Nitrogen Addition Shapes Soil Phosphorus Availability in Two Reforested Tropical Forests in Southern China

Xiankai Lu1, Jiangming Mo1,4, Frank S. Gilliam2, Hua Fang3, Feifei Zhu1, Yunting Fang1, Wei Zhang1, and Juan Huang1

1 Key Laboratory of Vegetation Restoration and Management of Degraded Ecosystems, South China Botanical Garden, Chinese Academy of Sciences, Guangzhou, 510650, China
2 Department of Biological Sciences, Marshall University, Huntington, West Virginia, 25755-2510, U.S.A.
3 Guangzhou Institute of Geography, Guangzhou, Guangdong, 510070, China

ABSTRACT

Scant information is available on how soil phosphorus (P) availability responds to atmospheric nitrogen (N) deposition, especially in the tropical zones. This study examined the effect of N addition on soil P availability, and compared this effect between forest sites of contrasting land-use history. Effects of N addition on soil properties, litterfall production, P release from decomposing litter, and soil P availability were studied in a disturbed (reforested pine forest with previous understory vegetation and litter harvesting) and a rehabilitated (reforested mixed pine/broadleaf forest with no understory vegetation and litter harvesting) tropical forest in southern China. Experimental N-treatments (above ambient) were the following: Control (no N addition), N50 (50 kg N ha⁻¹ yr⁻¹), and N100 (100 kg N ha⁻¹ yr⁻¹). Results indicated that N addition significantly decreased soil P availability in the disturbed forest. In the rehabilitated forest, however, soil P availability was significantly increased by N addition. Decreases in soil P availability may be correlated with decreases in rates of P release from decomposing litter in the N-treated plots, whereas the increase in soil P availability was correlated with an increase in litterfall production. Our results suggest that response of soil P availability to N deposition in the reforested tropical forests in southern China may vary greatly with temporal changes in tree species composition and soil nutrient status, caused by different land-use practices.

Abstract in Chinese is available in the online version of this article.

Key words: land-use history; litter decomposition; nitrogen addition; nitrogen deposition; phosphorus availability; phosphorus release; reforested forest; tropical forests.

Received 11 January 2011; revision accepted 18 June 2011.

© 2011 The Author(s)

This article was published online on 31 October 2011. An error was subsequently identified. This notice is included in the online and print versions to indicate that both have been corrected on 14 November 2011.
In tropical soils, available P mainly originates from weathering of parent material and biologic processes. Once weathering ceases to provide significant inputs of P, vegetation largely depends on the recycling of P through biologic processes, such as litter decomposition (Walker & Syers 1976, Tiessen et al. 1994, McGroddy et al. 2004b). The addition of exogenous N to the soil-litter subsystem has shown variable effects on the rate of litter decomposition in forest ecosystems (Vitousek 1998, Knorr et al. 2005, Mo et al. 2006), and thus, potentially, the availability of phosphorus.

Rates of N deposition to Chinese tropical and subtropical forests have increased in recent years, largely due to increased agriculture and industrialization. Liu et al. (2006) reported that emissions of reactive N increased from 1980 (7 Tg N yr$^{-1}$) to 2005 (20 Tg N yr$^{-1}$) in China. Lü & Tian (2007) concluded that total N deposition rates are highest (65 kg N ha$^{-1}$ yr$^{-1}$) in south-central China, with a mean rate of 19 kg N ha$^{-1}$ yr$^{-1}$, which is higher than that for most of North America and Europe, where N-mediated threats to forest ecosystem health have been suggested (Bobbink et al. 2010; Percy & Ferretti 2004).

Many primary forests in China have been deforested during the past several centuries (Wang et al. 1982, Liu et al. 2000, Li 2004), with only two percent of the nation’s total forest resources remaining intact (Liu et al. 2000). Attempts to reverse land degradation have been made in many subtropical and tropical regions of China, with extensive areas having been reforested with native pine species (e.g., *Pinus massoniana* Lamb) to prevent further degradation of the landscape. Although cutting of the trees is usually prohibited, harvesting of understory vegetation and litter is often allowed to satisfy fuel needs of local people (Brown et al. 1995, Mo et al. 1995, 2003). These reforested stands are often referred to as disturbed forests (having experienced understory vegetation and litter removal) and rehabilitated forests (reforested without such removal) (Mo et al. 2003). Reforested areas cover more than half of the total forested regions of subtropical and tropical China (Brown et al. 1995, Mo et al. 2003, SFA 2007). Despite this, the interactive effects of N deposition and land-use changes on element cycling are poorly known (Mo et al. 2003, 2007).

The objective of our present study was to examine the effects of N addition on soil P availability and to compare these effects between reforested sites of different land-use history. We have previously reported that N addition significantly suppressed release rate of P from decomposing litter in the disturbed and the rehabilitated forests in southern China during the first 24 mo (July 2003 to June 2005) of a N fertilization experiment (Mo et al. 2007), suggesting that soil P availability may decrease with increased N deposition. Accordingly, we hypothesize that soil P availability in both disturbed and rehabilitated forests would be reduced by continued N addition.

**METHODS**

**SITE DESCRIPTION.**—This study was conducted in the Dinghushan Biosphere Reserve, which is central Guangdong Province in southern China (112°10’E, 23°10’N), and occupies an area of approximately 1200 ha. Three forest types occur in the reserve: monsoon evergreen broadleaf forest (mature forest), mixed pine/broadleaf forest (rehabilitated forest), and pine forest (disturbed forest). The reserve has a monsoon climate, and is located in a tropical moist forest life zone (sensu Holdridge 1967). The mean annual rainfall of 1927 mm has a distinct seasonal pattern, with 75 percent falling from March to August, and only 6 percent from December to February (Huang & Fan 1982). Annual average relative humidity is 80 percent. Mean annual temperature is 21.0°C, with an average temperature of the coldest month (January) and hottest (July) of 12.6°C and 28.0°C, respectively (Huang & Fan 1982). This region has been experiencing high ambient N deposition (21–38 kg N ha$^{-1}$ yr$^{-1}$ as inorganic N in bulk precipitation), at least since 1990s (Huang et al. 1994, Zhou & Yan 2001, Mo et al. 2002, Fang et al. 2008). Fang et al. (2008) estimated wet deposition N as 34 and 32 kg N ha$^{-1}$ yr$^{-1}$ in 2004 and 2005, respectively.

We selected two types of forest for this study: a rehabilitated forest and a disturbed forest (Mo et al. 2006, Lu et al. 2011). The two forest types are about 4 km from each other, both originating from pine planting following clear-cutting in the 1930s, with sites being heavily eroded and degraded (Wang et al. 1982, Mo et al. 1995, 2003). In addition, the disturbed forest was under continuous land-use pressure (particularly harvesting of understory vegetation and litter) from 1930 to 1998. As a result, the tree layer remained dominated by *P. massoniana* (Brown et al. 1995, Mo et al. 1995, 2003). In contrast, lack of such harvesting in the rehabilitated forest has allowed colonization from natural dispersal of regional broadleaf species (mainly *Schima superba* Chard. & Champ.), and has changed overstory composition. As a result, the tree layer has become dominated by *P. massoniana* and some broadleaf species (Mo et al. 2003, 2007).

We established our research sites in both forests in 2002. A pretreatment assessment conducted in June 2003 revealed that the major tree canopy species of the rehabilitated forest were *P. massoniana* and *S. superba*; disturbed forest was dominated by *P. massoniana* (Table 1). The soils in both types of forest are oxisols with variable depths (Brown et al. 1995, Mo et al. 2003).

**EXPERIMENTAL TREATMENTS.**—Nitrogen addition experiments were initiated in both forest types in July 2003. Three replicates of each of three N addition treatments were established in both rehabilitated and disturbed forests: Control (without N added), N50 (50 kg N ha$^{-1}$ yr$^{-1}$), and N100 (100 kg N ha$^{-1}$ yr$^{-1}$). A total of 18, 20-m × 20-m plots were established, nine in each of rehabilitated and disturbed forests, with each plot surrounded by a > 10-m wide buffer strip. Field plots and treatments were laid out randomly. Ammonium nitrate (NH$_4$NO$_3$) solution was sprayed monthly by hand onto the forest floor as 12 equal applications. In each plot, fertilizer was weighed, mixed with 20 L of water, and applied below the canopy using a backpack sprayer. Two passes were made across each plot to ensure an even distribution of fertilizer. The Control plots received 20 L water with no N added.

**FIELD SAMPLING.**—In both disturbed and rehabilitated forests, three litterfall traps (0.5 m × 0.5 m) with a mesh size of 1 mm...
TABLE 1. Indices of the tree layer in a disturbed and a rehabilitated tropical forest at Dinghushan Biosphere Reserve in southern China. The survey was conducted in 2003, and the area was 1800 m² for each forest. All trees were recorded when the dbh (diameter at breast height) ≥ 2.5 cm.

<table>
<thead>
<tr>
<th>Species</th>
<th>Density (stems/ha)</th>
<th>Mean (m) height</th>
<th>Mean (cm) dbh</th>
<th>Basal area (m²/ha)</th>
<th>Relative area (%) basal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disturbed forest</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pinus massoniana</td>
<td>354</td>
<td>7.4</td>
<td>17.7</td>
<td>10.5</td>
<td>95.3</td>
</tr>
<tr>
<td>Schima superba</td>
<td>38</td>
<td>4.5</td>
<td>4.0</td>
<td>0.1</td>
<td>0.4</td>
</tr>
<tr>
<td>Other plants</td>
<td>188</td>
<td>3.9</td>
<td>3.9</td>
<td>0.5</td>
<td>4.2</td>
</tr>
<tr>
<td>Total</td>
<td>580</td>
<td></td>
<td></td>
<td>11.1</td>
<td>100.0</td>
</tr>
<tr>
<td>Rehabilitated forest</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Schima superba</td>
<td>1158</td>
<td>4.6</td>
<td>6.4</td>
<td>5.5</td>
<td>54.3</td>
</tr>
<tr>
<td>Pinus massoniana</td>
<td>96</td>
<td>8.0</td>
<td>21.9</td>
<td>4.1</td>
<td>40.4</td>
</tr>
<tr>
<td>Other plants</td>
<td>150</td>
<td>3.9</td>
<td>4.4</td>
<td>0.5</td>
<td>5.3</td>
</tr>
<tr>
<td>Total</td>
<td>1404</td>
<td></td>
<td></td>
<td>10.1</td>
<td>100.0</td>
</tr>
</tbody>
</table>

Note: Data are cited from Lu et al. (2011). [This table was corrected after publication.]

were placed randomly in each plot (nine litter traps per treatment) about 0.5 m above the ground surface at the start of the study (27 August 2003). These traps were emptied once every month during the entire study period from September 2003 to August 2007.

To quantify responses of litterfall decomposition rates to N addition in both forests, we selected identical litter for all treatments: S. superba and Castanopsis chinensis (Sprengel) Hance, which are the main native tree species and contribute most of the leaf litter in forest ecosystems of Dinghushan Biosphere Reserve (Mo et al. 2006, Fang et al. 2007). Fresh leaf litter was collected using litter traps and nylon mesh, which were laid on the forest floor under the trees in the study sites during May and June 2005. All litter materials were air-dried to a constant weight and mixed before filling the mesh bags. Litter bags were made of 25 × 25 cm polyvinyl plastic with 0.5 mm mesh size at the bottom and 2 mm mesh size at the top (Fang et al. 2007). Six subsamples (about 12.0 g per subsample) from each kind of litter type were analyzed for initial litter chemistry (total C, N, and P concentrations). There was no significant difference between litter types in initial litter chemistry (Table 2).

<table>
<thead>
<tr>
<th>Species/litter type</th>
<th>C (mg g⁻¹)</th>
<th>N (mg g⁻¹)</th>
<th>P (mg g⁻¹)</th>
<th>C/N</th>
<th>C/P</th>
<th>N/P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Castanopsis chinensis</td>
<td>434(7) a</td>
<td>15.1(0.1) a</td>
<td>0.79(0.01) a</td>
<td>28.9(0.3)</td>
<td>551(10) a</td>
<td>19.1(0.3) a</td>
</tr>
<tr>
<td>Schima superba</td>
<td>445(8) a</td>
<td>14.8(0.3) a</td>
<td>0.56(0.01) a</td>
<td>30.2(0.9)</td>
<td>802(20) a</td>
<td>26.6(0.8) a</td>
</tr>
</tbody>
</table>

Note: Values are means with 1 SE in parentheses, N = 6; Within a single column in a given species sharing the same letter are not significant different at P < 0.05 level. The element ratios are given by mass:mass.
time t (October 2005 to September 2006). Litter P content was calculated by multiplying P concentration by mass remaining (Fang et al. 2007, Mo et al. 2007). The exponential model \( N/0 = e^{-kt} \) was fit to the data using least squares regression of the natural logarithm of mean litter P fraction remaining (Kuperman 1999, Mo et al. 2006), where ‘c’ is the base of natural logarithm, and ‘k’ is the litter P decomposition coefficient. We used independent sample t-tests to determine differences in soil P availability between the control plots of two forests, and differences in initial litter chemistry between the selected tree species. In addition, we conducted the planned contrast analysis to test differences between control plots and N-treatment plots. All statistical analyses were conducted using SPSS 13.0 (SPPS, Chicago, Ill, Illinois, U.S.A.) for Windows.

The statistical design of this study includes simple pseudoreplication, a common characteristic whole stand-level forest ecosystem studies, with each forest representing an experimental condition with a sample size of one (Hurlbert 1984); thus, our data should be interpreted with that in mind. It is our contention, however, that any effects we report are related to site history, rather than pre-existing differences among stands, given the close similarities among watersheds in several ‘site’ variables such as soil texture, soil organic matter, and extractable nutrients (Lu et al. 2011).

RESULTS

SOIL RESPONSE TO N ADDITION.—Available soil P of the Control plots was significantly higher in the disturbed forest \((0.43 \pm 0.04 \text{ mg kg}^{-1} \text{ soil})\) than that in the rehabilitated forest \((0.25 \pm 0.02 \text{ mg kg}^{-1} \text{ soil})\) \(F_{1, 4} = 6.04, P = 0.001\). Effects of N addition on soil P availability varied depending on the rates of N addition and forest types (Fig. 1). In the disturbed forest, both N-treatments significantly decreased soil available P \((P = 0.049 \text{ and} 0.035 \text{ in the N50 and N100 plots, respectively)}. In contrast, soil available P increased with increasing rate of N addition in the rehabilitated forest, a difference that was significant between the Control and the N100 plots \((P = 0.001)\).

Nitrogen addition did not significantly affect \((P > 0.05)\) soil pH, soil SOC, or extractable Al and Fe in either forest (Table 3). Further analysis showed that there was no significant correlation between soil available P and any of the above variables of soil properties (data not shown).

PHOSPHORUS RELEASE FROM DECOMPOSING LITTER UNDER N ADDITION.—Temporal patterns of P content in decomposing litter were similar, decreasing exponentially with time in both forests and across all treatments (Table 4; Fig. 2). In the disturbed forest, rates of litter P loss decreased with increasing N-treatment levels for both types of litter. Litter P decomposition coefficients \((k)\) were in the order: Control \((0.51) > N50 (0.35) > N100 (0.34)\) for S. superba, and Control \((0.79) > N50 (0.68) > N100 (0.64)\) for C. chinensis (Table 4). The difference between Control and N50/N100 plots was significant for S. superba \((F_{1, 4} = 25.6, P = 0.001)\), but not for C. chinensis \((F_{1, 4} = 2.8, P = 0.141)\) (Table 4). The effect of N addition on litter P loss, however, depended on the length of litter exposure and the litter type \((F_{1, 4} = 13.9 \text{ for the interaction between time*litter type*treatment}, P = 0.003)\). After the first 6 mo, there was no significant difference between treatments for S. superba, but a significant difference was found between treatments in the remaining sampling dates \((F_{1, 4} = 4.1 \text{ for the interaction between time*treatment}, P = 0.003)\) (Fig. 2A). In contrast, a significant difference between treatments was found only on the third sampling date for C. chinensis \((F_{1, 4} = 4.1 \text{ for the interaction between time*treatment}, P = 0.003)\) (Fig. 2B). Repeated measures ANOVA showed that both rates of N addition significantly reduced the loss of P relative to the Control plots over the entire decomposition period for S. superba \((F_{1, 4} = 27.8, P = 0.001)\), but not significant for C. chinensis \((F_{1, 4} = 1.7, P = 0.265)\).

In the rehabilitated forest, there was no significant difference in P content of decomposing litter between treatments in any
The interaction between time and treatment was not significant in P loss for both types of litter ($F_{[8, 24]} = 1.7$ for *S. superba*, $P = 0.259$; and $F_{[8, 24]} = 0.4$ for *C. chinensis*, $P = 0.665$). As a result, repeated measures ANOVA showed that N addition had no significant effect on P content of the decomposing litter for both types of litter over the entire decomposition period ($F_{[2, 6]} = 0.7$ for *S. superba*, $P = 0.541$; $F_{[2, 6]} = 1.7$ for *C. chinensis*, $P = 0.253$; Fig. 2C–D). This is also confirmed by comparison of litter P loss coefficients (Table 4).

### TABLE 4. Litter P decomposition (loss) coefficients (k) for the selected tree species in two reforested tropical forests in southern China.

| Forest type/litter type | Control |  |  |  |  |  |  |  |
|-------------------------|---------|---|---|---|---|---|---|
|                         | k ($\gamma^{-1}$) | $r^2$ | P | k ($\gamma^{-1}$) | $r^2$ | P | k ($\gamma^{-1}$) |
| Disturbed               |         |    |   |         |    |   |         |
| *S. superba*            | 0.51(0.02)$^a$ | 0.97(0.01) | 0.003(0.0001) | 0.35(0.00)$^b$ | 0.96(0.00) | 0.004(0.0001) | 0.34(0.03)$^b$ | 0.87(0.02) | 0.022(0.0004) |
| *C. chinensis*          | 0.79(0.02)$^a$ | 0.96(0.00) | 0.003(0.0000) | 0.68(0.08)$^a$ | 0.97(0.01) | 0.003(0.0002) | 0.64(0.02)$^a$ | 0.99(0.01) | 0.001(0.0001) |
| Rehabilitated           |         |    |   |         |    |   |         |
| *S. superba*            | 0.50(0.08)$^a$ | 0.91(0.03) | 0.013(0.005) | 0.55(0.06)$^a$ | 0.94(0.02) | 0.007(0.002) | 0.59(0.03)$^a$ | 0.96(0.01) | 0.004(0.0001) |
| *C. chinensis*          | 0.70(0.05)$^a$ | 0.89(0.06) | 0.027(0.020) | 0.69(0.04)$^a$ | 0.94(0.02) | 0.008(0.004) | 0.75(0.04)$^a$ | 0.98(0.01) | 0.001(0.0001) |

Note: Values are means with 1 SE in parentheses, $N = 3$; Decomposition coefficients (k), coefficients of determination ($r^2$), and P-values are based on a single negative exponential model; Different letter within a single column indicate significant difference at $P < 0.01$. 

**FIGURE 2.** Changes in P content of decomposing litter in a disturbed and a rehabilitated tropical forest in southern China from October 2005 to September 2006. Bars indicate ±1 SE. Asterisk (*) indicates significant difference between Control plots and N-treatment at $P < 0.05$ level using planned contrast analysis ($N = 3$).
RESPONSE OF LITTERFALL PRODUCTION TO N ADDITION.—Total litterfall varied seasonally, across years, and between forests over the 4-yr period (Fig. 3). Mean monthly litterfall in the Control plots was higher in the growing season (April–September) than that in the winter (December–February) in both forests. Contribution of leaf litter to total litter was about 75 and 80 percent in the disturbed and rehabilitated forests, respectively, during the study period (data not shown).

Repeated measures ANOVA revealed no significant effects of N additions on total litterfall production in the disturbed forest during the study period ($F_{2, 6} = 3.2, P = 0.112$). In the rehabilitated forest, however, N addition increased total litter production, which was significantly ($F_{2, 6} = 7.7, P = 0.022$) higher in the N-treatment plots than that in the Control plots for the last 2 yr of the experiment (Fig. 3).

DISCUSSION

Our results demonstrate that N addition significantly altered availability of soil P in two forests. Based on evidence published in the literature, several mechanisms are possible in explaining this response: (1) changes in soil pH, subsequently altering geochemical equilibrium on sorption and precipitation of P with Al and Fe (McCormick & Borden 1972, Sherman et al. 2006, Weand et al. 2010); (2) changes in the production of low molecular-weight organic acids that would affect the release of inorganic P associated with Al and Fe (Zou et al. 1995, Singh & Amberger 1998, Harrold & Tabatabai 2006); (3) changes in rates of litterfall and/or decomposition (Zou et al. 1995, Mo et al. 2007); (4) changes in microbial immobilization of solution P (Zou et al. 1995); and (5) changes in quality and quantity of organic inputs.
Although there have been many studies on the response of litter addition (October 2005 to September 2006) significantly suppressed release rate of P from decomposing litter of *P. massoniana* in the disturbed forest in the first 24-mo (July 2003 to June 2005) of this N fertilization experiment (Mo et al. 2007), and similar to other studies in tropical forests reporting that fertilization can induce microbial nutrient retention within decomposing litter in nutrient-poor systems (McGroddy et al. 2004b, Cleveland et al. 2006). Hence, an important source of available P could be greatly reduced by inhibiting litter P release from decomposing litter. As above-ground litter production was equivalent between the Control plots and N-treatment plots, slower rates of P release from decomposing litter probably decreased return of P to soil, thus decreasing soil available P.

In the rehabilitated forest, further N addition (October 2005 to September 2006) had no significant effect on loss of P from decomposing litter, although the previous results of the first 24-mo (July 2003 to June 2005) in this N fertilization experiment showed that N addition significantly suppressed release rate of P from decomposing litter of *P. massoniana* relative to the controls (Mo et al. 2007). Ribeiro et al. (2002) also found that N addition stimulated release of P during litter decomposition. Considering both the use of identical litter from the decomposition experiment and the N-limited status of both disturbed and rehabilitated forests (Mo et al. 2006), attenuation of suppressing effects on litter P release may contribute to the increased soil available P, as microbes typically immobilize P from soil solution during decomposition in a P-poor system (McGroddy et al. 2004b, Olander & Vitousek 2004, Cleveland et al. 2006).

Under continual N addition, carbon and P may become a limiting factor for litter decomposition in both the disturbed and rehabilitated forests. It has been suggested that chronic N addition to N-limited forests soil will initially stimulate soil microbial activity, but over time, would result in a carbon-limited state after microbial demand for N was satisfied (Aber et al. 1998), with fresh materials decomposing faster than older materials (Berg & Soderstrom 1979). We have shown that N addition significantly increased the production of litterfall in the rehabilitated forest, e.g., at the third (September 2005 to August 2006) and fourth (September 2006 to August 2007) year of the experiment, but not in the disturbed forest (Fig. 3). These results indicate that the amount of substrate resupplied by litterfall in the N-treated plots is larger in the rehabilitated forest than in the disturbed forest under the same rate of N addition, suggesting that more carbon and P are available for soil microbial activity in the rehabilitated forest. Although N addition had no effect on the loss of P from decomposing litter in the rehabilitated forest, higher production of total litterfall probably facilitated soil P accumulation.

Given the central role of litter production and decomposition in determining the response of soil available P to N addition, their contrasting response patterns to N addition in the disturbed versus rehabilitated forests may have arisen from changes in tree species composition and soil nutrient status, brought about by different land-use practices. As mentioned, both forests originated...
from man-made pine forests. In the rehabilitated forest, after more than 70 yr of rehabilitation, establishment of regional broadleaf species via natural dispersal has changed plant composition, which is dominated by *P. massoniana* and a few broadleaf species (mainly *S. superba* (Mo et al. 2003). The disturbed forest, however, has been under nearly constant human pressure (generally the harvesting of understory and litter to satisfy fuel needs of local people) from time of planting until the late 1990s (Wang et al. 1982, Brown et al. 1995, Mo et al. 1995). These intensive, ongoing removals in the disturbed forest resulted in minimal remnant vegetation cover, minimized competitive interaction among understory plants, and maintained overstory dominance by *P. massoniana* by eliminating seedlings of potential dominant/co-dominant species (Mo et al. 1995, 2003, Lu et al. 2011). Presently, the disturbed forest is dominated by pines, representing >95 percent of total basal area (Table 1). In contrast, the rehabilitated forest was co-dominated by pines and broadleaf trees, 40 and 60 percent relative basal area, respectively (Table 1).

Different forest land-use practices have also led to differences in soil nutrient status. Previous work showed that both forests are still N-limited because of previous deforestation activities that led to substantial loss of soil N (Mo et al. 2003, 2006). Mo et al. (1995) have shown, however, that harvest practices in the disturbed forest removed about 44–73 percent of the total quantity nutrients in the litter and understory production, an amount that appeared to exceed most nutrient input from atmospheric deposition. Mo et al. (2003) further suggested that successful resupply of soil nutrients (e.g., available soil N) in the rehabilitated forests is possible, but not in the disturbed forest because of continued harvest practices.

Contrasting plant species and nutrient status may lead to the differences in plant growth and litterfall production between the two forests. Lu et al. (2011) suggested that low soil N and nutrient status may hinder the growth of the trees in the disturbed forest. The slow growth and low nutrient turnover rates of *P. massoniana*, even at high N inputs, may result in no significant changes of plant growth (e.g., canopy closure) during the study period (Lu et al. 2011). In the rehabilitated forest, successful restoration of soil nutrients on severely degraded lands might be possible (Mo et al. 2003), and N addition was shown to increase plant growth (Lu et al. 2011). Hence, we hypothesize that the increased litterfall could be the result of greater plant growth with N addition in the rehabilitated forest.

In conclusion, our results suggest that continued N addition can affect soil P availability in reforested regions of southern China, but that this response may vary greatly with overstory tree species composition and soil nutrient status as influenced by land-use practices. In the disturbed forest, the significant decrease of soil available P under N-treatments may be the result of the significant suppressing of P release in the absence of change in litter production. In contrast, the rehabilitated forest showed that N addition significantly increased available soil P, an increase that contributed to higher litter production and attenuation of suppressing effects on release of litter P. This study provides new insights on how elevated N deposition affects P availability, with potential applications toward ecosystem health and services. Increases in available soil P in the rehabilitated forest have implications for the potential effectiveness of increased P in reducing the symptoms of N saturation in the future. Our results were evaluated for just one site of each forest type; thus, applications of our results to other reforested forests in southern China should be made with caution. Indeed, confirmation of our results will require further investigation in other regions.

**ACKNOWLEDGMENTS**

We thank the subject editor and the two anonymous reviewers for their valuable comments and suggestions on the manuscript, and Drs. Sandra Brown, Per Gundersen, Guirui Yu, and Linghao Li for their invaluable suggestions on the earlier version of the paper. This study was funded by the National Basic Research Program of China (2010CB833502), National Natural Science Foundation of China (Nos. 30970521, 30900202), National Science Foundation of Guangdong Province, China (Grant 8351065005000001), and the Knowledge Innovation Program of the Chinese Academy of Sciences (Grant No.KSCX2-EW-J-28).

**LITERATURE CITED**


