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A global meta-analysis of soil exchangeable cations, pH, carbon, and nitrogen with afforestation

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Abstract. Afforestation, the conversion of non-forested lands to forest plantations, can sequester atmospheric carbon dioxide, but the rapid growth and harvesting of biomass may deplete nutrients and degrade soils if managed improperly. The goal of this study is to evaluate how afforestation affects mineral soil quality, including pH, sodium, exchangeable cations, organic carbon, and nitrogen, and to examine the magnitude of these changes regionally where afforestation rates are high. We also examine potential mechanisms to reduce the impacts of afforestation on soils and to maintain long-term productivity.

Across diverse plantation types (153 sites) to a depth of 30 cm of mineral soil, we observed significant decreases in nutrient cations (Ca, K, Mg), increases in sodium (Na), or both with afforestation. Across the data set, afforestation reduced soil concentrations of the macronutrient Ca by 29% on average ($P < 0.05$). Afforestation by *Pinus* alone decreased soil K by 23% ($P < 0.05$). Overall, plantations of all genera also led to a mean 71% increase of soil Na ($P < 0.05$). Mean pH decreased 0.3 units ($P < 0.05$) with afforestation.

Afforestation caused a 6.7% and 15% ($P < 0.05$) decrease in soil C and N content respectively, though the effect was driven principally by *Pinus* plantations (15% and 20% decrease, $P < 0.05$). Carbon to nitrogen ratios in soils under plantations were 5.7–11.6% higher ($P < 0.05$). In several regions with high rates of afforestation, cumulative losses of N, Ca, and Mg are likely in the range of tens of millions of metric tons. The decreases indicate that trees take up considerable amounts of nutrients from soils; harvesting this biomass repeatedly could impair long-term soil fertility and productivity in some locations. Based on this study and a review of other literature, we suggest that proper site preparation and sustainable harvest practices, such as avoiding the removal or burning of harvest residue, could minimize the impact of afforestation on soils. These sustainable practices would in turn slow soil compaction, erosion, and organic matter loss, maintaining soil fertility to the greatest extent possible.

Key words: acidification; afforestation; base cations; salinity; soil carbon; soil nutrients; sustainable harvest.

Introduction

Afforestation, planting trees on land that has not previously been forested for at least 50 years, has been featured as a potential mechanism to sequester carbon dioxide (Vitousek 1991, Houghton et al. 1999, Wright et al. 2000, McCarl and Schneider 2001, Hoffert et al. 2002, Jackson et al. 2002, Jackson and Schlesinger 2004, Pacala and Socolow 2004, Lal 2008). Afforestation has also gained attention as a means for developed countries to mitigate their carbon emissions through offset programs such as the Clean Development Mechanism of the Kyoto Protocol. However, the fast growth rates of plantations compared to other vegetation types can lead to higher demand for soil nutrients (Mendham et al. 2003*b*, Merino et al. 2004, Zhang et al. 2004). Depending on how sustainably the harvested biomass is managed, its frequent removal can deplete soil nutrients from these ecosystems, lowering primary productivity of future rotations and reducing their long-term potential as carbon sinks (Bi et al. 2007). The goal of our study was to quantify the effects of afforestation on soil nutrients and to suggest forestry practices that can ameliorate any negative impacts of afforestation.

Globally, the scope of afforestation has rapidly increased in recent decades. As of 2005, roughly 140 million ha were grown as afforested plantations, with ~2.8 million more hectares afforested per year (FAO 2006*b*). The afforestation rate is likely to increase, since many plantations typically produce greater economic returns than native forests, particularly those of

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nonnative *Pinus* and *Eucalyptus* (Cubbage et al. 2006; see Plate 1). With a potential 34 million more hectares afforested by 2020, managers need to understand the long-term effects of plantation establishment on soils and how this could affect long-term productivity.

Plantations have many potential economic and ecological benefits beyond simple carbon sequestration. For example, afforestation of marginal agricultural and grazing lands can reduce soil erosion and diversify and improve revenues (Geary 2001, Cubbage et al. 2006). Plantations may grow faster than natural forests and produce more timber products per year, reducing the amount of land needed to meet wood demand globally (Wright et al. 2000). Sustainable harvest of afforested plantations could therefore reduce the loss of primary forest, preserving biodiversity (ABARE and Jaako-Poyry 1999, FAO 2001*b*). Forested plantations already contribute >35% of the world's industrial wood products, even though plantations account for only ~4% of the global forested area (ABARE and Jaako-Poyry 1999, FAO 2001*b*).

Afforestation can occur in many different forms. The UN Food and Agriculture Organization (FAO) defines afforestation as either the establishment of forests on historically treeless areas or on land cleared of native forests for at least 50 years. Although afforestation is typically conducted with fast-growing, exotic tree species, native species are used in a smaller subset of afforested areas. These different scenarios (treeless vs. deforested regions, exotic vs. native species) can potentially lead to different trajectories of ecosystem change, including different rates of C storage, nutrient depletion, and biomass increment; nevertheless, convergent trends such as the redistribution of soil nutrients to tree biomass and soil acidification may emerge. In this study we include data on many different pathways of afforestation in order to find effects that are common across afforestation scenarios.

By redistributing nutrients from soils to biomass, afforestation has potentially strong effects on plant macronutrients (Jobbagy and Jackson 2003, 2004*b*, Farley et al. 2008). Nutrient uptake and subsequent harvest and removal of biomass can deplete cations, including calcium (Ca), magnesium (Mg), and potassium (K) (Richter et al. 1994, Mendham et al. 2003*a*, Zhang et al. 2004). Jobbágy and Jackson (2003) showed that the redistribution of base cations from soils to biomass acidified the surface soil of *Eucalyptus* plantations in Argentina; this phenomenon was also observed globally for *Pinus* and *Eucalyptus* plantations (Jackson et al. 2005). Sodium redistribution caused by afforestation with *Eucalyptus* plantations can even salinize soils in some locations; in the Argentine pampas, for instance, afforestation caused a 4–19-fold salinization of soils and ground water compared to native grasslands (Jobbagy and Jackson 2004*a*).

Previous research, primarily in New Zealand and Australia, has shown that afforestation can significantly alter both soil carbon (C) and nitrogen (N) stocks. *Pinus* and *Eucalyptus* afforestation was shown to decrease soil carbon content by a mean of 10% (Davis and Condron 2002, Guo and Gifford 2002). In the same region, a study of afforestation with *Pinus radiata* found a reduction in total soil N by more than 45% (Parfitt et al. 2003*a*, *b*). In addition, afforestation in Australia was shown to slow the rate of N supply by soil (N mineralization) to plant-available forms (O'Connell et al. 2003). The potential loss of C and N from soils with afforestation suggests that future plantation productivity on these soils might be less than in the initial rotations.

This study examines the effects of afforestation on soils through a formal meta-analysis of soil changes, including soil cations, acidity, carbon, and nitrogen. Our study examines data from numerous different families and genera of plantation species from globally distributed sites. Based on the results of site-specific and regional studies, we predict that afforestation will lead to more acidity and Na in soils and lower nutrient cations, carbon, and nitrogen contents. We discuss potential management tools to reduce the long-term impacts of afforestation on soil and to enable plantations to continue as productive, sustainable sinks for carbon sequestration.

Methods

Literature search and calculations

Data sources on the effects of afforestation on soil were assembled from the scientific literature through the end of 2007. We contacted investigators and searched the online databases Web of Science and Agricola for available papers (data *available online*).[5,6] We limited the search parameters to papers whose title, abstract, or keywords referred to afforestation or plantation; soil; and grass, grassland, pasture, or shrubland. Of the papers returned by the search, we selected those that had a paired-sample or chronosequence design; the final data set contained 71 papers with 153 independent sites (Tables 1 and 2).

The data set contains analyses of afforestation with many different tree species, which we grouped into the following categories: *Eucalyptus* spp., *Pinus* spp., angiosperms other than *Eucalyptus* (henceforth "other angiosperms"), and conifers other than *Pinus* (henceforth "other conifers"). We chose these groupings since *Eucalyptus* and *Pinus* were the most commonly planted genera (FAO 2006*a*). The proportion of studies in each genus or type in our analysis is similar to the global distribution of genera and types (FAO 2001*a*). For instance, ~50% of sites in this analysis are *Pinus* plantations compared to ~45% of the afforested area globally (Table 1; FAO 2001*a*, *b*). The majority of the

[5] ⟨http://isiknowledge.com⟩
[6] ⟨http://agricola.nal.usda.gov⟩

TABLE 1. Studies included in this meta-analysis.

Reference	Country	Plantation type
Adams et al. (2001)	New Zealand	other conifer
Adams et al. (2001)	New Zealand	pine
Alfredsson et al. (1998)	New Zealand	other conifer
Alfredsson et al. (1998)	New Zealand	pine
Alriksson and Olsson (1995)	Sweden	other conifer
Barton et al. (1999)	Scotland	pine
Binkley and Resh (1999)	USA	eucalyptus
Binkley et al. (1989)	USA	pine
Burton et al. (2007)	Australia	pine
Chen et al. (2007)	China	pine
Chen et al. (2000)	New Zealand	pine
Condron and Newman (1998)	New Zealand	other conifer
Condron and Newman (1998)	New Zealand	pine
Davis (1994)	New Zealand	pine
Davis (1995)	New Zealand	pine
Davis (2001)	New Zealand	pine
Davis and Lang (1991)	New Zealand	pine
Del Galdo et al. (2003)	Italy	other angiosperm
Garbin et al. (2006)	Brazil	pine
Garg and Jain (1992)	India	other angiosperm
Giddens et al. (1997)	New Zealand	pine
Gilmore and Boggess (1963)	USA	pine
Groenendijk et al. (2002)	New Zealand	pine
Guevara-Escobar et al. (2002)	New Zealand	other angiosperm
Guo et al. (2007)	Australia	pine
Hawke and O'Connor (1993)	New Zealand	pine
Hofstede et al. (2002)	Ecuador	other angiosperm
Hofstede et al. (2002)	Ecuador	pine
Huygens et al. (2005)	Chile	pine
Jain and Singh (1998)	India	other angiosperm
Jobbagy and Jackson (2003)	Argentina	eucalyptus
Jug et al. (1999)	Germany	other angiosperm
Lilienfein et al. (2000)	Brazil	pine
Lima et al. (2006)	Brazil	eucalyptus
Mao et al. (1992)	China	eucalyptus
Markewitz et al. (1998)	USA	pine
Martens et al. (2004)	USA	other angiosperm
Menyailo et al. (2002)	Russia	other angiosperm
Menyailo et al. (2002)	Russia	other conifer
Menyailo et al. (2002)	Russia	pine
Merino et al. (2004)	Spain	other angiosperm
Montagnini (2000)	Costa Rica	other angiosperm
Musto (1992)	South Africa	eucalyptus
Musto (1992)	South Africa	other angiosperm
Musto (1992)	South Africa	pine
Muys and Lust (1993)	Belgium	other angiosperm
Nielsen et al. (1999)	Denmark	other conifer
Noble et al. (1999)	Australia	other angiosperm
Noble et al. (1999)	Australia	pine
Nosetto et al. (2006)	Argentina	pine
O'Connell et al. (2003)	Australia	eucalyptus
Ohta (1990)	Phillipines	other angiosperm
Ohta (1990)	Phillipines	pine
Parfitt et al. (1997)	New Zealand	pine
Parfitt et al. (2003*b*)	New Zealand	pine
Payet et al. (2001)	South Africa	pine
Prosser et al. (1993)	Australia	eucalyptus
Quideau and Bockheim (1997)	USA	pine
Resh et al. (2002)	Puerto Rico	eucalyptus
Resh et al. (2002)	Puerto Rico	other angiosperm
Reynolds et al. (1988)	UK	other conifer
Rhoades and Binkley (1996)	USA	eucalyptus
Rhoades and Binkley (1996)	USA	other angiosperm
Richter et al. (1994)	USA	pine
Ross et al. (1999)	New Zealand	pine
Ross et al. (2002)	New Zealand	pine
Saggar et al. (2001)	New Zealand	pine
Schipper and Sparling (2000)	New Zealand	pine
Scott et al. (2006)	New Zealand	pine
Sharrow and Ismail (2004)	USA	other conifer
Singh et al. (1998)	India	eucalyptus

TABLE 1. Continued.

Reference	Country	Plantation type
Singh et al. (1998)	India	other angiosperm
Sparling et al. (2000)	New Zealand	pine
Vesterdal et al. (2002)	Denmark	other conifer
Williams et al. (1977)	UK	pine
Wu et al. (2006)	China	pine
Yeates and Saggar (1998)	New Zealand	pine
Yeates et al. (2000)	New Zealand	pine
Yuste et al. (2007)	USA	pine
Zhao et al. (2007)	China	pine
Zinn et al. (2002)	Brazil	eucalyptus
Zinn et al. (2002)	Brazil	pine

original vegetation types in our database were grasslands or pastures (73%), followed by abandoned or degraded agricultural lands (25%), whereas only three sites (2%) corresponded to shrublands with incomplete canopy closure.

Across studies, the depth of the mineral soil varied greatly from 2.5 to 100 cm. There were only six studies with data on forest floor organic horizons, which together with inconsistent definitions of organic horizons, led us to restrict our analysis to mineral soils. We further restricted our analysis to the top 30 cm of mineral soil since that depth increment contains the highest concentrations of soil organic matter and has the strongest reaction to afforestation (Jobbagy and Jackson 2000).

If a study reported C or N as a percentage of soil mass, we converted the values to metric tons (C or N) ha^{-1} by multiplying the percentage carbon or nitrogen by 100, bulk density (g soil/cm), and sampling depth (cm). Some of the studies did not report soil bulk density. Initially, we attempted to estimate bulk density from soil texture, but this, too, was rarely reported. Therefore, where needed we estimated bulk density (BD in g soil/cm^3) using Eq. 1 (Post and Kwon 2000):

$$\mathrm{BD} = \frac{100}{\dfrac{\mathrm{OM\%}}{0.244} + \dfrac{100 - \mathrm{OM\%}}{1.64}} \qquad (1)$$

where OM% is the percentage of soil organic matter, assuming that organic matter equals percentage soil carbon divided by 0.58 (Mann 1986).

For all soil variables in this meta-analysis (soil pH, cations, C, and N) there were multiple methods used by different studies. Cations were measured predominately by flame atomic absorption spectroscopy (84%) with a smaller number (16%) by inductively coupled plasma mass spectrometry. Soil pH was measured with pH electrodes in deionized water (77%), 0.01 kmol/L $CaCl_2$ (21%), and $BaCl_2$ (2%). Soil C and N were determined predominately by combustion (70%), with 28% of C analyses by Walkley-Black/dichromate digestion, 2% by loss on ignition, and 23% of N analyses by Kjeldahl digestion. To compensate for potential methodological artifacts, we used the proportion response (response ratio of afforested value/control value described below) of variables for each site.

Meta-analysis

Our goal was to determine the mean effect of afforestation on soil variables. We calculated the effect size of afforestation on a soil variable for a given site as a response ratio, $r = X^E/X^C$, where X^E is the mean value for a site of a given soil variable under afforestation, and X^C is the mean value of the same site's control (Hedges et al. 1999, Gurevitch and Hedges 2001). To match the scale of pH (logarithmic) to the linear scales of all other variables in the meta-analysis, we transformed to hydrogen ion concentration values ($10^{-\mathrm{pH units}}$) to calculate response ratios, yet we present the results of the meta-analysis in pH units for ease of interpretation. The response ratio was then transformed by the natural logarithm to make the values linear, so that an increase in a variable due to afforestation would be proportional and on the same scale as a decrease.

Ideally, the meta-analysis of response ratios should be weighted by the sample size and variances for a study.

TABLE 2. Number of studies in this meta-analysis by variable and afforestation type.

	Analyses								
Afforestation type	Na	Ca	Mg	K	pH	BS%	C	N	C:N
Eucalyptus	8	30	10	10	16	3	26	16	16
Other angiosperm	5	12	12	12	16	4	16	13	13
Pinus	33	46	46	42	68	28	71	61	61
Other conifers	6	8	8	8	9	6	7	7	7
Overall	52	96	76	72	109	41	120	97	97

Note: BS% is base saturation percentage.

TABLE 3. Percentage change due to afforestation.

	Analyses (%)			
Afforestation type	Na	Ca	Mg	K
Eucalyptus	250 (61, 674)	−37 (−25, −47)		
Other angiosperm	32 (15, 50)			
Pinus	81 (42, 136)	−31 (−17, −43)		−23 (−2.1, −42)
Other conifers		−16 (−2.4, −29)	−52 (−27, −70)	
Overall	71 (35, 120)	−29 (−20, −37)		

Notes: BS% is base saturation percentage. Values reported are mean percentage gain (positive values) or loss (negative values) generated by bootstrapping, with 95% confidence interval in parentheses.

However, for some studies we were unable to determine independent sample sizes and variances, and the data were frequently not normally distributed. To compensate for small sample sizes, variance, nonnormality, and to include as many studies as possible, we used a nonparametric approach to statistical analyses, i.e., an unweighted meta-analysis (Gurevitch and Hedges 2001, Guo and Gifford 2002). Mean effect size (log response ratio) and 95% confidence intervals were generated by bootstrapping (10 000 iterations) in SAS (SAS Institute, Cary, North Carolina, USA; Efron and Tibshirani 1993). A mean effect size was significantly different from 0 if its 95% confidence interval did not overlap 0 (Gurevitch and Hedges 2001).

For ease of interpretation, we present the means of the variables for control and afforested rather than response ratios in some figures. This presentation is intended to provide a reasonable range of values on an absolute scale. However, the means alone of all sites in this study can mask the underlying effect size of afforestation due to variability in initial control values. To show the magnitude of the effect of afforestation while controlling for differences in initial control values, we also present the mean response ratios (transformed to percentage change due to afforestation) with 95% confidence intervals in square brackets. These percentages represent the mean percentage change for a given grass- or shrubland that has been afforested.

We also tested for correlations among the response ratios of the measured variables. Since several distributions were not Gaussian, we used the nonparametric Spearman's rank correlation coefficient. Correlation coefficients and tests of statistical significance were calculated using PROC CORR in SAS.

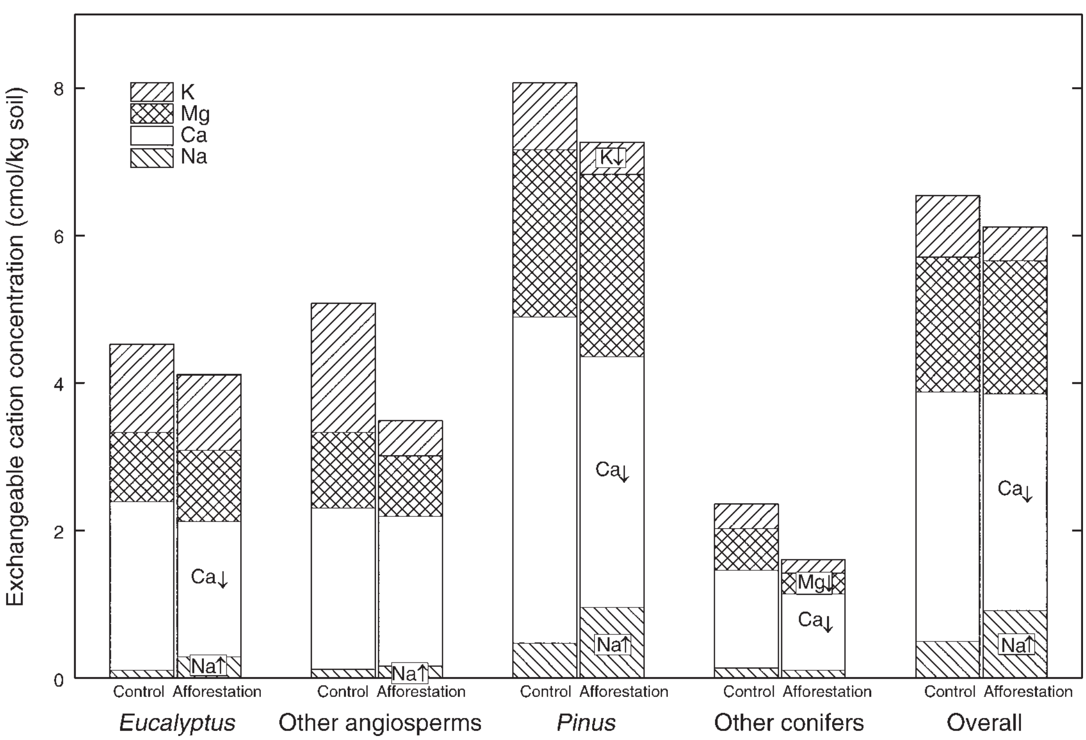

FIG. 1. Changes in soil exchangeable cations with afforestation. Significant ($P < 0.05$) increases of a cation due to afforestation within a plantation type are indicated by the cation name and an up arrow; decreases are indicated by a down arrow. See Table 2 for the number of studies used for each analysis, and Table 3 for means and 95% confidence intervals.

TABLE 3. Extended.

Analyses (%)				
pH	BS%	C	N	C:N
−13 (−7.9, −18)				5.7 (0.05, 11.3)
−5.9 (−4.3, −7.5)	−21 (−2.9, −37)	−15 (−8.6, −21)	−20 (−12, −27)	11.6 (3.5, 20)
−6.5 (−1.2, −12)	−10 (−4.5, −17)			5.9 (0.18, 11)
−5.7 (−4.3 −7.5)	−17 (−2.3, −17)		−15 (−8.6, −21)	9.9 (4.2, 16)

RESULTS

Exchangeable cations

Across diverse plantation types, we observed decreases in nutrient cations (Ca, K, Mg), increases in sodium (Na), or both concurrently with afforestation. Afforestation by *Eucalyptus*, *Pinus*, other conifers, and all vegetation types combined decreased soil Ca relative to controls by 37%, 31%, 16%, and 29%, respectively (each analysis $P < 0.05$; Fig. 1, Table 3). Afforestation with other conifers decreased Mg concentration by 52% ($P < 0.05$; Fig. 1, Table 3). However, there was no significant effect of afforestation on soil Mg attributable to afforestation with *Eucalyptus*, *Pinus*, or other angiosperms. Afforestation with *Pinus* led to 23% lower concentrations of K (Fig. 1, Table 3). Afforestation with *Eucalyptus,* other angiosperms, and *Pinus* raised soil Na relative to controls by 250%, 32%, and 81%, respectively (Fig. 1, Table 3). Afforestation by other conifer genera did not induce a significant change in soil Na levels. If all afforestation types were combined, then soil Na concentration increased 71% relative to controls.

Soil pH and base saturation

Exchangeable cation concentrations and soil pH are closely linked, and plantations also typically increased acidity and lowered exchangeable base cation saturation. Afforestation with *Pinus*, other conifers, and all vegetation combined reduced base saturation by 21%, 10%, and 17%, respectively ($P < 0.05$ for each; Fig. 2, Table 3). There was no effect of *Eucalyptus* or other angiosperms on base saturation (Fig. 2). Afforestation with *Eucalyptus* acidified soils vs. controls from pH 6.0 to 5.3 (Fig. 2, Table 3). *Pinus* plantations led to a moderate acidification from pH 5.7 to 5.4; other conifer plantations acidified soil from pH 4.6 to 4.4, and across all plantation types from 5.6 to 5.3 ($P < 0.05$; Fig. 2, Table 3).

Across all plantation types, there was a negative correlation (Spearman's $\rho = -0.58$, $P = 0.006$) between the response ratios for hydrogen ion concentration (soil pH) and base saturation (Fig. 3). We also found a negative correlation (Spearman's $\rho = -0.56$, $P < 0.0001$) between response ratios for hydrogen ion concentration and calcium (Fig. 3). Greater decreases in base saturation and calcium due to afforestation therefore correlate with greater decreases in pH.

Carbon and nitrogen

Overall, the effects of afforestation on soil organic C and N were the greatest for *Pinus* plantations. Afforestation with *Pinus* decreased soil C stocks (g/m^2) by 15% on average (Fig. 4, Table 3). However, there was no significant change in soil C with afforestation for

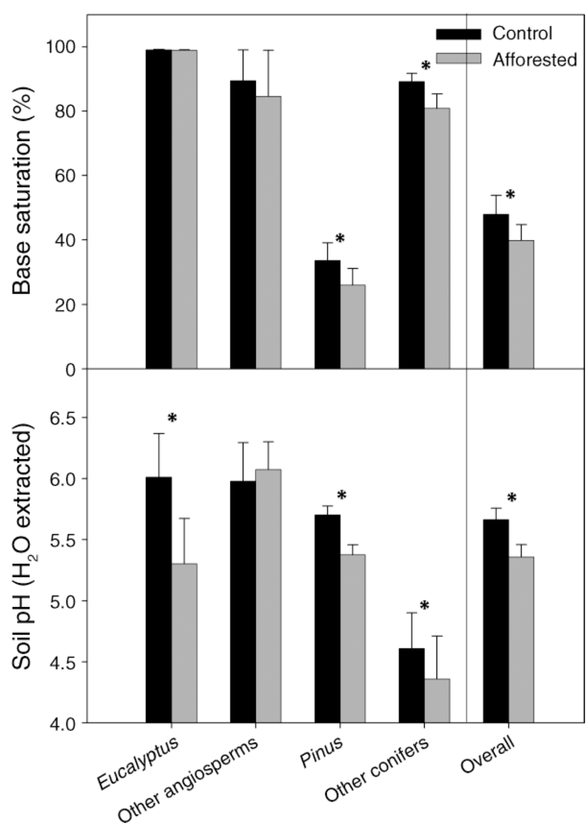

FIG. 2. Changes in base saturation and soil pH with afforestation. An asterisk indicates a significant difference ($P < 0.05$) between control and afforested for a given plantation type, and error bars represent standard error. Significance for pH was calculated based on $[H^+]$ but is presented here in pH units for ease of interpretation. See Table 2 for the number of studies used for each analysis, and Table 3 for means and 95% confidence intervals.

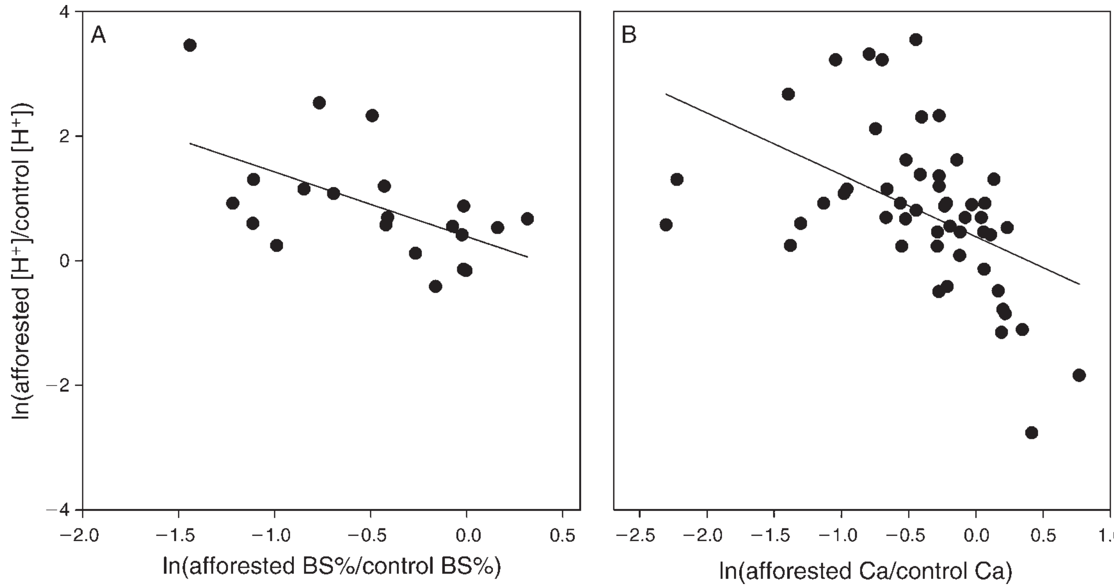

FIG. 3. Relationship between [H^+], base saturation (BS%), and Ca. In panel A, the response ratio of BS% is negatively correlated with the response ratio of [H^+] (Spearman's $\rho = -0.58$, $P = 0.006$, $N = 21$); 15 points are from *Pinus*, one from *Eucalyptus*, four from other conifers, and one from "other angiosperms." In panel B, the response ratio of Ca is negatively correlated with the response ratio of [H^+] (Spearman's $\rho = -0.56$, $P < 0.0001$, $N = 51$); 28 points are *Pinus*, 10 from *Eucalyptus*, four from other conifers, and nine others from other angiosperms. Soils were analyzed up to 30 cm deep with a mean of 14 cm.

Eucalyptus, other conifers, or other angiosperms. Similarly, soil N decreased with afforestation in *Pinus* plantations and overall by 20% and 15%, respectively, but overall changes were driven exclusively by changes for *Pinus*. There was no significant change in soil N due to *Eucalyptus*, other angiosperms, or other conifers (Fig. 4, Table 3). Soil C:N increased significantly by 5.7%, 11.6%, 5.9%, and 9.9% with afforestation by *Eucalyptus*, *Pinus*, other conifers, and all types combined, respectively (Fig. 4, Table 3).

Afforestation effects across regions with high rates of afforestation

The consistent direction and magnitude of effects across many genera and regions suggest that these effects are fairly general and may provide a reasonable estimate for global effects of afforestation on soil nutrients. Based on this study and on United Nations Food and Agriculture Organization (FAO) estimates of plantation area in regions with high rates of afforestation, we calculated the total amount of nutrients gained or lost from soils globally (Table 4). Given that much of the lost nutrient stock is likely stored in biomass and litter, these numbers represent large potential exports of harvested nutrients (and additions of Na). The largest losses of C and N from soils were in North and Central America with considerable losses also in China and Europe (Table 4). China and North and Central America lost the most Ca and China gained the most Na (Table 4).

DISCUSSION

Our afforestation analysis revealed consistent effects on soil properties across a broad range of locations and tree genera. Depletion of exchangeable cations was observed in three of four plantation types. Increases in soil Na were also found across three plantation types (Fig. 1). Consistent with the findings of Jackson et al. (2005), *Eucalyptus* and *Pinus* plantations significantly acidified soils (Fig. 2); we also found that other conifers acidified soils (Fig. 2). Soil C and N levels decreased, but only for *Pinus* plantations; however, afforestation with either *Eucalyptus* or *Pinus* significantly raised the soil C:N (Fig. 4). The fact that most of the significant differences were due to *Eucalyptus* or *Pinus* afforestation could be due either to the great availability of studies for those genera (and hence a greater power to detect differences) or because *Eucalyptus* or *Pinus* plantations are often not native to the region in which they are planted. The higher growth rates of these exotic plantations could lead to more drastic changes in soils than plantation using a native species of tree.

Exchangeable cations and sodium

Several potential mechanisms may explain the differences in exchangeable cations observed with afforestation (Fig. 1): uptake outpacing rates of supply, increased leaching to groundwater, or decreases in mineral weathering. However, in the Argentine pampas, afforestation of grasslands with *Eucalyptus camaldulensis* was found to decrease mineral soil cations by redistribution from soil to biomass pools (Jobbagy and Jackson 2003). This redistribution of cations by *Eucalyptus* is attributable to increased cation uptake of plantations compared with the native grasses (Jobbagy and Jackson 2003). This mechanism likely explains our study's finding of the depletion of Ca, K, and Mg from soils across many different plantation genera (Fig. 1). Additionally, Jobbágy and Jackson (2003) found decreased cation exchange capacity (CEC) with losses in cations; the lower CEC could indicate a reduced capacity of soils to store cations, which suggests that new inputs of cations (gypsum or lime fertilizer) might not always increase Ca stocks to pre-afforestation levels.

If we estimate the change in stocks of exchangeable soil cations (kg/ha) from the data in Fig. 1 and the mean bulk density and sampling depth from the data set, then the mean loss of Ca from soils due to afforestation with *Eucalyptus*, *Pinus*, and other conifers is 0.53, 1.25, and 0.34 Mg/ha, respectively. These soil losses are within the range of published values for total biomass Ca; for example, several *Eucalyptus* plantations in Australia had an estimated total biomass Ca of 0.32 Mg/ha and a *Pinus radiata* plantation in Spain was estimated at 0.33 Mg Ca/ha (Turner and Lambert 1986, Ouro et al. 2001). Additionally, the mean amount of soil magnesium lost due to afforestation with other conifers in this study was 0.27 Mg/ha, compared to an estimate of *Pinus radiata* magnesium stocks of 0.68 Mg/ha (Ouro et al. 2001). The similarity in soil losses to total biomass content of Ca and Mg supports the hypothesis that uptake by plantations is a major driver of soil Ca and Mg loss (Richter et al. 1994) and that plantation management leaving as much residue in place as possible will minimize problems of soil fertility in the future.

The observed losses of exchangeable cations from mineral soils could decrease productivity of successive plantation rotations. Atmospheric inputs of Ca, K, and Mg are usually less than plant uptake, which is typically supplied through mineral weathering, mineralization, and leaching from plant biomass (Schlesinger 1997). Best practices of retaining logging residues and debarking harvested plantations on site could substantially reduce cation losses from afforestation. Residual parts of harvested trees with little commercial value (leaves, branches, and bark) contain the majority of Ca and Mg in forest biomass. Typically these residues are removed from the site or burned, leading to export or losses of cations through accelerated leaching (FAO 2002). Aboveground biomass in bark, leaves, twigs, and reproductive structures at Coweeta LTER contained 86% and 63% of the total biomass Ca and Mg (Day and Monk 1977). Retaining these residual components without mounding or burning (reducing leaching and erosion losses) could lead to lower long-term losses of soil Ca and Mg (Mendham et al. 2003*a*).

The observed increase in soil Na was likely caused by afforestation's effect on hydrology (Fig. 1). Jobbágy and Jackson posited that this increased Na due to *Eucalyptus*

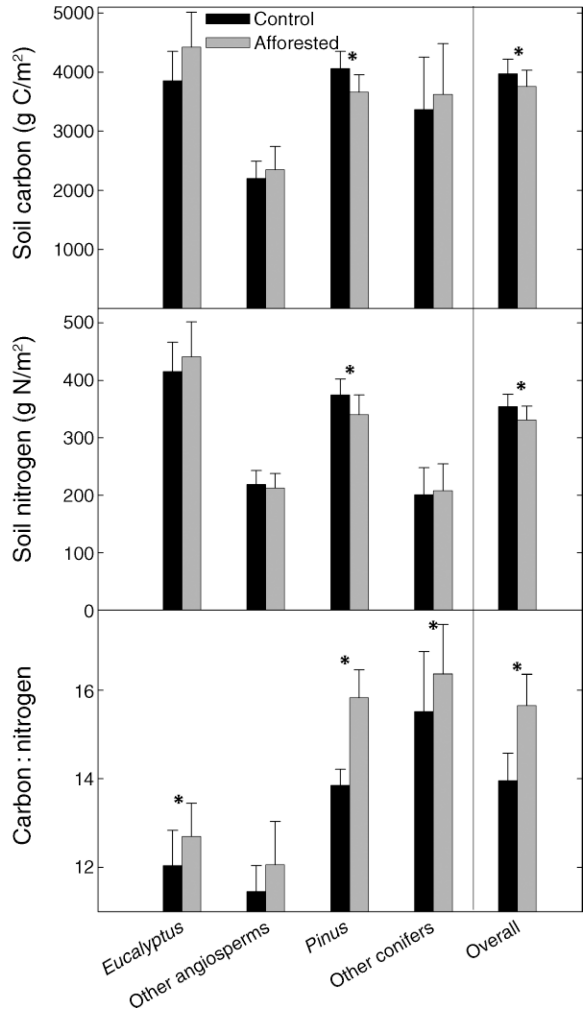

FIG. 4. Changes in soil carbon and nitrogen. An asterisk indicates a significant difference ($P < 0.05$) between control and afforested land for a given plantation type, and error bars represent standard error. See Table 2 for the number of studies used for each analysis, and Table 3 for means and 95% confidence intervals.

afforestation could be caused by three mechanisms: enhanced capillary rise of water through soil due to drier soil under plantations, decreased leaching to deep groundwater, or water uptake by roots from deeper soil depths (Jobbagy and Jackson 2004*a*, *b*). Because sodium is not essential to plant biochemistry, plants exclude it while taking up water and other cations. (Marschner 1995, Schlesinger 1997, Jobbagy and Jackson 2004*b*). Jobbágy and Jackson (2004*b*) demonstrated that increased water uptake by *Eucalyptus* plantations with sodium exclusion led to soil and groundwater salinization.

Soil pH, base saturation, and the soil exchange complex

Comparing soils from a similar climate, forest soils tend to be more acidic than grassland soils (Schlesinger

TABLE 4. Mean total significant losses or additions of nutrients with afforestation in surface soils across different regions of the world.

Region or country, afforestation type	Area (10^3 ha)	Soil C lost (10^3 Mg)	Soil N lost (10^3 Mg)	Soil Ca lost (10^3 Mg)	Soil Mg lost (10^3 Mg)	Soil Na added (10^3 Mg)
South Africa						
Eucalyptus	566			302		354
Pinus	724	2957	259	904		120
Other broadleaf	123					2
China						
Eucalyptus	2397			1277		1498
Pinus	10 031	40 971	3582	12 521		1665
Other conifer	16 160			5544	2392	
Other broadleaf	13 304					186
South and South East Asia						
Eucalyptus	4047			2156		2529
Pinus	1734	1115	619	2164		288
Other conifer	273			94		
Other broadleaf	11 104					155
Europe						
Pinus	10 945	44 704	3908	13 661		1817
Other conifer	9077			3114	1343	
Other broadleaf	3730					52
North and Central America						
Eucalyptus	198			105		124
Pinus	15 440	63 063	5513	19 272		2563
Other conifer	88			30	13	
Other broadleaf	511					7
Australia and New Zealand						
Eucalyptus	549			292		343
Pinus	2602	10 628	929	3248		432
Other conifer	163			56	24	
Brazil, Argentina, and Chile						
Eucalyptus	3777			2012		2360
Pinus	4253	17 371	1519	5309		706
Other conifer	104			36	15	
Other broadleaf	585					8

Notes: Estimated total area (ha) of afforested area for regions is based on UN FAO data from voluntary country reports; empty cells indicate that not all countries in every region are represented (FAO 2006*a*, *b*). The estimates of area are conservative; only deliberately planted forests designated for harvest were counted. Losses or additions of C and nutrients were calculated from differences in means from Figs. 1 and 4. Total C and N were measured up to 30 cm of mineral soil. Total stocks for Ca, Mg, and Na were estimated using the mean bulk density of soils (1.068 g/cm^3) and mean sampling depth (0.14 m) in our database.

1997, Chapin et al. 2002). This difference in acidity can be generated through several mechanisms, including increased production of organic acids or generation of carbonic acid from higher rates of autotrophic respiration (Richter and Markewitz 1995). The increased acidity of forests may also be caused by increased uptake of cations by trees and consequent changes in the proportions of cations adsorbed to the soil exchange complex (Jobbagy and Jackson 2003, 2004*b*). The consistent effects on cations by afforestation in our analysis suggest that changes in the proportions of cations could be a major driver behind the higher acidity of forest soils (Figs. 1 and 2). For example, afforestation decreased Ca, Mg, and K in many different plantation types, and concurrently increased concentrations of Na and H^+ (Figs. 1 and 3). Additionally, the correlation between decreased base saturation and Ca with increased H^+ suggests that the exchangeable cations (Ca, Mg, K) taken up by afforested plantations tend to be replaced on soil exchange sites by H^+. The consequence of this change is a soil exchangeable pool with a higher proportion of H and Na ions. The consistency of these effects in this study across broad geographic regions and differing tree plantation types suggests that relocation of cations could be a general mechanism driving the acidity of forests across many different ecosystems.

Although acidification was significant for *Eucalyptus*, *Pinus*, and other conifer plantations, the pH of the control soils (grassland or shrubland) also varied (Fig. 2). This result suggests that the mechanism of acidification across plantation species is likely similar, but the actual impact of the change in pH depends on the conditions of the control site. The relationship between soil pH and soil fertility (e.g., cations) is not linear because of the logarithmic scale of pH; for example, soils with pH between ~5 and 8 have consistently high percentages of Ca in their exchangeable cation pool, but

PLATE 1. Grazing lands near Minas, Uruguay, that have recently been planted with *Eucalyptus*. This region is largely grasslands and has no historical record of significant forests prior to European settlement. These plantations grow rapidly and are usually harvested within 7–10 years of planting. Photo credit: S. T. Berthrong.

below pH 5 the Ca percentage drops precipitously (Brady and Weil 2002). In our study, afforestation with *Pinus* and other conifers lowered pH from 5.7 to 5.3 and 4.6 to 4.3. Although the changes are similar in number of pH units, the acidification in *Pinus* plantations probably has more implications for soil fertility, since the control soils for other conifers were already acidic (>10 times higher proton concentration) and had less exchangeable bases to lose. For example, see the calcium losses in this study (Fig. 1).

Soil carbon and nitrogen

We found a significant decrease in soil organic C and N with *Pinus* afforestation, but not with other species. This result agrees with the conclusions of Guo and Gifford (2002) who found afforestation by pines (but not broadleaf species) significantly reduced soil C. Since soil C and N are indices of soil fertility, the losses of C and N from soil under *Pinus* plantations may indicate a general loss in soil fertility (Brady and Weil 2002). However, unlike Guo and Gifford (2002), we did not find correlations between plantation age or depth of sampling and the log response ratio. Given that most of the sites were in their first rotation, observed soil responses to afforestation may not yet have come to equilibrium. Also, since afforested plantations are repeatedly harvested, they might not reach equilibrium in the same sense as a natural ecosystem recovering from disturbance.

The loss of soil C under a plantation with higher primary productivity seems counterintuitive; however, this loss could be due to differences in the distribution and decomposability of plantation biomass (Guo and Gifford 2002). Plantation tree roots are longer-lived and coarser than typical grass roots, and contribute less to soil organic material (Post and Kwon 2000, Guo and Gifford 2002). Additionally, plantations deposit more C as litter to the forest floor, but there was insufficient data available to evaluate how much C globally was stored as afforested forest floor material (Jobbagy and Jackson 2000, Post and Kwon 2000, Guo and Gifford 2002). A study in Australia found that debris from a *Pinus radiata* plantation stored a large amount of C, but this only offsets 22% of the carbon lost from the mineral soils due to afforestation (Guo et al. 2006). Additionally, C in plantation forest floor material incorporates more slowly into soil organic matter than in native grass systems (Guo et al. 2006).

Carbon loss from soils as a result of *Pinus* afforestation influences potential rates of C sequestration. The mean loss of soil C under pine plantations in this analysis was 4.1 Mg C/ha; an average plantation with a 20-year rotation time can be assumed to contain ~75 Mg C/ha on average (Vitousek 1991). The loss of soil C due to afforestation is therefore a modest 5.5% of C sequestered in vegetation. Harvesting of plantations usually results in additional losses of C from soils from increased rates of decomposition (Vitousek 1991). Though these losses of C from afforested soils are less than the sequestration potential in biomass, they are large enough to be considered in C budgets of these systems.

A potential method of reducing the impact of soil C and N loss is to retain logging residues on site. As mentioned in *Discussion: Exchangeable cations and sodium*, logging residues are usually removed or burned before subsequent rotations are planted, which leads to a large loss of C and N (FAO 2002). Removing or burning residues from harvested plantations also decreases soil C and N contents; in Australia, for instance, burning logging residues led to a loss of 200–350 kg N/ha (Merino and Edeso 1999, Mendham et al. 2003*b*). Conversely, retention of logging residues led to higher soil organic matter and N contents and higher rates of net N mineralization (Goncalves et al. 2000, FAO 2002). Retention also led to increased productivity compared to burned sites in subsequent plantation rotations (Bouillet et al. 2000, Fan et al. 2000, Xu et al. 2000).

Soil C:N increased with both *Pinus* and *Eucalyptus* afforestation (Fig. 4). Though the changes in C:N with *Eucalyptus* were not as large as for *Pinus*, the increase in C:N is potentially an indicator of lower soil organic matter quality (Brady and Weil 2002). Since *Pinus* plantations decreased both soil C and N contents, the increased C:N in *Pinus* plantations suggests that the depletion of N is more rapid (Fig. 4). This more rapid decrease in N is likely due to increased plant uptake of N compared to native grasslands (Jobbagy and Jackson 2004*a*). Another possible implication of increased C:N ratios in these systems is increased microbial N immobilization (Brady and Weil 2002, Berthrong and Finzi 2006). Microbes immobilize more N in their biomass as C:N increases; as a consequence, mineralization rates are lower, which leads to lower plant-available nitrogen and lower productivity.

Implications and conclusions

This global study indicates that afforestation often leads to more acidic and nutrient-deficient mineral soils, but best management practices can help overcome some of these changes. Although these soil changes could impair the productivity of successive rotations, it is unclear how long it will take to see noticeable productivity declines. Turner and Lambert (1986) estimated that in Australia it would take ~320 years (four rotations) before nutrient depletion (P and Ca) would impair productivity; however, this was estimated by total biomass nutrient stocks and total soil pools, and not from actual measures of productivity over successive plantings (Turner and Lambert 1986). Estimates of Chinese fir plantation yields show that annual biomass production by the third rotation drops by more than 50% (Zhang et al. 2004). In addition to nutrient depletion, plantation harvesting has been shown to compact soil, which reduced regeneration of new seedlings by up to 51.5% (Balbuena et al. 2002).

With afforestation likely to continue as a useful mechanism for offsetting carbon dioxide emissions, management practices should be implemented to reduce soil impacts and improve sustainability. For instance, maintaining soil fertility could be accomplished through site and harvest management tools. Retention of logging slash, on site debarking and retention, and reduced burning of slash have been shown to reduce nutrient (Ca, Mg, N, and others) export, loss of soil organic material, erosion losses, and soil compaction (Merino and Edeso 1999, Bouillet et al. 2000, Ouro et al. 2001, Mendham et al. 2003*b*, Merino et al. 2004). Depending on initial site conditions, combinations of these different management conservation practices could improve sustainability.

With demand rising for timber products, managers need either to harvest stocks from remaining native forests or to increase sustainable plantation forestry (ABARE and Jaako-Poyry 1999). Reducing soil degradation from afforestation and harvesting is important for productivity and reducing habitat loss. If managed sustainably, afforestation could simultaneously preserve remaining native forests and function as a long-term CO_2 sink.

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