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Chapter 19

Management Options for Mitigating Nitrogen (N) Losses from N-Saturated Mixed-Conifer Forests in California

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Abstract

Mixed-conifer forests of southern California are exposed to nitrogen (N) deposition levels that impair carbon (C) and N cycling, enhance forest flammability, increase the risk of fire occurrence and air pollution emissions in fire, and increase nitrate runoff and soil N emissions both pre- and postfire. N-deposition abatement policies and prescribed fire treatments have been proposed to mitigate the interactive effects of fire suppression, N deposition, and wildfire occurrence. To test the most effective management options for N-enriched forests, a simulation study was done using a parameterization of the DAYCENT model for a mixed-conifer forest site currently experiencing 70 kg N ha^{-1} yr^{-1}. Five N deposition scenarios were defined, ranging from 5 to 70 kg N ha^{-1} yr^{-1}. Five abatement strategies ranging from 0% to 100% reductions in N deposition were considered for each N-deposition scenario. The influence of prescribed fire was tested for the selected N deposition and abatement scenarios, considering 15-, 30-, and 60-year intervals (PF15, PF30, and PF60, respectively), or no prescribed fires. When the most extreme N-deposition scenario was compared to the lowest, fuel loads were increased by 121%, resulting in 70% increases in wildfire emissions of particulate matter (PM_{10} and $PM_{2.5}$), methane (CH_4), carbon monoxide (CO), carbon dioxide (CO_2), and sulfur dioxide (SO_2). The estimated increase in wildfire nitrogen oxide (NO_x) emissions ranged from 56% to 210%. The larger values were derived when variations in fuel N content were taken into account. The combination of reduced N deposition and prescribed fire was most effective in reducing long-term N losses to the atmosphere and

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in runoff. The PF15 treatment combined with 50–75% reduced N deposition were the best options for reducing N losses before and after fire. However, even prescribed fire at longer intervals and in combination with 25–50% reduced N deposition still resulted in large reductions in ecosystem losses of N. Implementation of such treatments would be considered a major achievement towards mitigating the symptoms of N saturation, even though in sites chronically exposed to 70 kg N ha^{-1} yr^{-1} a 100% reduction in N deposition may require many years to return N losses to baseline levels.

19.1. Introduction

Mixed-conifer forests in California are a multilayered community with a variable species composition extending across a range of moisture regimes. The dominant overstory species are ponderosa pine (*Pinus ponderosa*) or Jeffrey pine (*P. jeffreyi* Grev. and Balf.), and commonly associated species include California black oak (*Quercus kelloggi* Newb.), white fir (*Abies concolor* Gord. & Glend.), and incense cedar (*Calocedrus decurrens* [Torr.] Florin). Air pollution impacts on this ecosystem type are well documented, with tropospheric ozone and nitrogen (N) deposition as the most hazardous pollutants, impairing tree performance and ecosystem nutrient cycling (Fenn et al., 2003c; Miller & McBride, 1999).

The effects of N deposition on mixed-conifer forest sites in California have been studied across air pollution gradients consisting of twenty-seven sites ranging from 1000 to 2500 m a.s.l. and located in the Sierra Nevada Mountains, San Gabriel Mountains northeast of Los Angeles, and the San Bernardino Mountains east of Los Angeles (Fenn et al., 2008). Most of the sites are on weathered or decomposed granitic rock. The soils are generally sandy loam in texture, and percent base saturation ranges from 70% to 100%. Nitrogen deposition across the different sites ranges from 1.5 kg N ha^{-1} yr^{-1} to 71.5 kg N ha^{-1} yr^{-1} (Breiner et al., 2007), with sites receiving N loads that are among the highest in the world and showing symptoms of N saturation (Fenn et al., 2003a). Specific reviews on the effects of N deposition in this type of forest ecosystem have been provided for the sites located in the San Bernardino and San Gabriel Mountains (Fenn & Poth, 1999) as well as for Sierra Nevada sites (Fenn et al., 2003c). Increasing N loads have resulted in a series of well-documented effects, such as increased foliar N concentrations, increased tree growth rates of ponderosa pine, increased tree mortality as a result of stand densification, increased depth and N content of the forest floor, decreased C:N (carbon:nitrogen) ratios in litter and mineral soil, increased nitrification rates, increased emissions of soil gaseous compounds (nitric oxide [NO] and nitrous oxide [N_2O]), depletion of soil cation pools, increased soil acidification, and increased leaching of nitrate (NO_3^-) in soil solution and NO_3^- export in stream water. Nitrification is the key process driving these N-saturation responses. Soil conditions of high base cation saturation, moderate pH values, and the aerobic condition of these coarse-textured soils appear to be among the factors favoring active nitrification in the soils of California mixed-conifer forests. As a result, the N cycle is strongly nitrate dominated and higher in sites with elevated N deposition. Nitrogen cycling dominated by nitrate is typical of N-saturated sites (Fenn et al., 1998). Based on these findings Breiner et al. (2007) and Fenn et al. (2008) have recently established empirical dose-response relationships for N deposition inputs and N cycling processes with a high predictive power.

Recurrent fire is an integral component of mixed-conifer forests because the Mediterranean climate of winter rain and dry summers results in inefficient decomposition, rapid fuel buildup, and high fire hazard. Great changes in fire frequency have been documented when comparing presettlement conditions with current fire occurrence. Fire exclusion began in the late 1800s and early to mid-1900s in most parts of the Pacific regions (Houghton et al., 2000). Active fire suppression starting in the first quarter of the last century has altered fire regimes as well as ecosystem structure, resulting in increased fire intervals and fuel buildup and changes in species composition. For instance, Minnich et al. (1995) estimated that current conditions would lead to an estimated fire rotation period of 360 years, and reported that 60 years of suppression caused stem density increases of 100–200 stems/ha (dbh > 10 cm), with increases proportional to mean annual precipitation. Stand-thickening has been accompanied by an increasing density of standing dead trees. High mortality was due to an overabundance of trees competing for moisture and nutrients, with the shift in species dominance further heightened by differential responses to drought, bark beetle infestation, and air pollution. As a result, more intense and more extensive fires may occur than under a normal fire regime, converting the fire regime from a patchy mosaic continuum to a sustained cyclic stand-replacement fire regime (Minnich, 1999).

These wildfire events have large influences on the release of N. As Johnson et al. (2004) have pointed out, the potential loss of N with fire is far greater than due to leaching in most ecosystems, except for catchments experiencing high N-fixation rates or in N-saturated ecosystems. However, considering the whole C and N budget in the ecosystem, the effects of fire on soil C and N are very dependent on fire intensity and time since fire (Johnson & Curtis, 2001; Wan et al., 2001). Fire also influences water quality as postfire increases in mineral leaching rates have been observed that show ammonium (NH_4^+) loss predominant over NO_3^- export beginning immediately after fire and up to one year postfire, while NO_3^- export prevailed afterwards. Three years after fire, mineral N leaching was greatly reduced (Johnson et al., 2007). However, in N-saturated chaparral catchments near Los Angeles, California, NO_3^- dominated runoff in unburned catchments and postfire runoff NO_3^- was far greater than NH_4^+ in burned catchments (Riggan et al., 1994).

In addition to increased mineral leaching rates and subsequent impacts on stream water quality, increased C and N losses may also occur as a result of precipitation events occurring soon after the fire event by inducing runoff and erosion that would contribute to nutrient export from the forest floor and possibly also from the surface soil horizon (Johnson et al., 2008). Finally, fire-induced emissions of air pollutants may have a large impact on regional air quality (Cheng et al., 1998; Liu, 2004), enhancing regional haze (McMeeking et al., 2006), and adversely impacting human health, especially near densely populated areas (Clinton et al., 2006; Massie et al., 2006; Wu et al., 2006).

Atmospheric N deposition may enhance the direct and indirect impacts of fire (Fenn et al., 2003c). At the ecosystem level, N deposition has been found to change the chemical composition of different forest components, such as foliage, forest floor, and upper soil layers, resulting in an alteration of C and N cycles. N deposition also increases stand density and tree volume growth and increases foliar N content. These processes affect the chemical properties of fuel and increase fuel loading in the ecosystem, thus enhancing forest flammability and increasing the risk of fire occurrence with associated impacts on fire severity and intensity. Moreover, N deposition may also worsen the adverse consequences of fire, such as air pollution emissions (Yokelson et al., 2007) and nitrate runoff to stream water (Johnson et al., 2008; Riggan et al., 1994).

Prescribed fire is a forest management technique that lessens the damage from wildfire by removing a portion of the accumulating dead fuels and reducing the stature of the developing understory when burning conditions are not severe (Liu, 2004). Current U.S. policies for federal lands emphasize the use of prescribed fire, either alone or in combination with other techniques, to meet fuel reduction objectives (HFRA, 2003). Prescribed fire has been considered as a tool for alleviating the condition of N enrichment in forest and chaparral ecosystems (Fenn et al., 1998; Meixner et al., 2006).

Air pollution control would be the ultimate solution for alleviating the symptoms of N excess that occur under chronic N-deposition exposures.

The effectiveness of these measures would depend on the magnitude of N deposition and the number of years the forests have been exposed to elevated N-deposition rates, and thus the cumulative N-deposition loading. For instance, Fenn et al. (2008) indicated that a fivefold N-deposition reduction might be needed to prevent N losses at the San Bernardino Mountain site experiencing the highest deposition rate. It would be difficult to achieve this goal, at least in the short term, and therefore it is very likely that this site will continue to experience elevated N losses for the foreseeable future. Neither is prescribed fire alone expected to reverse N saturation symptoms (Johnson et al., 2008; Meixner et al., 2006). We propose that an appropriate combination of reduced N-deposition inputs, removal of a significant fraction of the accumulated N stores and reduced stand densification by prescribed fire, and stimulated postfire regrowth could be effective in mitigating the symptoms of N saturation.

19.2. Objectives

This chapter discusses different forest management and N-deposition scenarios that would result in long-term and short-term effects in ecosystem N cycling. A simulation study was carried out to test the following hypotheses:

- In the long-term, elevated N deposition leads to greater fuel loads and affects the N content of fuels. As a result, wildfire air pollution emissions will increase in forests experiencing high N-deposition rates.
- In the long-term, elevated N deposition enhances N losses from the ecosystem through increased gaseous emissions from soil and increased N runoff.
- In the short term, elevated N deposition enhances N losses to the atmosphere and stream water after prescribed fire events.
- These adverse effects could be mitigated through air pollution control policies and prescribed fire practices.

19.3. Materials and methods

19.3.1. Experimental design

In order to test these hypotheses, a simulation study was performed based on a California mixed-conifer forest site (Camp Paivika, CP) experiencing a high N deposition rate (71 kg N ha^{-1} yr^{-1}). The following N treatments were selected: N5, N15, N25, N40, and N70 corresponding to 5, 15, 25, 40, and 70 kg N ha^{-1} yr^{-1}, respectively. To estimate the benefits of different air pollution control policies, 0%, 25%, 50%, 75%, and 100% reductions in N deposition were considered for all the N deposition scenarios starting from 2006 and ending in the year 2210. Also, the influence of varying prescribed fire intervals on N cycling were tested for the selected N deposition scenarios, consisting of 15-, 30-, and 60-year intervals (PF15, PF30, and PF60, respectively), or no prescribed fires from 2006 to 2210. Wildfire occurred twice throughout the simulation, in the years 2100 and 2200.

The parameters chosen to evaluate the effect of the different treatments on N cycling were (1) fuel loads, N content in fuel loads, and air pollution emissions from wildfire; (2) nitrate export in stream water; and (3) nitrogenous trace gas emissions from soil. Long- and short-term ecosystem responses to N deposition and fire were assessed.

19.3.2. N Biogeochemical cycling simulation

DAYCENT version 4.5 was used in this study. The DAYCENT biogeochemistry model (Del Grosso et al., 2000; Parton et al., 1998; Parton et al., 2001) is the daily time step version of the CENTURY model (Parton et al., 1993). It simulates the biogeochemical processes of carbon, nitrogen, phosphorus, and sulfur associated with soil organic matter (SOM) in multiple ecosystem types. One feature of DAYCENT important to this study is the improvement of N cycling algorithms, including the simulation of N_2O, NO, and N_2 emissions resulting from nitrification and denitrification, and detailed soil N mineralization-immobilization processes associated with various organic matter pools (including microbial), which are distinguished by their decomposition rates. Required inputs to the model include daily maximum/minimum temperature and precipitation, site-specific soil properties and hydraulic characteristics, and current and historical land use.

Model runs were initialized and stabilized by repeatedly using site-specific weather data for 900 years (i.e., initially started from 1000 AD), background N deposition (1.0 kg N ha^{-1} yr^{-1}), and fire occurrence every 100 years. The model provides approximations of historical tree growth, litter C, SOM, and C:N ratios in the study area (Allen et al., 2007; Fenn et al., 2005; Grulke et al., 2001; Grulke & Balduman, 1999). Some parameters were modified in order to achieve reasonably well-fitted outputs with observations.

For model runs, N deposition was maintained at a background level of 1.0 kg N ha^{-1} yr^{-1} from 1900 to 1930 and gradually increased linearly to the targeted values from 1931–1940. N deposition remained at the targeted values from 1941 to 2005 allowing annual random 10% variations in those values. The rationale and criteria followed to calibrate the model can be found elsewhere (Fenn et al., 2008). The various N deposition and prescribed fire scenarios were applied from 2006 to 2210, the year simulations ceased. Wildfire events occurred in the years 2100 and 2200.

To estimate the benefits of applying different air pollution control policies and forest management protocols in forests where the N-deposition levels have exceeded the empirical critical load for "N as a nutrient" effects (17 kg N ha^{-1} yr^{-1}; Fenn et al., 2008), only the N25 and N70 scenarios were considered. The parameters simulated for these evaluations were NO_3^- leaching and N emissions from soil.

19.3.3. Estimation of wildfire emissions

The First Order Fire Effects Model (FOFEM 5.0; Reinhardt, 2003) was selected to estimate air pollution emissions from wildfire events. The fuel loads corresponding to the different N deposition scenarios in the year 2099, 1 year before the first wildfire event occurred in the simulation, were estimated by DayCent and used as inputs for the FOFEM model. The selected inputs for FOFEM were Pacific West Region, SAF 243 Sierra Nevada Mixed-Conifer Forest cover type, and natural fuel type. The wildfire occurred in fall under very dry conditions. Duff was set at the default light value for this ecosystem type (20 t $acre^{-1}$), with a depth of 2.5 in. and 20% humidity. A centered log distribution was selected for woody fuels greater than 3 in. Relative humidity for woody fuel >3 in. in diameter, and 0.25–1 in. was 10% and 6%, respectively, as calculated by the model under the selected very dry conditions. These harsh conditions were chosen to simulate a likely scenario of wildfire occurrence in the area. Simulations were carried out considering three different wildfire severities: 100% crown burnt, 75% crown burnt, and 50% crown burnt. The other input variables remained constant for the three wildfire severity simulations.

The total fuel load and the percentage of fuel consumed for each fuel class were calculated by the model. The air pollution emissions from the different wildfire severities and two N-deposition scenarios (N25, N70) were also calculated by the model providing estimations of particulate matter (PM_{10} and $PM_{2.5}$), methane (CH_4), carbon monoxide (CO), carbon dioxide (CO_2), nitrogen oxides (NO_x), and sulfur dioxide (SO_2) emissions in lbs $acre^{-1}$. Since the FOFEM model does not allow for evaluations of the influence of N concentrations of the fuel material, the NO_x emission factors (EF) were recalculated using two procedures (Dennis et al., 2002; Lacaux et al., 1996) based on N concentrations in the different fuel materials. The two methods differ based on the algorithms for the calculation of NO_x EF as can be found in Eq. (19.1) (Dennis et al., 2002) and Eq. (19.2) (Lacaux et al., 1996).

$$\text{EF NO}_x(\text{lb ton}^{-1}) = -3 + 7.8\ (0.7 \times \%\text{fuel N}) \qquad (19.1)$$

$$\text{EF NO}_x(\text{g kg}^{-1}) = 9.5 \times \text{N}(\%) - 0.49 \qquad (19.2)$$

The default EF value of 1.14 g kg^{-1} was applied following Dennis et al. (2002) for those fuel types for which N concentration was not provided by the DayCent model (shrubs and herbaceous components) or for woody materials, because negative values were obtained when applying Eqs (19.1 or 19.2). This default EF value is close to the 1.17 value provided by Urbanski et al. (2008) as an average EF for temperate forests. The final value of NO_x emissions was obtained by multiplying the estimated EFs by the amount of fuel burned calculated from FOFEM.

19.4. Results

Long- and short-term interactive effects of N deposition, forest management practices, and reduced air pollution were found.

19.4.1. Long-term responses to N deposition and fire management

19.4.1.1. Fuel Loads and N Content in Fuel Load

Increasing N-deposition rates increased forest fuel loads. The largest difference in fuel loading in the year preceding wildfire occurrence (2099) was found between the N5 and N70 deposition levels, resulting in a 121% increase (Fig. 19.1). The long-term effects of N-deposition rates over 5 kg N ha^{-1} yr^{-1} were irreversible as N reduction was ineffective in reducing fuel loads to the N5 levels. This was the case even when 100% reductions in N deposition were considered (Fig. 19.1a). However, prescribed fire was very effective in reducing fuel loads to N5 levels, even under the N70 deposition scenario. PF15 and PF30 were more successful in this regard than PF60 (Fig. 19.1b).

Similarly, N content in fuel load also increased with increasing N deposition. The highest N concentration increases were found in litter (data not shown), which were slightly higher than for foliage. The critical value of 12 g kg^{-1} foliar N was reached for N-deposition rates $\geqslant$25 kg N ha^{-1} yr^{-1} (Fig. 19.2a), in agreement with the empirical critical load value for this parameter proposed by Breiner et al. (2007), which was

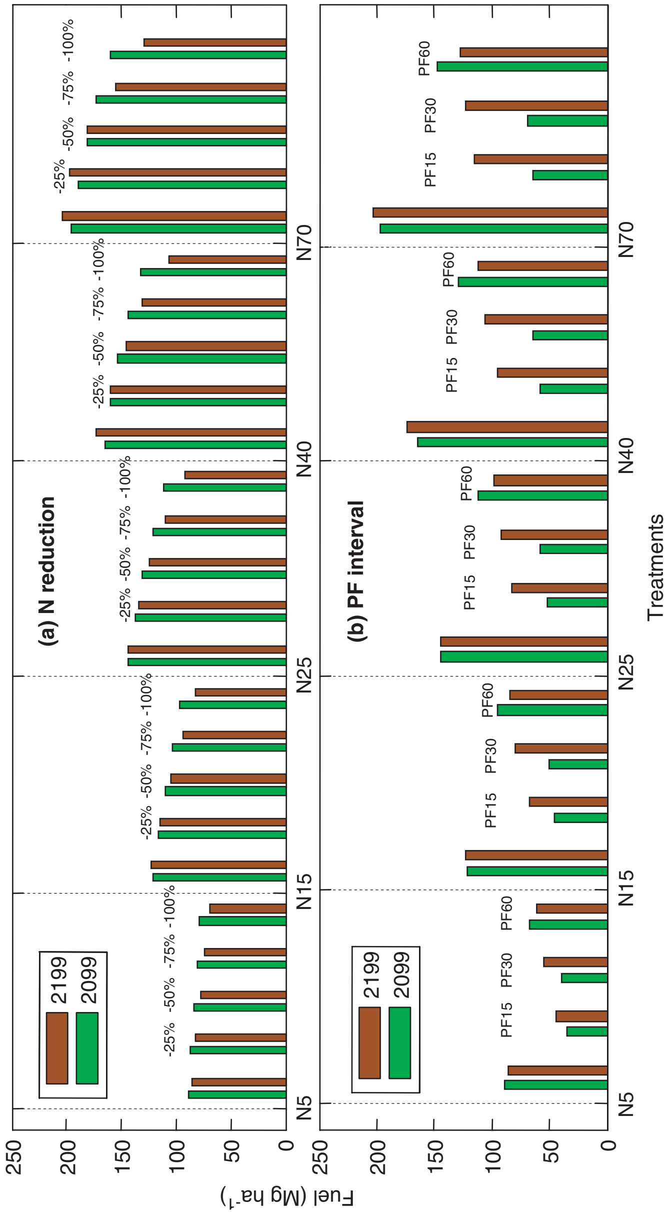

Figure 19.1. Fuel loads in the years right before wildfire occurrence under two simulation scenarios: (a) 0%, 25%, 75%, and 100% of N deposition abatement beginning in 2006, and (b) prescribed fire (PF) performed at 15-, 30-, and 60-year intervals. N5, N15, N 25, N40, and N70 correspond to N-deposition rates of 5, 15, 25, 40, and 70 kg N ha^{-1} yr^{-1}, respectively.

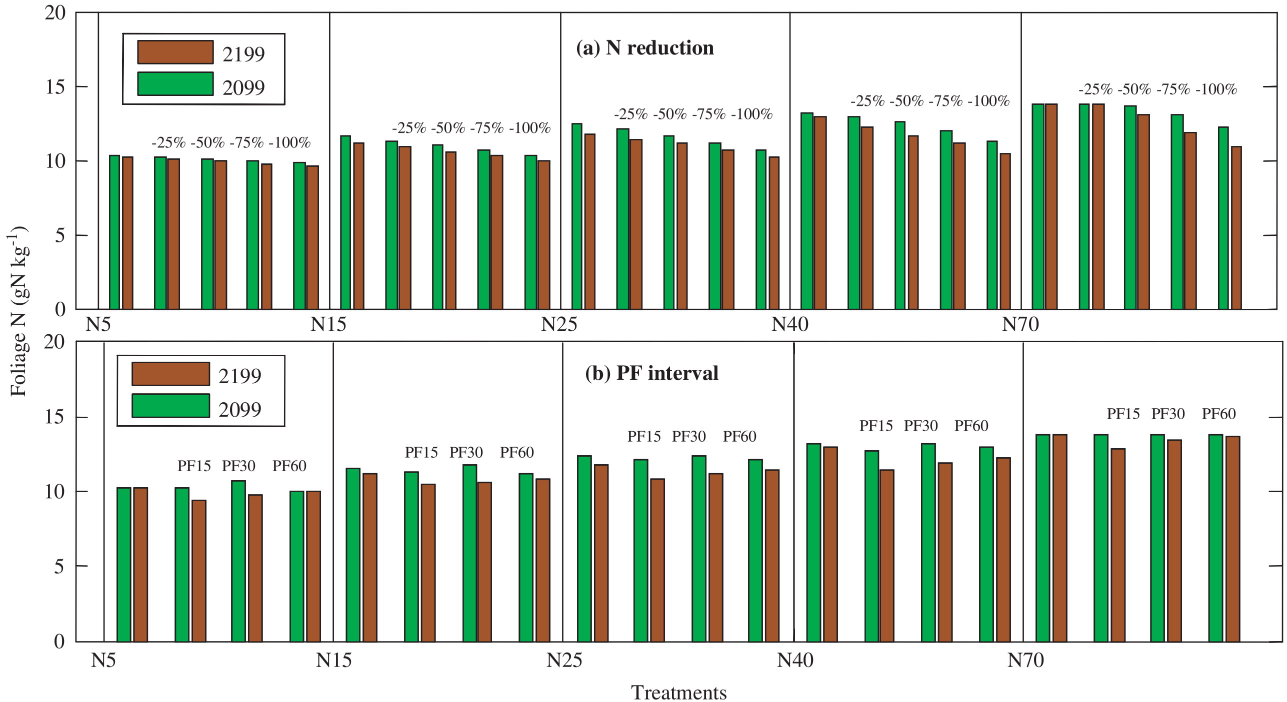

Figure 19.2. Foliage N concentration in the years right before wildfire occurrence under two simulation scenarios: (a) 0%, 25%, 75%, and 100% of N-deposition abatement beginning in 2006, and (b) prescribed fire (PF) performed at 15-, 30-, and 60-year intervals. N5, N15, N25, N40, and N70 correspond to N-deposition rates of 5, 15, 25, 40, and 70 kg N $ha^{-1} yr^{-1}$, respectively.

26 kg N ha^{-1} yr^{-1}. The percent of N-deposition reduction needed to return the system to foliage N concentrations below the critical value depended on the N-deposition scenario: 50% and 75% reductions for the N25 and N40 scenarios. Interestingly, the foliar N concentration did not return to levels below the critical value under the N70 scenario, even when 100% reductions in N-deposition rates were considered. Prescribed fire treatments were not effective in reducing foliar N when N-deposition rates were over 25 kg ha^{-1} yr^{-1} (Fig. 19.2b).

19.4.1.2. Wildfire Air Pollution Emissions

Fire severity influenced wildfire emissions of some pollutants (data not shown). For a given N-deposition scenario, fire severity, considered as percentage of crown burned, had no effect on the emissions of PM_{10}, $PM_{2.5}$, CH_4 or CO. However, when comparing 50% and 100% of the crown burned, emissions differed by 7–9% for CO_2, and 5–7% for SO_2. When the same comparison was performed to evaluate the effect of fire severity on NO_x emissions, large differences were found depending on the method used, ranging from 72–93% for FOFEM, 5% for Dennis et al. (2002) and 7–13% for the Lacaux et al. (1996) method.

As a result of the influence of N deposition on fuel loads, wildfire air pollution emissions also increased with N deposition. To evaluate the effect of N deposition on wildfire emissions, a wildfire that burned 75% of the canopy was simulated in the year 2100. No prior fire occurred after 1900 (to simulate fire exclusion policies), and N deposition was set at 5 and 70 kg ha^{-1} yr^{-1}. Emissions of PM_{10}, $PM_{2.5}$, CH_4, CO, CO_2, and SO_2 following the 2100 wildfire were about 70% higher in the N70 than the N5 treatment (Figs. 19.3a and b). Estimates of the increase in wildfire NO_x emissions between the N25 and N70 treatments when 75% of the crown was burned varied from 56% to 210% (Fig. 19.4) according to the three methods used. The increased NO_x emissions estimated by the FOFEM model were attributed to increased fuel build up in the N70 scenario. NO_x emissions increased by 56% in the N70 treatment compared to the N5 scenario according to FOFEM. However, these differences increased considerably (ranging from 166% to 210%) when comparing the N70 and N5 scenarios by the two methods that consider the influence of N-deposition rates on fuel N content (Fig. 19.4).

19.4.1.3. N Export in Stream Water

N deposition had a large effect on N export in stream water. N export was double the background levels by around 1980, 40 years after

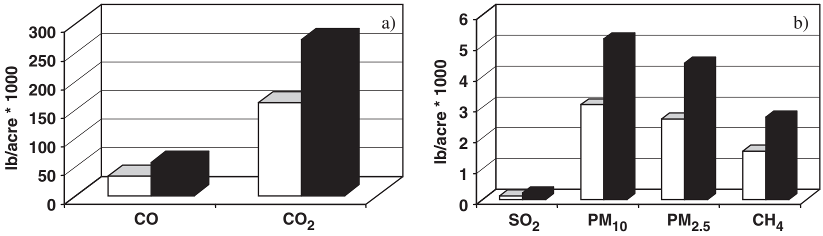

Figure 19.3. Estimation of CO, CO_2, SO_2, PM_{10}, $PM_{2.5}$, and CH_4 emissions from a wildfire (75% of the crown burned) occurring in the year 2100 and under N-deposition rates of 5 (white bars) and 70 (black bars) $kg\,ha^{-1}\,yr^{-1}$. The previous wildfire occurred prior to 1900.

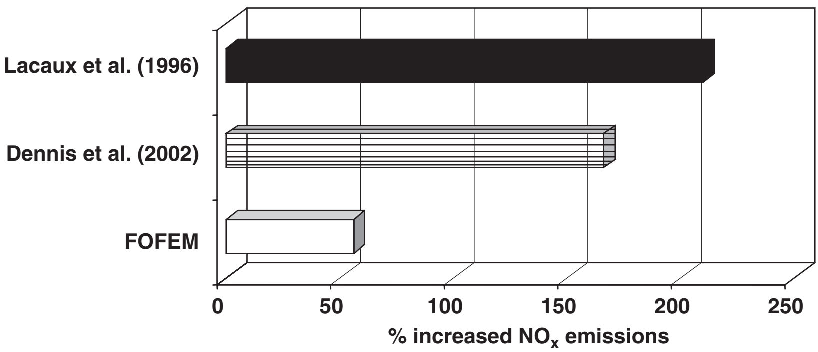

Figure 19.4. Percentage of increased NO_x emissions from a wildfire (75% of the crown burned) under N70 relative to N25 N-deposition scenarios as calculated by three different methods (Dennis et al., 2002; Lacaux et al., 1996, and FOFEM). The wildfire occurred in the year 2100, and the previous wildfire occurred prior to 1900.

N deposition increased to 25 $kg\,ha^{-1}\,yr^{-1}$ in the N25 scenario. Maximum N export reached up to five times background levels after 169 years of N deposition in 2109. The fastest return to background levels of NO_3^- (8 years) was achieved by reducing N deposition by 100% (Fig. 19.5a). Deposition reductions of 25 to 75% were proportionately less effective than the 100% reduction scenario (Fig. 19.5a). However, NO_3^- concentrations in the 75% reduction treatment were only slightly elevated over the 100% reduction.

Prescribed fire treatments alone did not reduce N runoff to basal levels. However, the combination of prescribed fire and N-deposition abatement was very effective in reaching stable N-export values in the background level

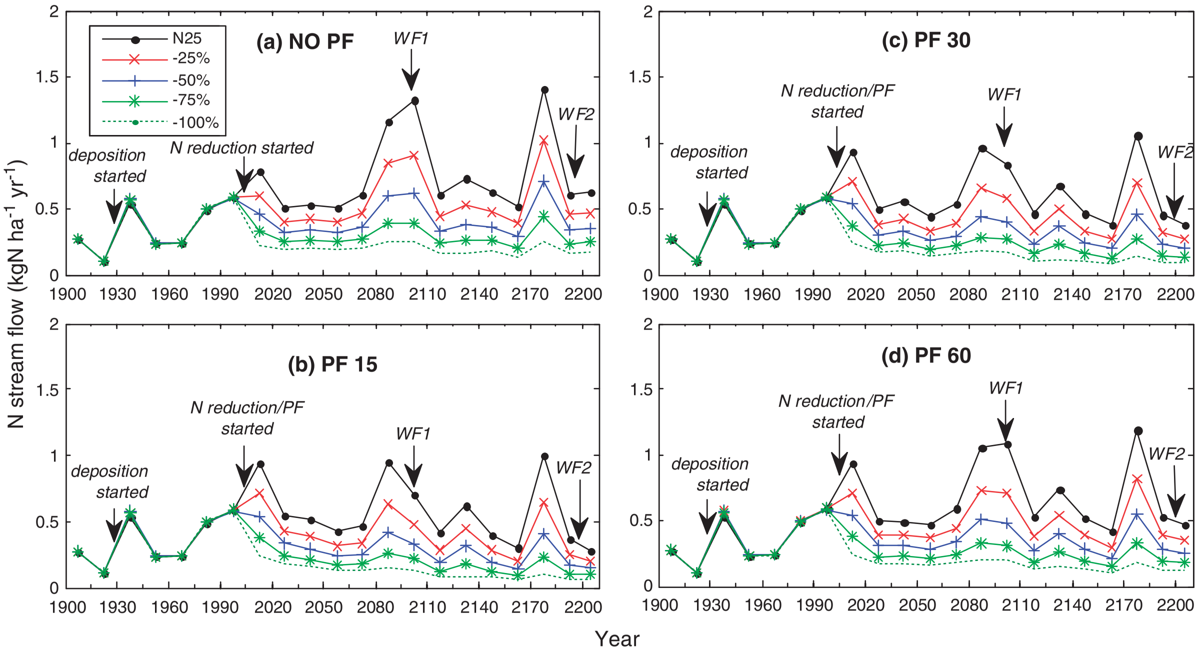

Figure 19.5. N leaching losses under the N-deposition scenario of 25 kg N ha^{-1} yr^{-1}, averaged every 15 years, impacted by different N-deposition reductions and three prescribed fire treatments (PF): (a) no prescribed fire, (b) 15-year interval prescribed fires, (c) 30-year interval prescribed fires, and (d) 60-year interval prescribed fires. Black line, no N-deposition reduction; red line, 25% N-deposition reduction; blue line, 50% N-deposition reduction; solid green line, 70% N-deposition reduction; and dark green dotted line, 100% N-deposition reduction. WF1 and WF2, occurrence of wildfire events.

range even after wildfire occurrence, particularly with a 75% N-deposition reduction and any prescribed fire interval treatment (Figs. 19.5b–d). Similarly, the combination of PF15 and 50% reductions ended in runoff values only slightly higher than background levels (Fig. 19.5b).

Because returning N losses to background levels may not be a realistic goal in the short term for heavily affected sites, an acceptable level of 0.5 kg N ha^{-1} yr^{-1} in stream water NO_3^- export was set based on previous research (Fenn et al., 2008). Under the N25 scenario, this threshold could be reached by using different strategies such as (1) 75% reduction in N deposition with or without prescribed fire, (2) 50% reduced N deposition and prescribed fire of any interval, and (3) 25% reduction in N deposition and prescribed fire, although with this level of deposition reduction there were three periods of N runoff slightly above the threshold (Fig 19.5). Under the N25 scenario, prescribed fire alone resulted in many periods when stream water NO_3^- export was below the 0.5 kg N ha^{-1} yr^{-1} NO_3^- runoff threshold, although N-deposition reductions $\geqslant$50% were needed to maintain N-runoff levels below the threshold (Fig. 19.5). In the absence of prescribed fire, N runoff exceeded the threshold intermittently in the 50% N-reduction treatment (Fig. 19.5).

Under the N70 scenario, N export was 7 times higher than background levels by 1980, 40 years after N deposition increased to 70 kg ha^{-1} yr^{-1}. Nitrogen export reached as much as 73 times higher than background levels after 169 years of N deposition in the N70 treatment (Fig. 19.6). Air pollution control polices or prescribed fire treatments alone were ineffective in reducing NO_3^- in runoff to background levels. Under the N70 scenario, the only way to completely return the system to basal levels, admittedly an unrealistic near-term goal, was to combine prescribed fire and reduced N deposition. A 100% N-deposition reduction and PF15 were needed to achieve this goal within 52 years. The PF30 treatment combined with 100% N-deposition reductions resulted in runoff levels slightly higher than background values.

The previously established acceptable N-export level (0.5 kg ha^{-1} yr^{-1}) was eventually approximated in the N70 scenario with 100% reductions in N deposition. When N deposition was reduced by 75–100% in combination with any prescribed fire treatment, N export also decreased to acceptable levels (Fig. 19.6). It should be appreciated that even a 50% reduction in N-deposition inputs, particularly when combined with any prescribed fire interval, also resulted in dramatic decreases in NO_3^- export both before and after wildfire (Fig. 19.6). If N deposition was decreased by at least 50% then a prescribed fire interval of 30 years was nearly as effective as the PF15 treatment (Fig. 19.6).

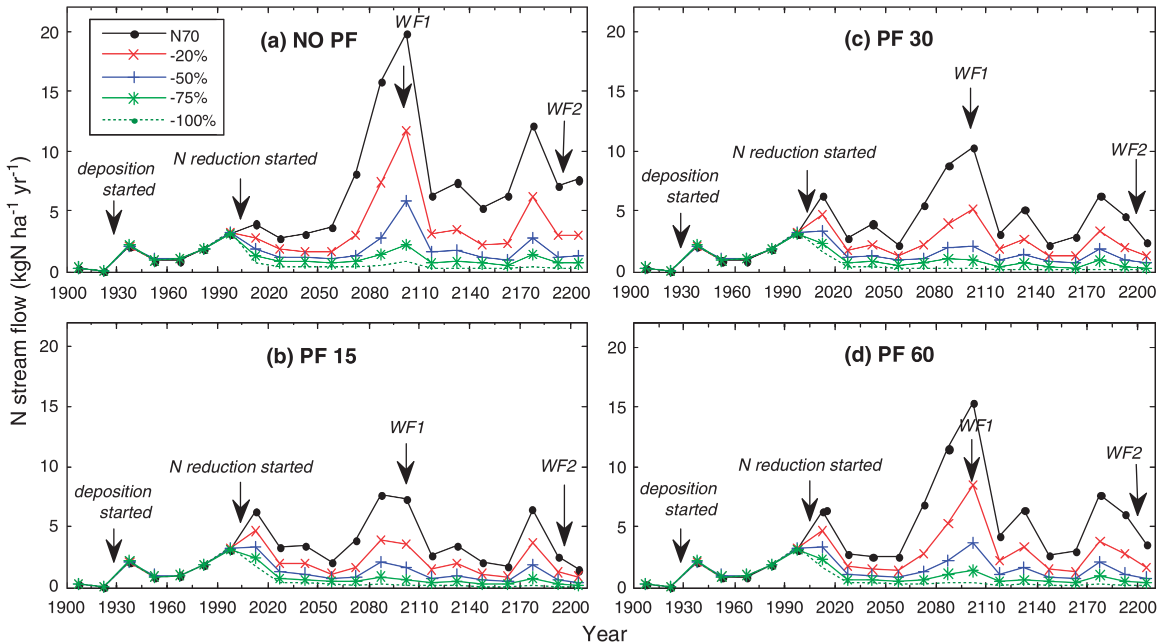

Figure 19.6. N leaching losses under the N-deposition scenario of 70 kg N ha^{-1} yr^{-1}, averaged every 15 years, impacted by different N-deposition reductions and three prescribed fire treatments (PF): (a) no prescribed fire, (b) 15-year interval prescribed fires, (c) 30-year interval prescribed fires, and (d) 60-year interval prescribed fires. Black line, no N-deposition reduction; red line, 25% N-deposition reduction; blue line, 50% N deposition reduction; solid green line, 70% N deposition reduction; and dark green dotted line, 100% N-deposition reduction. WF1 and WF2, occurrence of wildfire events.

19.4.1.4. Soil N Emissions

Increased N deposition enhanced soil N emissions, with N losses due to this process higher than N losses in stream water for a given N-deposition input. Prior to the year 2100 soil N emissions under the N25 and N70 scenarios reached values that were 27 and 144 times higher, respectively, than basal levels (Figs. 19.7–19.8). Soil emissions could be most efficiently reduced by combining air pollution abatement policies and prescribed fire treatments. Under the N25 scenario, background levels could only be achieved by combining 100% N-deposition reductions and PF15 for 82 years (Fig. 19.7). In the case of the N70 scenario, air pollution control and/or prescribed fire did not result in baseline soil emissions. An N-deposition reduction of 100% and a 15-year prescribed fire interval when maintained for 82 years resulted in soil N-emission levels four times greater than basal levels (Fig. 19.8).

Prescribed fire alone with a 15-year return interval reduced maximum gaseous N emissions by approximately 50% in both the N25 and N70 treatments (Figs. 19.7–19.8). In the N70 scenario, even with a 75% reduction in deposition after 22 years of N deposition control, annual N emissions from soil were 7–10 $kg^{-1}\,ha^{-1}$, and when PF15 was combined with the 75% reduced deposition, soil emissions were still elevated, ranging from 3–7 $kg\,ha^{-1}\,yr^{-1}$. By comparison, when the same period was considered in the N25 scenario, a 75% reduction in deposition combined with the PF15 treatment resulted in soil trace gas emissions ranging from values slightly over 2 $kg\,ha^{-1}\,yr^{-1}$ to less than 1 $kg\,ha^{-1}\,yr^{-1}$ (Figs. 19.7–19.8).

19.4.2. Short-term ecosystem responses to N deposition and fire management

Prescribed fire occurrence resulted in consistent patterns of N losses regardless of N deposition scenarios. Soil N emissions peaked 1 month after prescribed fire took place, while N export to stream water peaked 11, 23, and 35 months after prescribed fire (Figs. 19.9–19.12). However, the magnitude of these effects was influenced by N deposition. For instance, maximum nitrate leaching was 10 times higher for N70 than for N25 scenarios. Similarly, peak soil N emissions under the N70 scenario were double those of the N25 treatment following fire (Figs. 19.9–19.12), and also exhibited unusually high peak emissions during the second year postfire (Fig. 19.10). The PF15 treatment was very effective in reducing N export to stream water after prescribed fire, comparable to a 75% reduction in N deposition under the N25 scenario, and was more effective in the first year than a 100% reduction in N deposition under the N70 scenario (Figs. 19.10–19.11). In the N25 scenario the PF15 treatment or a

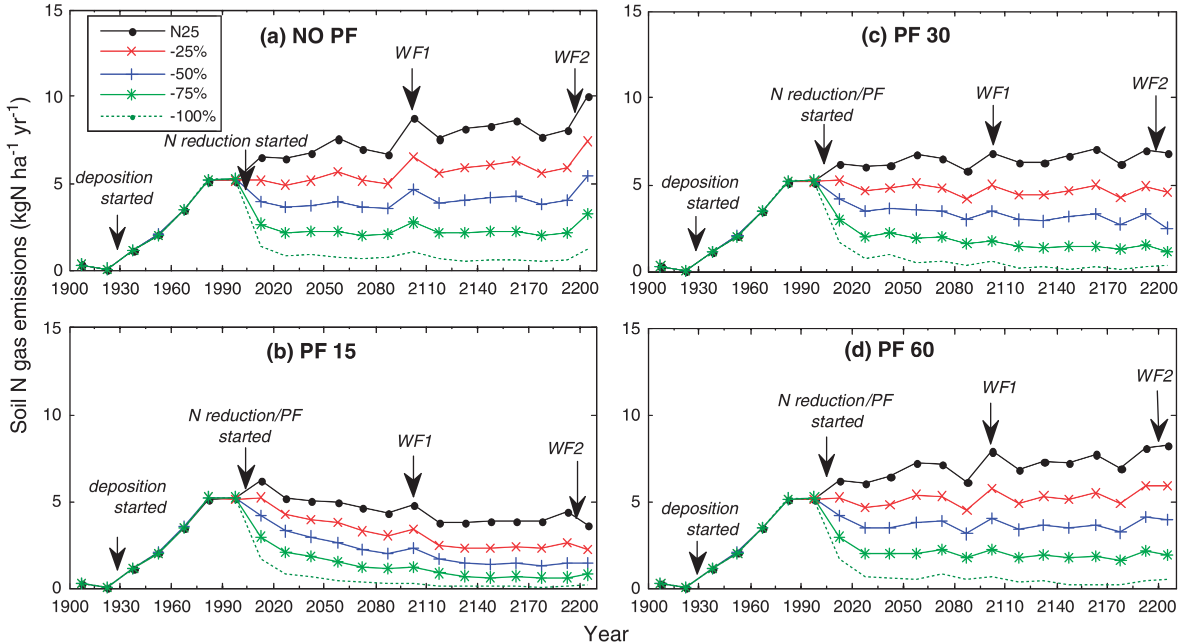

Figure 19.7. Soil N trace gas emissions under the N-deposition scenario of 25 kg N ha^{-1} yr^{-1}, averaged every 15 years, impacted by different N-deposition reductions and three prescribed fire treatments (PF): (a) no prescribed fire, (b) 15-year interval prescribed fires, (c) 30-year interval prescribed fires, and (d) 60-year interval prescribed fires. Black line, no N-deposition reduction; red line, 25% N-deposition reduction; blue line, 50% N-deposition reduction; solid green line, 70% N-deposition reduction; and dark green dotted line, 100% N-deposition reduction. WF1 and WF2, occurrence of wildfire events.

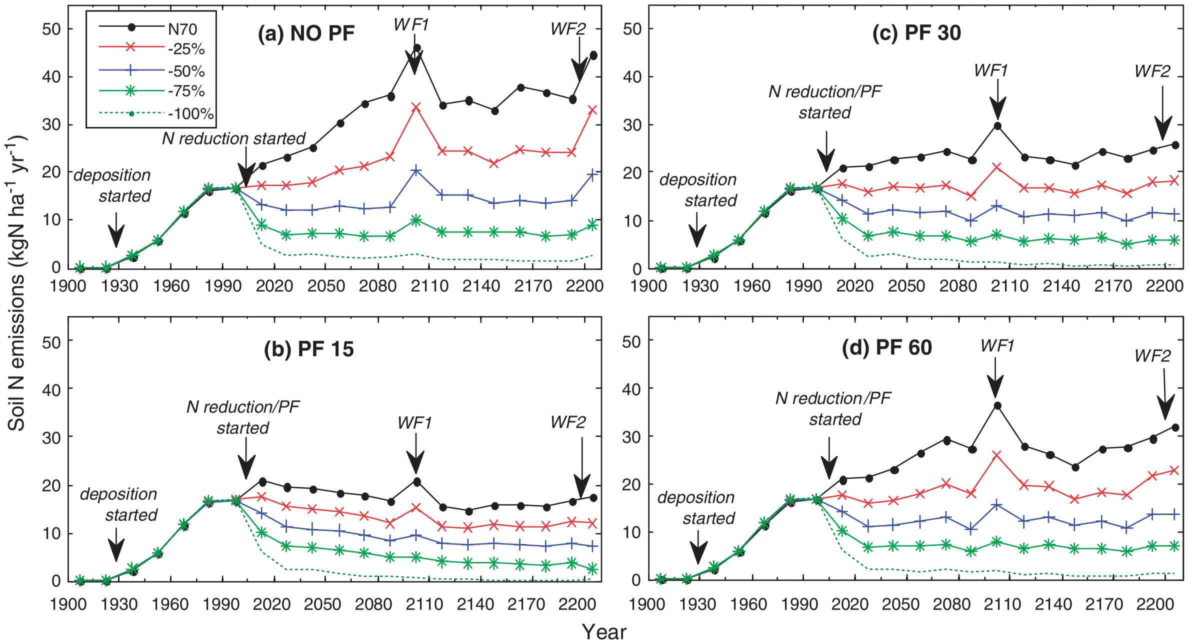

Figure 19.8. Soil N trace gas emissions under the N-deposition scenario of 70 kg N ha^{-1} yr^{-1}, averaged every 15 years, impacted by different N-deposition reductions and three prescribed fire treatments (PF): (a) no prescribed fire, (b) 15-year interval prescribed fires, (c) 30-year interval prescribed fires, and (d) 60-year interval prescribed fires. Black line, no N-deposition reduction; red line, 25% N-deposition reduction; blue line, 50% N-deposition reduction; solid green line, 70% N-deposition reduction; and dark green dotted line, 100% N-deposition reduction. WF1 and WF2, occurrence of wildfire events.

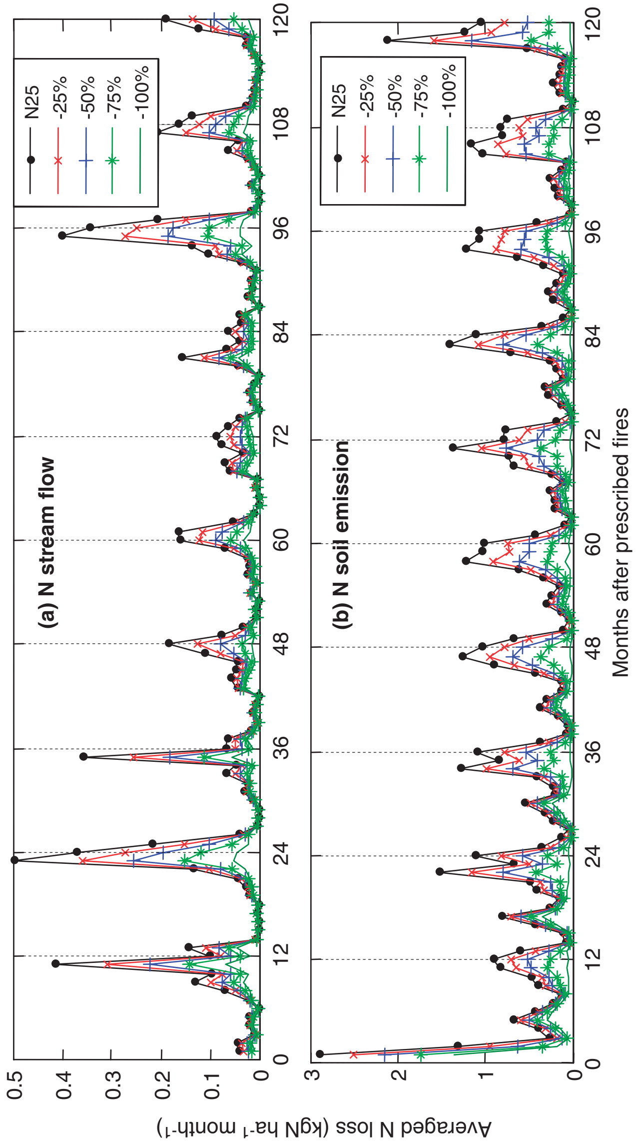

Figure 19.9. N in stream flow (a) and soil trace gas emissions (b) after prescribed fires (PF) under the N deposition scenario of 25 kg N ha^{-1} yr^{-1}, averaged for three PF treatments, impacted by different percentages of N-deposition reduction. Black line, no reduction; red line, 25% reduction; blue line, 50% reduction; light green line, 75% reduction; dark green line, 100% reduction.

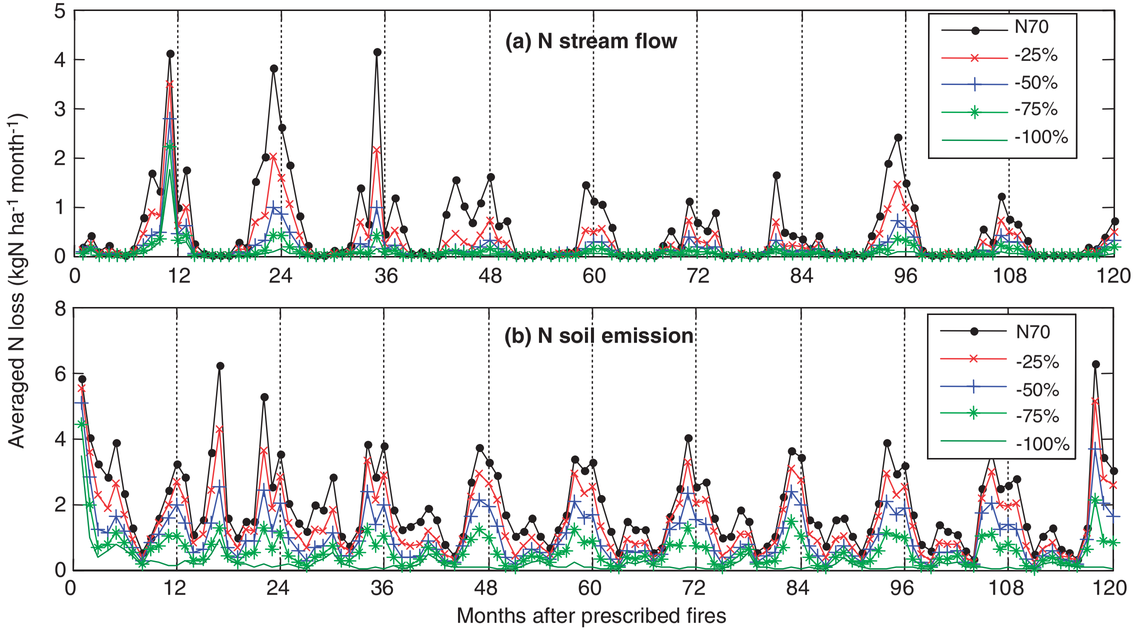

Figure 19.10. N in stream flow (a) and soil trace gas emissions (b) after prescribed fires (PF) under the N deposition scenario of 70 kg N ha^{-1} yr^{-1}, averaged for three PF treatments, impacted by different percentages of N-deposition reduction. Black line, no reduction; red line, 25% reduction; blue line, 50% reduction; light green line, 75% reduction; dark green line, 100% reduction.

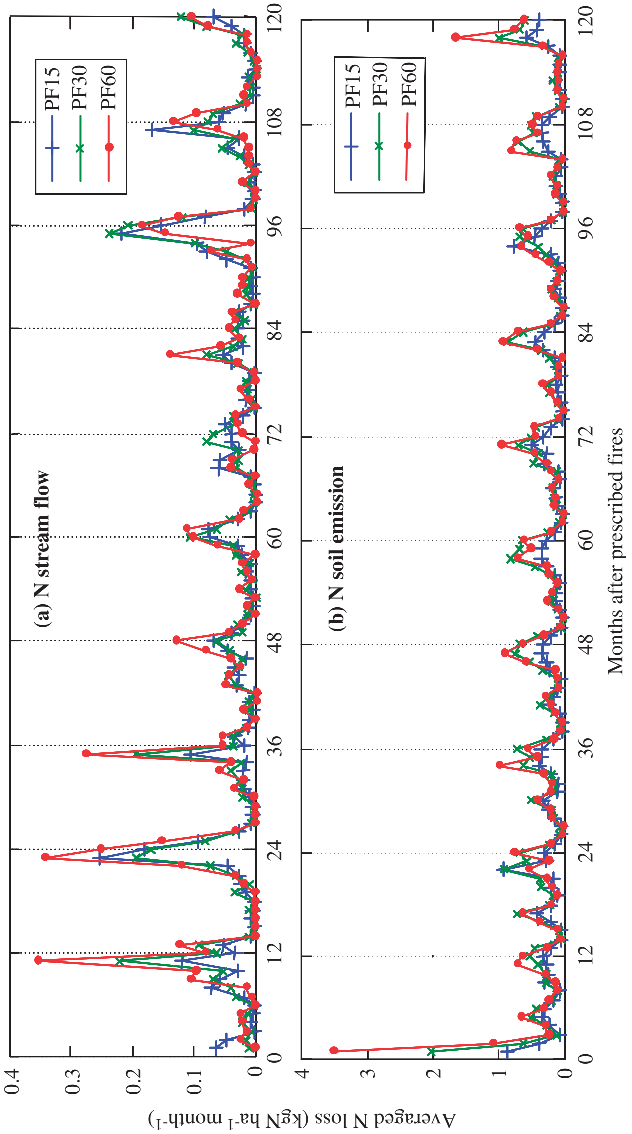

Figure 19.11. N in stream flow (a) and soil trace gas emissions (b) after prescribed fires (PF) under a deposition scenario of 25 kg N ha^{-1} yr^{-1} and considering three different PF intervals: 15 years (blue line), 30 years (green line), and 60 years (red line).

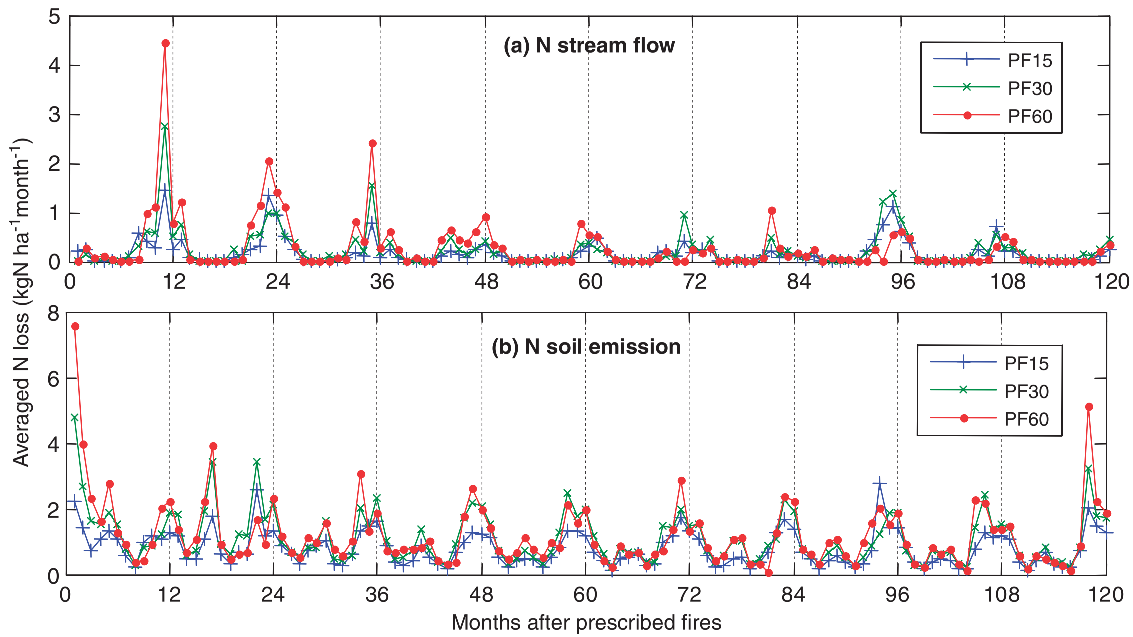

Figure 19.12. N in stream flow (a) and soil trace gas emissions (b) after prescribed fires (PF) under a deposition scenario of 70 kg N ha^{-1} yr^{-1} and considering three different PF intervals: 15 years (blue line), 30 years (green line), and 60 years (red line).

75% reduction in N deposition were the most effective treatments for mitigating short-term soil N losses following prescribed fire. In the N70 scenario only a 100% reduction in N deposition reduced peak soil emissions below $1\,kg\,ha^{-1}\,mo^{-1}$, but the PF15 or 75% N deposition reduction treatments reduced peak soil emissions to approximately $1.5–2.0\,kg\,ha^{-1}\,mo^{-1}$, compared to emissions of $4–6\,kg\,ha^{-1}\,mo^{-1}$ in the N70 treatment (Figs. 19.9–19.12).

19.5. Discussion

19.5.1. Simulation of ecosystem N losses

The simulations demonstrated that increased N deposition enhanced fuel-load buildup, soil nitrogen emissions, and N runoff to stream water, in agreement with previously reported field observations from California mixed-conifer forests (Fenn & Poth, 1999, 2001; Fenn et al., 2003c; Meixner & Fenn, 2004) and other ecosystems of the western United States (Fenn et al., 2003a). Simulated N emissions from soil were greater than N runoff regardless of the N deposition scenario. This result should be interpreted cautiously, considering that current parameterization of the DayCent model for CP appears to overestimate soil N emissions and underestimate stream water NO_3^- concentrations (Fenn et al., 2008; Fenn & Poth, 2001). Li et al. (2006) also reported that DayCent underestimated stream NO_3^- export in January in a small chaparral catchment in Sequoia National Park in central California. It should also be noted that DayCent does not simulate stream water NO_3^- concentrations per se; values are actually of soil seepage water NO_3^-. However, the soil emission patterns forecasted by the simulation properly matched previous field observations. Uncertainties apply to other aspects of the models used in this study as well. For instance, the co-exposure of this type of forest to N deposition and the elevated ozone concentrations occurring in southern California was not addressed in our simulations except to the extent that the model was parameterized with site-specific data from the San Bernardino Mountains, which would inherently include to some degree the effects of ozone in the field data used as input to DayCent. For example, the combined effects of ozone and N deposition are known to have major effects on C cycling and organic matter accumulation at CP (Fenn et al., 2003c).

These uncertainties in simulated N-cycling processes also feed into uncertainty in estimates of N-induced changes in fuel loads and therefore will influence the estimation of air pollutant emissions following wildfire.

A standardized protocol to estimate the influence of N concentrations in fuel on fire emissions would be very useful, as large differences in NO_x emissions were found when applying different methods (see below). Also, the influence of some global change components, such as increased atmospheric CO_2 concentrations and changes in temperature and precipitation regimes in the region, was not considered. Nevertheless, the rates of N deposition considered in our study ranging from 5 to 70 kg N ha^{-1} yr^{-1} properly covers the range of deposition inputs measured in southern California (Breiner et al., 2007; Fenn et al., 2008) and other areas of the western United States affected by N emissions (Fenn et al., 2003b).

19.5.2. Fire emissions and N deposition

The output of our simulation exercises highlight the existence of important N deposition and fire interactions, as increasing atmospheric N inputs resulted in larger air pollution wildfire emissions, including greater N emissions from soil 1 month after prescribed fire, and increased N runoff levels 11, 23, and 35 months after prescribed fire. These results also demonstrate the importance of including the effects of N deposition on fuel load and fuel chemistry in the models currently available for estimating fire impacts on ecosystems. Estimated NO_x emissions from wildfire events increased from 166% to 210% when comparing the lowest (N5) and highest (N70) N-deposition levels based on two methods that considered both fuel loads and fuel N concentration. These increases are three to four times higher than would be predicted by using models such as FOFEM, which only consider fuel loads and not N concentrations in fuels. The differences between these two types of models were largely due to the N content in the most flammable materials, such as foliage and litter. This finding is in agreement with the measurements recently carried out by Yokelson et al. (2007) in forest fires occurring in the surroundings of Mexico City, an area experiencing high N-deposition rates (Fenn et al., 1999; Fenn et al., 2002). Emissions of hydrogen cyanide (HCN) and NO_x in the forests near Mexico City were three to four times greater than the average values from U.S. pine forests (Yokelson et al., 2007). These authors associated the high N-emission levels with N enrichment of fuels components.

Our simulations suggest that the combined effect of increased fuel loads and increased N fuel content caused the increases in the emission of N compounds following wildfires. Approximately 70% increases in the emission of PM_{10}, $PM_{2.5}$, CH_4, CO, CO_2, and SO_2 emissions were found when comparing N70 and N5 treatments, with subsequent implications for air quality and also for human health. Our results also show that N deposition increases the risk of fire occurrence, not only as a result of an increased fuel amount but also because litter represents a major portion of this increase and litter accumulation enhances forest flammability, leading also to increased wildfire severity, and according to the results of our simulation, to increases in NO_x, CO_2, and SO_2 emissions. These N deposition-wildfire interactions are of particular concern as the area burned by wildfires in ponderosa pine forests of the southwestern US has been increasing over the past three decades (Grady & Hart, 2006). In October 2003 alone 235,267 ha were burned by wildfire in southern California (Clinton et al., 2006). Much of the burned area is exposed to high levels of N deposition, which undoubtedly affected the level of emissions from the 2003 fires and from more recent burns in the area, including at CP and the surrounding areas.

19.5.3. Management options

Air pollution abatement and forest management strategies could be used to reduce the long-term effects of increased N deposition on ecosystem N cycling; however, their effectiveness will vary as influenced by the magnitude of N deposition and the ecosystem process or parameter considered. To reduce the costs and the time frame of air pollution control policies needed to reach background or at least acceptable levels of N losses from forests, the N deposition reduction targets required to reverse chronic undesirable effects in ecosystem functioning should be defined.

Air pollution control policies alone showed a limited ability to recover ecosystem functioning when impaired by elevated N deposition. For instance, with N deposition rates $\geqslant 15\,kg\ N\,ha^{-1}\,yr^{-1}$, air pollution control proved to be ineffective in returning fuel loads to the N5 levels, regardless of the time frame and the percentage of N deposition reduction that was applied. Nitrogen deposition reductions of 100%, a very unlikely and costly scenario, would be needed to reach background N runoff levels when forests are chronically exposed to $25\,kg\ N\,ha^{-1}\,yr^{-1}$, although 75% N reduction resulted in acceptable levels of NO_3^- runoff. A 100% reduction of N deposition would be required to achieve marginally acceptable levels of NO_3^- runoff in forests that experience N deposition rates over $70\,kg\ N\,ha^{-1}\,yr^{-1}$. A 100% reduction in N deposition caused a dramatic reduction in soil N emissions, but when combined with prescribed fire, emissions decreased further, eventually to levels observed with background N deposition.

Similarly, when the prescribed fire treatments were implemented singly, they were not effective in completely reversing the influence of N deposition in N ecosystem cycling, except for the reductions in fuel loads. The combination of prescribed fire and air pollution control proved to be a successful way to mitigate N-deposition effects on ecosystem N cycling and provided more options for reaching acceptable N runoff levels, although this became more difficult with extremely high N-deposition rates (e.g., 70 kg ha^{-1} yr^{-1}). Under the N25 scenario this goal could be achieved by combining PF15 and 50% reduction in N deposition or with a 75% reduction in N deposition even without prescribed fire. A more intense deposition reduction resulted in a shorter time period to reach to an acceptable N-runoff level. As this simulation study and field manipulation experiments (Gundersen et al., 1998) have demonstrated, in high deposition areas even large reductions in N deposition do not readily lead to significant reductions in the N content of key fuel components, such as litter and foliage. This is probably the result of a chronic excess of N availability, resulting in the luxurious consumption of this normally-limiting nutrient. Thus, it may be that only forest thinning, prescribed fire, and reduced N deposition can return these mixed-conifer forests to a less fire-prone and N-rich condition.

However, the use of prescribed fire to reduce fuel loads has been shown to have adverse impacts on key forest components and processes, such as soil properties (Moghaddas & Stephens, 2007), N runoff (Wan et al., 2001), increased mortality of older pine trees (Kolb et al., 2007), and wildlife (Tiedemann et al., 2000). In addition, the adverse effects of prescribed fire on air quality is a major public health issue, although this can be mitigated by conducting this forest management practice under the most favorable smoke dispersal conditions (Knapp et al., 2005). However, the release of CO_2 and other greenhouse gases is unavoidable (Carter & Foster, 2004). Moreover, there is experimental evidence showing that significant N losses from ecosystems occur as a result of prescribed fire treatments in areas experiencing high N-deposition loads (Johnson et al., 2008; Meixner et al., 2006; Riggan et al., 1994). Our study showed that this would be the case under the present N-deposition scenario at CP, especially when 30- or 60-year prescribed fire intervals are involved. However, soil N emissions or N runoff were almost four times lower and three times lower, respectively, when 15-year intervals were compared with the PF60 treatment. Therefore our results suggest that PF15 could be the best option to be combined with air pollution abatement policies to mitigate N-deposition impacts, although such frequent use of prescribed fire will be difficult to implement in these highly populated forests. Further research should be carried out to evaluate the potential use of alternative forest management techniques, such as thinning, possibly in tandem with less frequent prescribed fire, to be used singly or in combination with reduced N deposition to mitigate the long- and short-term adverse impacts of N deposition and fire events.

Under the N70 scenario unrealistic, costly and long-lasting 100% reductions would be needed to return ecosystem N losses to background levels. However, the 75% reduction in N deposition combined with prescribed fire may be the best scenario for reducing N losses, but even this is not realized in the short term. A 50% reduction in N deposition combined with prescribed fire does lead to large reductions in N export and would be considered as major progress towards mitigating the symptoms of N saturation. In forests, exposed to 20–30 kg N ha^{-1} yr^{-1}, a more common scenario in southern California than the extreme (70 kg N ha^{-1} yr^{-1}), we conclude that a 50% reduction in N deposition in combination with prescribed fire will be effective in returning the ecosystem to a more conservative state of N cycling.

19.6. Conclusions

The results of this simulation exercise highlight important interactions between N deposition and fire management practices. Forest fire suppression and increased N deposition in southern California contribute to increasing fuel loads and to an increase in fuel N content. Model simulations suggest that this will affect wildfire severity and result in increases in air pollution emissions from fires, increases in peak N emissions from soil right after fire, and increases in peak N export to stream water during the first three years postfire.

Both forest management and air pollution control policies could be used to mitigate these effects. Prescribed fire was effective in reducing fuel loads and mitigating short-term N export after fire events, while the combination of N-deposition reduction and prescribed fire was most effective in reducing long-term N losses to the atmosphere and in runoff. Implementation of 15-year prescribed fire intervals in these combinations would be the best option to avoid high N-ecosystem losses following fire, although in highly populated forests, such as in the San Bernardino Mountains, such short fire intervals are unlikely to be implemented. In any case, it would be extremely difficult to return N losses to near baseline levels when forests experience N deposition of 70 kg N ha^{-1} yr^{-1}, as even 100% reductions in N deposition for more than 80 years would not completely achieve this goal for some key N ecosystem cycling processes.

Therefore, early applications of air pollution control measures in combination with fuels reduction options such as prescribed fire treatments are recommended.

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