources is increasing globally and is already impacting shrublands, grasslands, and pastures. Where soil nitrogen has become very high, dominant shrubs and forbs are being lost and replaced by a few species of grasses and weeds. To date N deposition is high enough in Europe and southern California (greater than 30 kg/ha yr) to have negative effects on pastures and rangeland. In addition, many field fertilization studies corroborate the negative effects on species diversity, and serve as a model for expected changes in vegetation as global N deposition increases. N deposition is not only a conservation issue, but also entails loss of nutritious forbs, sometimes leaving behind only coarse grasses. Rehabilitation to increase species diversity will entail legislative efforts to decrease N pollution, followed by measures to control excess available N and increase plant diversity. Inorganic N may be lost from these systems after inputs stop because organic matter accumulation does not seem to occur, a corollary of increased mineralization rates. Thus there is reason to hope that rehabilitation can occur after N deposition rates decrease.

## KEYWORDS

Nitrogen deposition, rangelands, rehabilitation, restoration, conservation, biodiversity

## INTRODUCTION

Nitrogen deposition from anthropogenic sources is becoming a problem of global proportions that is affecting vegetation over large areas (Jefferies and Maron, 1997). While we seldom think of the world's rangelands as being affected by air pollution because of their great distance from urban and cropland N emission sources, there is evidence that N deposition will move great distances at high enough concentrations to cause eutrophication, even over the oceans (Owens et al., 1992). Deposition rates of N in southern California are up to 45 kg/ha/yr (Bytnerowicz and Fenn, 1996), the highest in the United States. A high value in the eastern U.S. is 18 kg/ha on mountain tops in New York State (Ollinger et al., 1993). Europe has the highest rates of N deposition in the world, averaging 20-50 kg/ha (Draaijers et al., 1989).

Plant available N is produced in the form of nitrates from internal combustion engines and ammonium from agricultural fertilizers and manures. The relative amounts of these depends on the source, with urban areas experiencing more nitrogen oxides (up to 90% in southern California, Bytnerowicz and Fenn, 1996) and rural areas receiving reduced nitrogen (up to 80% in the Netherlands, Bobbink and Willems, 1987). Human fixation of N now exceeds natural rates of fixation, so we are experiencing a doubling of the natural inputs of N (Galloway et al., 1995). The parts of the world most likely to experience accelerated N inputs are agricultural areas of southeast Asia. Pastures and rangelands in these areas will also experience N eutrophication.

While many range managers may welcome the unexpected N additions as an increased source of productivity, they will also cause new problems and challenges in management. Anthropogenic N deposition cannot be compared to the short term, infrequent fertilization schemes used to increase rangeland or pasture production (e.g., Rogler and Lorenz, 1957; Jacobsen et al., 1996). This is a chronic problem that adds additional nitrogen each year, increases weediness of the grassland, decreases biodiversity, causes a loss of palatable forbs and an increase in grass dominance, changes nutrient cycling patterns, and increases soil acidity (Jefferies and Maron, 1997). Our own studies have shown an increase in extractable soil N to unprecedented levels in southern California, up to 87 mg/kg soil, that we believe are causing loss of the native vegetation (Allen et al., 1996). Here we summarize literature on N deposition and N fertilization that give evidence of how global N deposition may affect rangelands and pastures, including evidence from our research in southern California rangelands.

## RESPONSES OF VEGETATION TO N DEPOSITION

The most extreme N deposition values in the world have been measured in the Netherlands, up to 115 kg/ha/yr (Jefferies and Maron, 1997), and the greatest changes in pasture species composition has been measured here. In the more polluted parts of the country one grass species, *Brachypodium pinnatum*, dominates pastures by 80% of the biomass, with a concurrent decline of diversity of forbs (Bobbink and Willems, 1987; Bobbink, 1991; Heil and Bobbink, 1993). N fertilization studies have confirmed that N deposition is the sole cause of this increased dominance and decline in diversity (Bobbink, 1991). Pastures are no longer of great economic value for their forage in the Netherlands, but rather are valued for their herbaceous biodiversity in a region that is otherwise forested. Conservation pastures are maintained by domestic grazing and mowing to preserve the many species of forbs that are part of the pasture community. However, N deposition is making this impossible, and without decreased N deposition or management to remove the dominant *Brachypodium*, some of the minor species may become extinct.

Most N deposition research has been done in forests, and has focused on whether the vegetation has reached N saturation (Aber et al., 1989; Bytnerowicz and Fenn, 1996). Natural forests are typically limited by N, which is experimentally tested by fertilizing with N and observing N uptake and retention. However, N-eutrophied forests have sufficient N for export out of the system, as measured by leaching and elevated stream nitrate levels. They also have increased tissue N, increased soil N, and increased mineralization rates. As the level of N deposition increases, species replacement and forest decline is seen. Because trees are long lived and the forest has a large capacity for N uptake, relatively few forests have shown species replacement. The exceptions are where forest dieback has occurred, although sulfur deposition and acidification are the main causes, and dieback cannot be attributed solely to N (Draaijers et al., 1989). In the severest areas, such as the Czech Republic, the forest has died back and been replaced by grasses that seem to be able to withstand the acidified soil and unbalanced nutrient conditions (Vosatka et al., 1991).

Species replacement caused by N deposition may be more rapid in vegetation with lower productivity than forests. In contrast to forests, rangelands have lower productivity and would become N saturated with lower levels of deposition. This would make them susceptible to ecosystem decline at lower levels of N deposition (Aber et al., 1989). The grasses and shrubs that dominate rangelands are shorter lived and subject to more rapid replacement by nitrophilous invaders. Thus the grasslands, shrublands and pastures of the world can be considered experimental sites to detect the effects of lower levels of N deposition, than the extremes that occur in some parts of Europe and the United States .

Because nitrogen is deposited from the atmosphere at a distance from the source where it is produced, gradients of N deposition exist around the globe. One such gradient occurs over southern California coastal sage scrub (CSS) vegetation (Allen et al., 1996). CSS shrublands have long been the most productive grazing lands in southern California. Since the rapid urbanization of southern California they have been converted either to housing tracts or to conservation reserves, with less and less land area grazed. Because CSS is a unique vegetation type endemic to California and Baja California, it supports many sensitive plant and animal species. Thus in recent years the highest societal value of the remaining CSS is for conservation and for recreation by the burgeoning human population. Conservation of CSS is becoming an increasing problem in some areas because of weed invasion, mainly annual grasses (species of *Avena, Bromus, Hordeum*) and forbs (*Brassica, Erodium*) from the Mediterranean. Minnich and Dezzani (in press) recorded a loss of up to 90% native shrub cover between 1930 and 1990 in the polluted urban areas of the northern Perris Plain near Riverside, California. However, shrub loss was small or did not occur in the rural areas that experience little or no air pollution. The causes of shrub loss and replacement by annual grasses are multiple, including fragmentation and frequent fire in some areas. We hypothesize that nitrogen deposition is one of those causes and present various lines of evidence to demonstrate this.

We measured gradients of N air concentrations and soil N values in 1994 in the Perris Plain, which extends from the urbanized areas around Riverside to rural areas to the south a distance of 50 km. Nitrogen deposition ranges from about 30 kg/ha/yr in the north (Bytnerowicz et al., 1987) to an estimated 5 kg/ha in the south (Allen et al., 1996). The high values of extractable soil N during the dry season were remarkable, up to 87 mg/kg N as nitrate plus ammonium, with low values of 5-10 mg/kg N in the south. Growing season values were not very different along the gradient, a high of 2-10 mg/kg. During the dry season soil N accumulates as plant growth decreases and as organic matter is mineralized. During the growing season, soil N is decreased by plant N uptake and some is possibly lost by leaching. Such high values of soil N are unprecedented for natural soils, and are even high for fertilized agricultural soils. We were concerned that these high values could affect shrub growth adversely, and began a series of greenhouse experiments.

Our growth studies of the dominant CSS shrubs, *Artemisia californica*, *Eriogonum fasciculatum*, and *Encelia farinosa*, show that all three have high rates of growth with increased N fertilizer in the greenhouse (Allen et al., 1996). However, after 6-9 months of maintenance at 50 ppm inorganic soil N, *Artemisia californica* began to die. Our greenhouse studies are corroborated by anecdotal field observations of shrub mortality. We see many dead and dying shrubs of all three dominant species in the northern, polluted Perris Plain, but fewer in the south where the air is cleaner. We are beginning more detailed measurements of shrub mortality. Berdowski (1993) also found high levels of insect herbivore-induced mortality on *Calluna vulgaris* subject to N deposition in Great Britain. The *Calluna* heathland is being replaced by the native grass *Deschampsia flexuosa*, while in California the CSS shrubs are replaced by Mediterranean annual grasses. We have made similar anecdotal observations that insect herbivory is high, in fact extreme, on shrubs in N polluted areas, although we do not know whether this contributes to shrub mortality. The high concentrations of tissue N may attract insects to these shrubs. The combination of high soil N and high levels of herbivory are causing mortality, and leading to shrub decline. This series of data and observations suggests that N deposition over some parts of the CSS is so high that Aber et al.'s (1989) N saturation phase has been surpassed, and we have proceeded to the vegetation replacement and ecosystem decline stage. In fact, field N fertilization studies have shown a N response in sites of low deposition, but there is no longer a growth response to N in sites of high deposition (Allen et al., 1996).

**LONGTERM N FERTILIZATION STUDIES AND GRASS DOMINANCE**

There are still relatively few examples of nitrogen deposition causing vegetation change, but many examples of range and pasture fertilization trials that can give us a lead on how vegetation will look in the future as N deposition expands. The most famous of these is the Rothamstead fertilization experiment, where the same plots were fertilized continuously for 120 years (Marrs and Gough, 1989). Ammonium fertilizer at 144 kg/ha/yr was especially effective in reducing species diversity from 17 to 2 remaining grass species, *Anthoxanthum odoratum* and *Alopecurus pratensis*. The other famous example is Tilman's series of fertilizer trials in species-rich Minnesota prairie, where high annual levels of N (10 to 272 kg/ha) resulted in dominance by the exotic *Agropyron repens* and loss of over 60% of the plant species in only four years of fertilization (Tilman, 1987). These two studies show that high soil N enables a strong dominant to persist at the expense of other species, and that this dominant is more likely to be a grass than a forb or shrub. Grasses are also among the species that disappear with N fertilizer, as *Agrostis capillaris* and *Festuca ovina* in the Rothamstead control plots (Marrs and Gough, 1989). Even a robust species such as *Schizachyrium scoparius* succumbed to competition from *Agropyron repens* in Tilman's experiments. However, these and the other studies cited above, such as Vosatka et al.'s (1991) observation that grasses replaced trees after forest die-off, suggest that species -poor grasslands will dominate in areas subject to high N deposition.

Other longterm grassland fertilizer experiments had variable results, that might be related to the time of fertilization and the initial fertility of the soil. In Colorado shortgrass prairie Milchunas and Lauenroth (1995) showed that after 5 years of 100 kg/ha annual N additions there were few species changes, until seven years after N additions were terminated. The species that increased were exotic weedy forbs, and they were especially abundant in the treatments that had received irrigation plus nitrogen. Native forbs increased in productivity relative to grasses in Kansas tallgrass prairie after four years of 100 kg/ha N (Seastedt et al., 1991). These Kansas prairie results are unusual in that most fertilizer studies showed loss of diversity or weed invasion after such high levels of N input. However, these are extremely clayey and calcareous soils that have a high capacity for absorption of N, and levels of N availability may not be as high in these soils even after such high fertilizer concentrations.

Another site that had increased forb growth with N additions up to 80kg/ha was a chalk grassland in Britain, but N additions were only carried out for one year so this does not simulate chronic additions (Wilson et al., 1995). The dominant grass *Brachypodium pinnatum* did not respond to N in one year, as was seen in Dutch grasslands (Bobbink, 1991). In a three year fertilization study with up to 140 kg/ha N, the only species that declined was a moss in a British *Festuca* grassland (Morecroft et al., 1994). Whether there are any longterm effects, as noted by Milchunas and Lauenroth (1995) seven years after the cessation of fertilization, is still to be seen in this study. The importance of longterm studies cannot be overemphasized. The longterm studies and those on coarse, N poor soils (especially Tilman's) indicate large shifts in vegetation toward a few dominant grass species, or in the case of Milchunas and Lauenroth (1995), weeds. Results of other studies are mixed, but a lack of response may be an indication that the study still needs to run longer, rather than a lack of response of the vegetation to nitrogen.

In our own field fertilization studies of coastal sage scrub vegetation, we have not yet seen a change from perennial shrubland to annual grassland after four years of fertilization with 60 kg/ha, which is twice the annual N deposition rate (Allen et al., unpubl. data). However, the levels of soil N at this experimental site do not simulate the high N that is found in polluted sites that have been receiving N deposition for decades. As reported above, we have been able to cause shrub mortality by fertilizing at these levels in the greenhouse, and our field fertilization trials do not nearly come up to such high levels. We anticipate we may wait some years before we observe vegetation changes caused by the field fertilization trials.

## REHABILITATION OF NITROGEN-EUTROPHIED RANGELANDS

The objective for rehabilitation of N eutrophied rangelands is to return the diversity of forbs, shrubs, and less dominant grasses, as appropriate. This objective increases the conservation value and also improves the grazing quality. The first requisite for this is to reduce the nitrogen inputs, and then reduce N in the soil if it has accumulated. Rehabilitation efforts will, after all, be negated if high N deposition was the cause of the species loss, and high N deposition continues. Reduction in air pollution is a political decision that can be accomplished to a large extent with legislative will. N deposition has probably already peaked in Europe and is projected to decline (Draaijers et al., 1989).

Various methods are in use to improve diversity that can be used under present day N loads, and will be even more effective as N deposition is reduced. In the *Brachypodium* dominated pastures of the Netherlands, haying in the early summer has been a proven method of increasing pasture diversity (Bobbink and Willems, 1991). However, cutting later in the growing season was not as effective. In fact, there has been some thought given to the idea that lack of response of vegetation to N in some parts of Europe may be related to continued heavy grazing, which removes dominant vegetation and maintains diversity (Jefferies and Maron, 1997).

In southern California where shrubs have been declining for a variety of reasons, shrub restoration efforts are underway. In many areas the soils are high in available nitrogen, creating conditions where the annual grasses are highly competitive. The first restoration goal is immobilization of inorganic N. This was accomplished by using mulch in a restoration plot on a pipeline disturbance. Bark mulch was more effective than straw mulch in promoting the growth of *Artemisia californica* (Zink, 1994). Bark also promoted growth of soil saprophytic fungi more than straw or unmulched soil, which is responsible for immobilization of nitrogen.

Obviously it will not be possible to apply bark mulch to vast extenses of rangelands that have excesses of nitrogen and undesirable species. It is not clear from the various N deposition studies whether there has been a permanent increase in soil N, that would create soil conditions uninhabitable to the former species. For soil N to increase permanently, there must be an increase in organic matter plus soil microorganisms to bind the nitrogen. Our measurements of southern California soils do not show significant increases in soil organic matter in high N sites (Allen et al., unpubl.). Wedin and Tilman (1996) have shown no accumulation of organic matter even after years of fertilization. The most likely reason for this is increased rates of mineralization in high N soils, which keeps carbon from building up in the soil. Thus many soils that have been impacted by N deposition will have higher levels of inorganic N, but organic N buildup will be slow and small. An exception to this will be sites that are invaded by nitrogen fixing plants, such as *Myrica faya* in the poor volcanic soils of Hawaii (Vitousek and Walker, 1989). These soils have permanently increased organic N, but in this case the source of N input is organic litter rather than inorganic N deposition.

There is a great deal of reason to hope that vegetation that has been altered by N deposition can be rehabilitated, because inorganic N may be lost from the system after input declines. Many grasslands worldwide have been converted to exotic, species-poor grasslands where there is no evidence for high N inputs (D'Antonio and Vitousek, 1992). These include invasion of the Great Basin of the U.S. by *Bromus tectorum*, of the California grasslands by Mediterranean grasses, and of Hawaiian shrublands by exotic grasses. Studies on N deposition and N fertilization can give us some clues about how restoration of these may begin. Available N and N mineralization rates are typically higher under grasses than forbs, shrubs, or forests (Ingham et al., 1989). Thus it would seem that practices that slow mineralization and bind inorganic nitrogen, such as mulching, and planting with species that promote slower mineralization (Wedin and Tilman, 1993) would push the system back to a lower available N . Thus revegetation efforts need to consider more than reestablishing the vegetation, they need to manage and restore N dynamics.

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## REFERENCES

**Aber, J.D., K.J. Nadelhoffer, P. Steudler and J.M. Melillo.** 1989. Nitrogen saturation in northern forest ecosystems. BioScience **39**: 378-386.

**Allen, E. B., P.E. Padgett, A. Bytnerowicz, and R. A. Minnich.** 1996. Nitrogen deposition effects on coastal sage vegetation of southern California. *In* Bytnerowicz, A., M.J. Arbaugh and S. Schilling, tech. coords. Proceedings of the international symposium on air pollution and climate change effects on forest ecosystems, Riverside, CA. Gen. Tech Rep. PSW-GTR 164, Albany, CA, Pacific SW Res. Station, USDA For. Serv. HTTP://www.rfl.pswfs.gov/pubs/psw-gtr-164/index.html

**Berdowski, J.J.M.** 1993. The effect of external stress and disturbance factors on *Calluna*-dominated heathland vegetation. Pages 85-124 *in* R. Aerts and G.W. Heil, eds. Heathlands: patterns and processes in a changing environment. Kluwer Academic, Dordrecht, The Netherlands.

**Bobbink, R.** 1991. Effects of nutrient enrichment in Dutch chalk grassland. J. Appl. Ecol. **28**: 28-41.

**Bobbink, R. and J.H. Willems.** 1987. Increasing dominance of *Brachypodium pinnatum* (L.) Beauv. in chalk grasslands: a threat to a species-rich ecosystem. Biol. Conserv. **40**: 301-314.

**Bobbink, R. and J.H. Willems.** 1991. Impact of different cutting regimes on the performance of *Brachypodium pinnatum* in Dutch chalk grassland. Biol. Conserv. **56**: 1-21.

**Bytnerowicz, A. and M.E. Fenn**. 1996. Nitrogen deposition in California forests - a review. Environ. Pollution **92**:127-146.

**D'Antonio, C.M. and P.M. Vitousek.** 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annu. Rev.

Ecol. Syst. **23**: 63-87.

**Draaijers, G.P., W.P.M.S. Ivens, M.M. Box and W. Bleuten.** 1989. The contribution of ammonia emissions from agriculture to the deposition of acidifying and eutrophying compounds onto forests. Environ. Pollut. **60**: 55-66.

**Galloway, J.N., W.H. Schlesinger, H.L. II, A. Michaels and J.L. Schnoor.** 1995. Nitrogen fixation: anthropogenic enhancement-environmental response. Global Biogeochem. Cycles **9**: 235-252.

**Heil, G.W. and R. Bobbink.** 1993. Impact of atmospheric deposition on dry heathlands. A stochastic model simulating competition between *Calluna vulgaris* and two grass species. Pages 181-200 *in* R. Aerts and G.W. Heil, eds. Heathlands: patterns and processes in a changing environment. Kluwer Academic, Dordrecht, The Netherlands.

**Ingham, E.R., D.C. Coleman and J.C. Moore**. 1989. An analysis of food-web structure and function in a sortgrass prairie, a mountain meadow, and a lodgepole pine forest. Biology And Fertility Of Soils **8**:29-37.

**Jacobsen, J.S., S.H. Lorbeer, H.A.R. Houlton and G.R. Carlson.** 1996. Nitrogen fertilization of dryland grasses in the Northern Great Plains. J. Range Manage. **49**: 340-345.

**Jefferies, R.L. and J.L. Maron.** 1997. The embarrassment of riches: atmospheric deposition of nitrogen and community and ecosystem processes. Perspectives **12**(2): 74-78.

**Marrs, R.H. and M.W. Gough**. 1989. Soil fertility—a potential problem for habitat restoration. Pages 29-44 *in* G.P. Buckley, ed. Biological habitat reconstruction. Belhaven Press, London.

**Milchunas, D.G. and W.K. Lauenroth.** 1995. Inertia in plant community structure: state changes after cessation of nutrient-enrichment stress. Ecol. Monogr. **5**: 452-458.

**Minnich, R. A. and R.J. Dezzani**. 1998. Historical decline of coastal sage scrub in the Riverside-Perris Plain, California. Revista Chilena de Historia Natural, in press.

**Morecroft, M.D., E.K. Sellers and J.A. Lee.** 1994. An experimental investigation into the effects of atmospheric nitrogen deposition on two semi-natural grasslands. J. Ecol. **82**: 475-483.

**Ollinger, S. V., J. D. Aber, Lovett. G.M., S. E. Millham, R. G. Lathrop, and J. M. Ellis**. 1993. A spatial model of atmospheric depostion for northeastern U.S. Ecological Applications **3**:459-472.

**Owens, J.P., J.N. Galloway, and R.A. Duce**. 1992. Episodic nitrogen deposition to oligotrophic oceans. Nature **357**:397-399.

**Rogler, G.A. and R.J. Lorenz.** 1957. N fertilization of N. Great Plains rangelands. J. Range Manage. **10**: 156-160.

**Seastedt, T.R., J.M. Briggs and D.J. Gibson.** 1991. Controls of nitrogen limitation in tallgrass prairie. Oecologia **87**: 72-79.

**Tilman, D.** 1987. Secondary succession and the pattern of plant dominance along experimental nitrogen gradients. Ecol. Monogr. **57**: 189-214.

**Vitousek, P.M. and L.R. Walker**. 1989. Biological invasion by *Myrica faya* in Hawaii: Plant demography, nitrogen fixation, and ecosystem effects. Ecol. Mongr. **59**:247-265.

**Vosatka, M., P. Cudlin and V. Mejstrik**. 1991. VAM populations in relation to grass invasion associated with forest decline. Environ. Pollution **73**:263-270.

**Wedin, D.A. and D. Tilman.** 1993. Nitrogen mineralizatin dynamics in grass monocultures. Oecologia **96**:186-192.

**Wedin, D.A. and D. Tilman**. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. Science **274**:1720-1723.

**Wilson, E.J., T.C.E. Wells and T.H. Sparks.** 1995. Are calcareous grasslands in the UK under threat from nitrogen deposition? - an experimental determination of a critical load. J. Ecol. **83**: 823-832.

**Zink, T.A**. 1994. The effects of recalcitrant organic amendment on disturbed coastal sage and creosote scrub habitats of southern California. M.S. Thesis. San Diego State University, San Diego, California.