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SPECIAL FEATURE: REVIEW

Soil nitrogen dynamics of forest ecosystems under environmental changes

Atmospheric deposition and leaching of nitrogen in Chinese forest ecosystems

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Abstract Data have been compiled from published sources on nitrogen (N) fluxes in precipitation, throughfall, and leaching from 69 forest ecosystems at 50 sites throughout China, to examine at a national level: (1) N input in precipitation and throughfall, (2) how precipitation N changes after the interaction with canopy, and (3) whether N leaching increases with increasing N deposition and, if so, to what extent. The deposition of dissolved inorganic N (DIN) in precipitation ranged from 2.6 to 48.2 kg N ha^{-1} year^{-1}, with an average of 16.6 kg N ha^{-1} year^{-1}. Ammonium was the dominant form of N at most sites, accounting for, on average, 63% of total inorganic N deposition. Nitrate accounted for the remaining 37%. On average, DIN fluxes increased through forest canopies, by 40% and 34% in broad-leaved and coniferous forests, respectively. No significant difference in throughfall DIN inputs was found between the two forest types. Overall, 22% of the throughfall DIN input was leached from forest ecosystems in China, which is lower than the 50–59% observed for European forests. Simple calculations indicate that Chinese forests have great potential to absorb carbon dioxide from the atmosphere, because of the large forest area and high N deposition.

Keywords Carbon sequestration · Chinese forests · Nitrogen deposition · Nitrogen leaching · Nitrogen retention

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Introduction

The rapid expansion of industry and intensive agriculture, and human population growth, in Asia has increased the use of reactive nitrogen (Nr) and its emission into the environment (Zheng et al. 2002; Galloway et al. 2004). In China, emission of Nr into the environment has significantly increased during recent decades; total NO_x emission increased from 8.4 Tg year^{-1} in 1990 to 11.3 Tg year^{-1} in 2000, and total NH_3 emission increased from 10.8 to 13.6 Tg year^{-1} over the same time period (Lu and Tian 2007), and further increases are predicted (Zheng et al. 2002). Nitrogen deposition of greater than 25 kg N ha^{-1} year^{-1}, a threshold above which N saturation is often observed in temperate forests in Europe (Gundersen et al. 1998a, 2006; Dise et al. 1998, 2009), has frequently been reported in many regions in China (Liu et al. 2006; He et al. 2007; Lu and Tian 2007; Zhang et al. 2008; Fang et al. 2008, 2011). In China, average wet deposition of inorganic N is estimated to be 9.9 kg N ha^{-1} $year^{-1}$ (Lu and Tian 2007). This is larger than in both the United States (3.0 kg N ha^{-1} $year^{-1}$) and western Europe (6.8 kg N ha^{-1} $year^{-1}$) (Holland et al. 2005).

Chronic elevated N deposition in forest ecosystems may lead to N saturation when the biotic demand for N is exceeded. Nitrogen saturation leads to increased rates of N cycling and losses of nitrate (NO_3^-) from the root zone, soil and surface water acidification, plant nutrient imbalances, and even, in some cases, forest decline (Fenn et al. 1998; Aber et al. 1998, 2003). The onset of N saturation and the magnitude of the effects of N deposition are largely dependent on the capacity to retain anthropogenic N input, i.e. ecosystem initial N status and the history and form of N input (Gundersen et al. 1998b; Matson et al. 1999).

On the other hand, retained N deposition is likely to enhance primary production of many terrestrial ecosystems, and thus increase carbon dioxide (CO_2) sequestration from the atmosphere, although there is large uncertainty about the magnitude of additional C sequestration on global and regional scales (Nadelhoffer et al. 1999; Magnani et al. 2007; Reay et al. 2008; Sutton et al. 2008; de Vries et al. 2009). Buildup of N is also likely to increase accumulation of soil organic matter in the form of increased leaf/needle biomass and litter production, and reduced decomposition of organic matter, depending on the stage of humus formation (Neff et al. 2002a; de Vries et al. 2009).

Forests cover 14% of China, and assessment of national forest inventory data shows that the carbon stock in forest biomass increased significantly from the 1980s to the 1990s (from 0.06 ± 0.03 to 0.09 ± 0.04 Pg C $year^{-1}$, Fang et al. 2007; Piao et al. 2009). An average net carbon sink of 0.18 ± 0.07 Pg C $year^{-1}$ was estimated for all Chinese terrestrial ecosystems during the same period, with, on average, 58% of this sink in the biomass and the rest in soil organic matter (Piao et al. 2009). This C sink has mainly been attributed to regional climate change and large-scale plantation programs active since the 1980s (Piao et al. 2009). However, we propose that atmospheric N deposition during the same time period may also be one of important contributors to increased forest biomass.

Although N deposition in precipitation has been mapped for China (Lu and Tian 2007), we do not know the total N loads of forests and the geographical distribution of elevated inputs. This is because forests may have N input different from that measured in precipitation N, because the forest canopy can efficiently scavenge gas and particulate N from the atmosphere (Lovett and Lindberg 1993; Fenn and Poth 2004; Chen and Mulder 2007a, b; Fang et al. 2011). Furthermore, the efficiency of C sequestration is related to N retention capacity (de Vries et al. 2006). To identify current forest N status and, thereby, the potential response to future scenarios of N deposition in China, synthesis of N fluxes, especially throughfall, at a national level is required.

In this paper a preliminary analysis of data compiled from 69 forest ecosystems across China was used to quantify N input in precipitation and throughfall, and N leaching losses at a national level. The objectives of this study were to examine:

1. the level of N input in precipitation and throughfall;
2. how the speciation of N in precipitation changes after the interaction with canopy; and
3. whether N leaching increases with increasing N deposition and, if so, to what extent.

We discuss how the patterns observed in China differ from those in Europe and North America. On the basis of these findings, the potential C sequestration induced by elevated N deposition in Chinese forest ecosystems was approximately estimated, and is discussed.

Methods

Data on forest N fluxes (precipitation, canopy throughfall, and leaching losses) were compiled from the literature since 1980 (Supplementary Table 1). Only data from forests with annual fluxes were included, with the exception of seven forests (in northern China) where N fluxes only for the rainy season were available, and so N fluxes might have been underestimated. In the earlier years (1980s and 1990s), N fluxes were often reported only as total dissolved N (TDN). More recently, dissolved inorganic N (DIN, NH_4^+, plus NO_3^-) fluxes have been reported, and the contribution from dissolved organic N (DON, the difference between TDN and DIN) has also been reported for some forests. For forests where N fluxes were monitored and reported for several years, average values were used in the analysis. Not all N fluxes data were available for all forests. In total, 69 forests with N fluxes at 50 sites were included in this study (Fig. 1). Forests were classified into broad-leaved or coniferous type on the basis of the dominant species.

In addition, the pH and N concentration were compiled when available. Information on study period, annual air temperature, water volume (e.g. amount of precipitation), forest age, elevation, and species composition were extracted from the original publications (Supplementary Table 1). Annual precipitation and temperature were obtained from related publications for the sites for which such information was not presented.

Results

Nitrogen deposition in precipitation

For 32 of the 50 forest sites, NH_4^+ and NO_3^- fluxes in precipitation were reported separately (Fig. 2a), whereas for the rest only TDN was reported. DIN fluxes ranged from 2.6 kg N ha^{-1} year^{-1} in Changbaishan, Jilin province, and 3.3 kg N ha^{-1} year^{-1} in Liangshui, Heilongjiang, to 48.2 kg N ha^{-1} year^{-1} at a site in Jiangxi province, with an average of 16.6 kg N ha^{-1} year^{-1} (Fig. 2a; Table 1). Average DIN flux in precipitation increased from 12.6 kg N ha^{-1} year^{-1} ($n = 10$) in the 1990s to 17.7 kg N ha^{-1} year^{-1} ($n = 20$) in the 2000s, but the increase was not statistically significant (data not shown). Ammonium ranged from 1.6 to 47.1 kg N ha^{-1} year^{-1}, and NO_3^- ranged from 1.0 to 15.6 kg N ha^{-1} year^{-1} (Figs. 2a, 3). At most sites ammonium was the dominant form of N in precipitation, accounting for, on average, 63% of total inorganic N (Fig. 3a). Variation of NH_4^+ loading explained 91% of the variability of DIN in precipitation among the sites (Fig. 3a). Nitrate accounted for, on average, 37% of the DIN, and represented 70% of the variability of DIN (Fig. 3b). Ammonium also correlated with NO_3^- ($R^2 = 0.47$, $P < 0.01$, $n = 31$; one site in Jiangxi province excluded), suggesting that high NH_4^+ input sites usually also had high NO_3^- input.

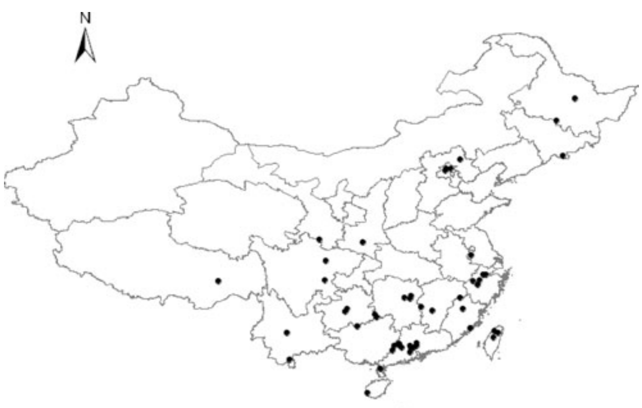

Fig. 1 The location of the 50 precipitation sites included in this synthesis

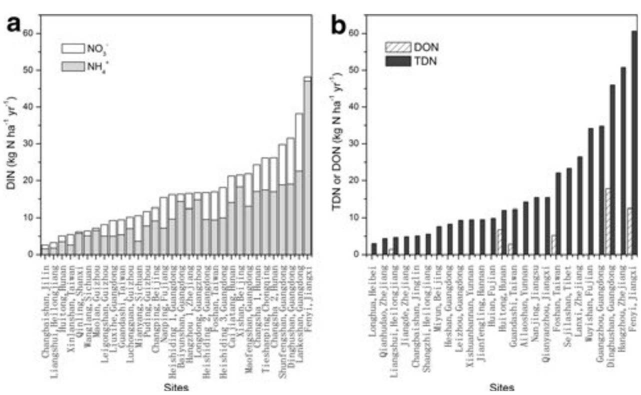

Fig. 2 Input of dissolved inorganic N (DIN, 32 sites, **a**) and total dissolved N (TDN, 25 sites, **b**) in precipitation on to forests in China

Data on TDN were available for 25 of the forest sites (for six of which DIN fluxes had also been measured, and thus were included in both Fig. 2a, b). TDN fluxes ranged from 3.0 to 60.6 kg N ha^{-1} year^{-1} (Fig. 2b). TDN flux averaged 18.0 kg N ha^{-1} year^{-1}. At six sites where both DIN and TDN were measured, DON fluxes were between 1.4 kg N ha^{-1} year^{-1} at Liangshui, Heilongjiang province, and 17.8 kg N ha^{-1} year^{-1} at Dinghushan, Guangdong. Average DON flux was 7.7 kg N ha^{-1} year^{-1} (Fig. 2b). At these sites, DON constituted between 21 and 56% of TDN in precipitation (average 32%).

Throughfall nitrogen deposition

Data on DIN in throughfall were available for 42 forests, of which 18 were broad-leaved and 24 were coniferous (Table 2). Ammonium flux varied substantially, from 0.5 to

Table 1 Nitrogen input (kg N ha^{-1} year^{-1}) in precipitation at 31 forest sites throughout China, in comparison with other studies

	NH_4^+	NO_3^-	DIN	References
Chinese forest site (1980–2009)	11.3	5.3	16.6	This study
Linzhi, Tibet, remote site (2005–2006)	1.6	0.6	2.2	Jia et al. (2009)
Waliguan, Qinghai, remote site (1997)	2.5	0.4	2.9	Tang et al. (2000)
China (1984–2004)	7.1	2.8	9.9	Lu and Tian (2007)
USA (1978–1994)	1.4	1.6	3.0	Holland et al. (2005)
Western Europe (1978–1994)	4.2	2.6	6.8	Holland et al. (2005)
Europe	7.1	5.5	11.2	Dise et al. (2009)
8 Chinese EANET sites in 2008	9.5	5.6	15.1	EANET (2009)
12 Japanese EANET sites in 2008	2.9	3.2	6.1	EANET (2009)
51 EANET sites in 2008	5.8	3.8	9.6	EANET (2009)

EANET, acid deposition monitoring network in east Asia

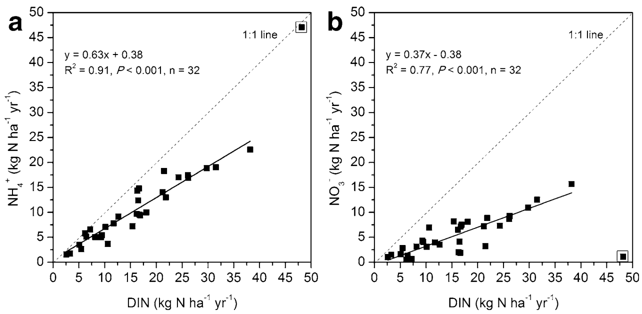

Fig. 3 Fluxes of DIN versus NH_4^+ (**a**) and NO_3^- (**b**) in precipitation at 31 forest sites across China. One site with extremely high NH_4^+ but low NO_3^- input (marked by *open squares*) was excluded from the regression analysis

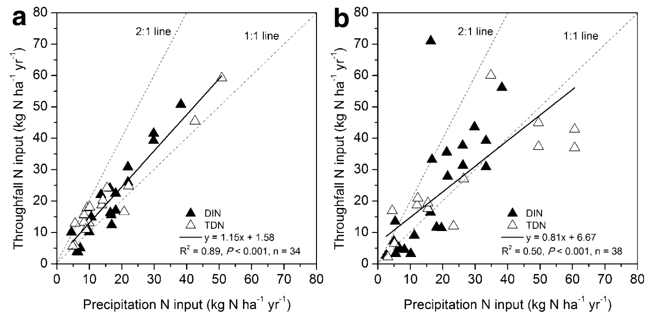

Fig. 4 Precipitation N input versus throughfall N input in broad-leaved forests (**a**) and coniferous forests (**b**) in China. In **a**, $y = 1.40x - 3.15$, $R^2 = 0.91$ for DIN, $n = 18$; $y = 1.02x + 4.23$, $R^2 = 0.93$, for TDN, $n = 16$. In **b**, $y = 1.34x - 0.28$, $R^2 = 0.58$ for DIN, $n = 24$, $y = 0.62x + 9.42$, $R^2 = 0.62$ for TDN, $n = 14$. For all correlations $P \leq 0.001$. The regression lines in figures are across both DIN and TDN

Table 2 Nitrogen input in throughfall in Chinese forests (kg N ha^{-1} year^{-1})

	NH_4^+	NO_3^-	DIN	TDN	DON
Broad-leaved forests					
Minimum	0.5	0.2	3.8	5.6	12.4
Maximum	25.8	25.0	50.8	59.2	20.1
Mean	10.5	9.9	20.4	19.7	16.3
SE	1.9	1.6	3.1	3.3	3.9
n	18	18	18	17	2
Coniferous forests					
Minimum	0.7	1.1	2.8	2.2	1.3
Maximum	68.9	26.7	71.0	60.1	18.2
Mean	13.5	7.0	21.0	23.8	8.3
SE	3.3	1.4	3.9	3.9	2.7
n	23	23	24	18	6
All forests					
Minimum	0.5	0.2	2.8	2.2	1.3
Maximum	68.9	26.7	71.0	60.1	20.1
Mean	12.2	8.3	20.8	21.8	10.3
SE	2.1	1.1	2.6	2.6	2.5
n	41	41	42	35	8

68.9 kg N ha^{-1} year^{-1}, whereas NO_3^- flux ranged from 0.2 to 26.7 kg N ha^{-1} year^{-1}. Average NH_4^+ and NO_3^- fluxes were 12.2 and 8.3 kg N ha^{-1} year^{-1}, respectively. DIN ranged from 2.8 to 71.0 kg N ha^{-1} year^{-1}, with an average of 20.8 kg N ha^{-1} year^{-1} (Table 2). TDN flux was estimated for 35 forests, for eight of which DIN data were also available (Table 2). The range of TDN was similar to that of DIN. Mean DON flux was 10.3 kg N ha^{-1} year^{-1} (Table 2). No significant difference in fluxes was observed for any N species or fractions between forest types ($P > 0.05$). A TDN input as high as 23.3 kg N ha^{-1} year^{-1} was unexpectedly observed in a spruce forest in Tibet (Xin and Zhai 2003; Fig. 2b) where N deposition is generally found to be among the lowest (Jia et al. 2009).

N input in throughfall is empirically correlated with the N input in precipitation. But this relationship varied between forest types and between N species (Fig. 4). In the broad-leaved forests throughfall N input was, on average, 15% higher than N inputs in precipitation, because of washing off of additional N (e.g., contributed by dry deposition or exchange) from the canopy (Fig. 4a). In contrast, in the coniferous forests, throughfall N input was 19% lower than precipitation N input. This reduction was caused by lower TDN input in throughfall than in precipitation (Fig. 4b). In fact, DIN input increased substantially after passing through the canopy in both the broad-leaved (by 40%, Fig. 4a) and coniferous forests (by 34%, Fig. 4b).

Nitrogen leaching

Data on annual N leaching losses below the rooting zone or via small streams were available for 34 forests, for seven of which both TDN and DIN data were given (Fig. 5). TDN leaching (20 forests) ranges from 0.05–39.0 kg N ha^{-1} year^{-1} (Fig. 5), average 8.8 kg N ha^{-1} year^{-1}. Of these forest ecosystems, 45% leached less than 5 kg N ha^{-1} year^{-1}, and 70% less than 10 kg N ha^{-1} year^{-1} (Fig. 6). DIN leaching (21 forests) was from 0.3 to 36.5 kg N ha^{-1} year^{-1}, with an overall mean of 5.7 kg N ha^{-1} year^{-1}. Of these forest ecosystems, 62% leached less than 5 kg N ha^{-1} year^{-1}, and 76% less than 10 kg N ha^{-1} year^{-1} (Fig. 6). High N leaching was found in warm and humid areas, for example Hainan, Guangdong, and Yunan (data not shown). For most of the 18 forests for which DIN concentrations were available, NO_3^- was the dominant N form in stream water and below the rooting zone, although NH_4^+ accounted for, on average, 25% of DIN and was actually the dominant DIN loss in three forests. The contribution of NH_4^+ to DIN leaching was usually high in forests

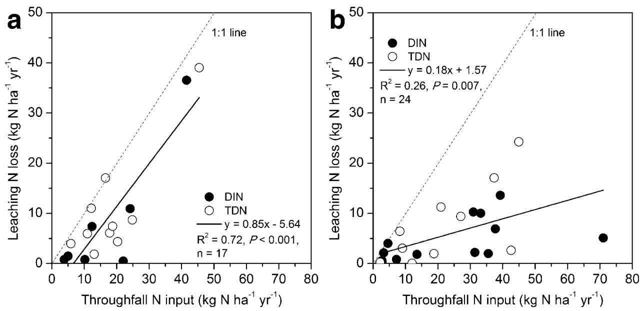

Fig. 5 Throughfall N input vs. leaching loss in broad-leaved forests (**a**) and coniferous forests (**b**) in China. In **a**, $y = 0.85x - 6.18$, $R^2 = 0.71$, $P = 0.011$ for DIN, $n = 7$; $y = 0.85x - 5.15$, $R^2 = 0.68$, $P = 0.002$ for TDN, $n = 10$. In **b**, $y = 0.12x + 1.64$, $R^2 = 0.26$, $P = 0.035$ for DIN, $n = 14$; $y = 0.14x - 0.39$, $R^2 = 0.40$, $P = 0.03$ for TDN, $n = 10$. The regression lines in figures are across both DIN and TDN

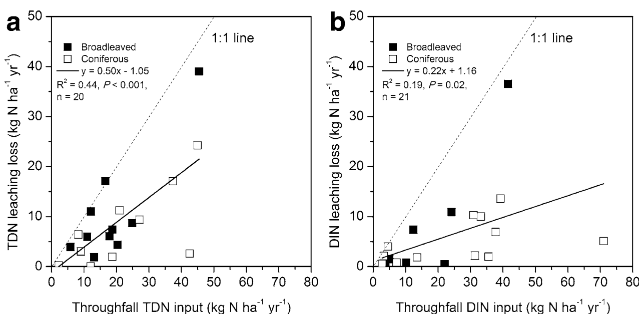

Fig. 6 Throughfall N input versus leaching loss in China for TDN (**a**) or DIN (**b**). The regression lines are across all forests

with low N leaching rate (0.5–2.2 kg N ha^{-1} year^{-1}, data not shown). This is consistent with the report for N loss in unpolluted temperate forests in South America where NH_4^+ was shown to be dominant in inorganic N in stream water (Perakis and Hedin 2002). DON flux was up to 16.8 kg N ha^{-1} year^{-1} in a broad-leaved forest in Guangdong (Fang et al. 2008), but the average value was only 3.1 kg N ha^{-1} year^{-1} for the other six forests (data not shown).

There was no significant difference in N leaching between the broad-leaved and coniferous forests, irrespective of N species, though mean N leaching was higher in broad-leaved forests than in coniferous forests (Fig. 6). For example, DIN leaching was, on average, 8.3 and 4.3 kg N ha^{-1} year^{-1} in broad-leaved and coniferous forests, respectively (Fig. 6b). The average fraction of throughfall N lost by leaching was much greater in the broad-leaved forests (85%) than in the coniferous forests (18%) for both DIN and TDN (Fig. 5). A possible bias that may contribute to the high N leaching rate from the broad-leaved forests was that they are more undisturbed than the coniferous forests. The leaching rate was also dependent on N species; overall, 50% of TDN and 22% of DIN input in throughfall was leached across all forests (Fig. 6).

Discussion

Nitrogen deposition

Precipitation DIN deposition on the studied forest sites in China was, on average, 16.6 kg N ha^{-1} year^{-1} which was more than 5 times the background level of 2.2–2.9 kg N ha^{-1} year^{-1} measured at two sites located in remote areas (Table 1). This is much higher than the previously estimated average value of 9.9 kg N ha^{-1} year^{-1} based on a mapping exercise for the whole of China (Lu and Tian 2007), but similar to the mean value for eight EANET sites in China (Table 1). However, the spatial pattern in precipitation DIN input in this study was similar to that obtained by Lu and Tian (2007), covering high DIN input in the central southern regions, for example Guangdong, Jiangxi, Zhejiang, Chongqing, Hunan, and low input in the underdeveloped areas in the west and north of China (Figs. 1, 2). The higher N input in precipitation on to the forest sites included in our study compared with the national level estimated by Lu and Tian (2007) may be because of uneven distribution of investigated forest sites with many sites in southern China, especially in the Pearl River Regions of Guangdong province, one of greatest economic centers in China with heavy air N pollution (Fig. 1, Fang et al. 2011). A stronger impact of urban rather than agricultural sources of atmospheric Nr is supported by the higher NO_3^- fraction in precipitation at our study sites (37%, Fig. 3b) compared with that in the national level estimate at 28% (Lu and Tian 2007). The DIN input with precipitation is also substantially higher than that measured in Europe, the US, and east Asia (Table 1).

The average TDN input in precipitation, at 18.0 kg N ha^{-1} year^{-1}, differs only slightly from that measured as DIN input, and the spatial patterns are similar (Fig. 2). The difference between TDN and DIN input at 1.4 kg N ha^{-1} year^{-1} could comprise DON, although TDN and DIN were not always measured at the same sites. Neff et al. (2002b) reported a median value of 2.2 kg N ha^{-1} year^{-1} for precipitation DON flux in a global synthesis. However, the measured DON flux was found to be, on average, 7.7 kg N ha^{-1} year^{-1} for the six Chinese forest sites included in our synthesis (Fig. 2b). This is in the upper range of that reported for other regions of the world (0.6–10.9 kg N ha^{-1} year^{-1}, Neff et al. 2002b), but in agreement with other reports from China: Xiao (2005) reported high DON input (on average 15.8 kg N ha^{-1} year^{-1}) for eleven sites in Zhangzhou city of Fujian province, and Zhang et al. (2008) reported that DON deposition in precipitation at 15 rural, suburban and urban sites ranged from 1 to 27 kg N ha^{-1} year^{-1}, with an average of 8.6 kg N ha^{-1} year^{-1}. At the 6 forest sites in our database, DON input accounted, on average, for 32%

of the TDN in precipitation, similar to the average fraction for the 15 sites studied throughout China (Zhang et al. 2008). These results suggest that high DON deposition does exist in the heavily air polluted regions in China and that DON is an important N fraction in precipitation inputs which needs more attention in future studies. The sources of high DON inputs are unclear (Fang et al. 2008). Conceptually, atmospheric organic N (AON), which is measured as DON in precipitation and throughfall, can be divided into three types—organic nitrate, reduced AON, and biological/terrestrial AON (Neff et al. 2002b). Organic nitrates are oxidized end products of reactions of hydrocarbons with NO_x ($NO + NO_2$) in polluted air masses such as those occurring over southern China. Thus, organic nitrates are likely to contribute to the high DON deposition observed in some regions with severe N pollution (Fang et al. 2008).

In this data compilation, throughfall DIN input ranged from 2.8 to 71.0 kg N ha^{-1} $year^{-1}$, which is similar to the range compiled for European forests (<5 kg N ha^{-1} $year^{-1}$ in remote forest ecosystems to > 60 kg N ha^{-1} $year^{-1}$ in heavily polluted areas; Gundersen 1995; Dise et al. 1998; Gundersen et al. 2006). In Europe sites with high throughfall, N input was mainly caused by increased NH_4^+ inputs, and NO_3^- was a dominant fraction at the lower deposition sites (Gundersen 1995). In China, it seems that both NH_4^+ and NO_3^- increased concurrently (Fig. 3), but with NH_4^+ overall accounting for 63% of the DIN input. Hence NH_4^+ is the dominant N form entering forests in both Europe and China, pointing to agricultural emissions as the main source of N. In contrast, in North America NO_3^- is slightly more important than NH_4^+ in N input to forests (Ollinger et al. 1993; Fenn and Poth 2004; Holland et al. 2005; Golden and Boyer 2009).

Forest canopies are efficient traps of gases and particles from the air, and, hence, N deposition in throughfall is usually substantially larger than both precipitation N deposition and total N deposition on open land (Fenn and Poth 2004). Conifers tend to have higher throughfall N input than broad-leaved forests, because of differences in leaf area, leaf retention, and canopy structure (Kristensen et al. 2004). However, precipitation N is also likely to be taken up by leaves through interactions with the canopy, particularly at low N input sites (Lovett and Lindberg 1993). At the sites included in our database both enhancement of throughfall N and a net canopy uptake was observed (Fig. 4).

In China the dry deposition of N (i.e., calculated as throughfall minus precipitation N) was almost always less than precipitation N (sites falling below the 2:1 line in Fig. 4) whereas in Europe dry deposition was up to twice the precipitation N (Kristensen et al. 2004). In North America, dry deposition usually contributes only a few kg N ha^{-1} $year^{-1}$ or ca. 20–46% of wet deposition in NE USA (Ollinger et al. 1993). An exception to this pattern in North America was observed in the summer-dry climate in the mountains of southern California downwind from the major cities. N deposition in precipitation was 3.3 and 12 kg N ha^{-1} $year^{-1}$ at a remote and an exposed site, respectively, whereas their particular throughfall N deposition was 15 and 144 kg N ha^{-1} $year^{-1}$, respectively (Fenn and Poth 2004). A reason for these continental differences could be that the European forests monitored are closer to the agricultural N sources in the patchy landscape than the forest monitored in China. This may also be the reason for the low dry deposition fraction in NE USA where data come from large forest areas remote from air pollution source regions. Gas and particle concentrations decline with distances from sources, in part because of incorporation in clouds. Another reason for low dry deposition in China could be the humid climatic conditions in the southern provinces from which most of the high deposition data originate. High humidity, frequent rainfall, and low wind speeds limit dry deposition and promote incorporation in clouds and rain, as opposed to the situation with high dry deposition in the dry climate of California discussed above.

In the broad-leaved forests throughfall N was more consistently higher than precipitation N (Fig. 4a) compared with in the coniferous forests, where, in turn, the variability was high (Fig. 4b). We did not observe a significant difference in throughfall DIN input between broad-leaved and coniferous forests across the dataset, as is usually observed in temperate forests (Kristensen et al. 2004). This could be because most Chinese broad-leaved trees are evergreens whereas temperate broad-leaved trees are deciduous. The observed lower throughfall N in temperate broad-leaved trees compared with conifers is largely attributed to the leafless winter period for the broad-leaved trees (Kristensen et al. 2004). Then again, the minimum importance of dry deposition in China discussed above makes it less likely that the canopy structure or forest type will modify the total N input measured in throughfall.

Nitrogen leaching

Input–output budgets from European forests have shown that above a threshold loading of approximately 10 kg N ha^{-1} $year^{-1}$ in throughfall N, many sites seem to be N saturated and have NO_3^- leaching rates above 5 kg N ha^{-1} $year^{-1}$ (Gundersen 1995; Nilsson et al. 1998; Dise et al. 1998, 2009; MacDonald et al. 2002; Kristensen et al. 2004). Below 10 kg N ha^{-1} $year^{-1}$ in throughfall, elevated NO_3^- leaching is rare in all data compilations from Europe (Gundersen et al. 1998b; Nilsson et al. 1998; MacDonald et al. 2002; Kristensen et al. 2004; Dise et al.

2009). All European sites receiving more than 25–30 kg N ha^{-1} $year^{-1}$ of N in throughfall had elevated NO_3^- leaching (Gundersen et al. 2006; Dise et al. 2009). In a compilation of input–output data from streams in NE USA, Aber et al. (2003) reported a threshold at N deposition of 7 kg N ha^{-1} $year^{-1}$. However, they emphasized that the N deposition was probably underestimated somewhat because it was the estimate for the base of the catchment, because deposition usually increases with elevation (Gundersen et al. 2006). The N input thresholds for elevated NO_3^- leaching found on both continents may therefore nearly be identical at approximately 10 kg N ha^{-1} $year^{-1}$ (Aber et al. 2003).

In China, however, elevated N leaching above 5 kg N ha^{-1} $year^{-1}$ occurs in forest ecosystems when they receive throughfall N of more than 5 kg N ha^{-1} $year^{-1}$ (Figs. 5, 6). The relatively high N leaching in some forests receiving low N input is different from those in European and American forests where a threshold is more distinct. The reason for this is unclear. One may argue that this is because of differences in climatic drivers. In temperate climate (Europe and North America) the soil water mainly moves in the dormant season (fall, winter, and spring) when there is no/low biological release of N, whereas in monsoon Asia, particularly in southern and eastern China, there is excess water in the most biologically active season (summer) where some N will be available for transport with moving water. In other words, with monsoon climate with warm and wet summers there will be some unavoidable N loss, controlled more by hydrological processes than by lack of biological demand (Ohte et al. 2001; Fang et al. 2008). N leaching above 10 kg N ha^{-1} $year^{-1}$ occurs when throughfall N input is higher than 17 kg N ha^{-1} $year^{-1}$ (Fig. 5). Nevertheless, forests receiving 5–45 kg N ha^{-1} $year^{-1}$ respond very differently with regard to N leaching; some forests retain N inputs almost fully and a few forests release almost all N input.

Only 22% of throughfall N input was leached from Chinese forest ecosystems if considering only DIN, although the leaching rate of throughfall input was half for TDN (Fig. 6). Ignoring N losses via other pathways, for example denitrification, this implies that approximately 80% of N input has been retained in the ecosystems. This retention is similar to the results based on ^{15}N field experiments in nine temperate forests (Nadelhoffer et al. 1999). The overall N leaching rate in the Chinese forests is lower than the 50–59% observed for European forests (Gundersen 1995; Gundersen et al. 1998a, 2006; Dise et al. 1998, 2009). This could be because the history of N pollution in China is shorter than in Europe. Economic development started to increase in the early 1980s in China, associated with the heavy use of N fertilizer and increased emission of NO_x and N_2O (Zheng et al. 2002). In contrast, both fertilizer N use and NO_x emission increased sharply in the 1960s and 1970s and peaked around the 1980s in Europe (van Egmond et al. 2002).

Another possible reason for the overall lower N leaching is that the N leaching in our data compilation included an estimate of N leaching into streams at seven of the 21 forests whereas the database for European forests mainly included that estimate below the rooting zone (Gundersen et al. 1998a, 2006; Dise et al. 2009). Nitrogen leaching from below the rooting zone plus overland flow directly reflects the dissolved N response of plant–soil interactions in the N cycle, whereas stream water reflects additional N processes, for example accumulation in bog areas, denitrification in the riparian or hyporheic zone, and in-stream N conversion. These "stream" processes mainly consume dissolved N, and thus lower concentrations of N should be expected in stream water than in soil water below the rooting zone (Gundersen et al. 2006).

Implications for CO_2 sequestration in Chinese forests

The effect of N deposition on forest C sequestration relies largely on the sensitivity of ecosystem C processes response to N. Debate on this topic has recently been intensified by Magnani et al. (2007). In this paper, a strong positive relationship was found between mean lifetime C sequestration (in terms of net ecosystem production; NEP_{av}) and N deposition. Their data indicated a carbon response of approximately 475 kg C per kg N in total N deposition. Sutton et al. (2008) found the response to be at 177 kg C per kg, when plotting the measured NEP_{av} data of Magnani et al. (2007) against better estimates of total N deposition data. However, these rates of C sequestration from N deposition input were still thought to be too high, because other factors co-varied with N deposition and may have contributed to the observed increase in NEP (de Vries et al. 2009).

A short-term (1–3 year) ^{15}N labeled tracer experiment in nine temperate forests suggested that 5% of the N input ended up in forest biomass whereas 70% was retained in soils with low C/N ratio (Nadelhoffer et al. 1999). Assuming a constant N uptake fraction of 0.05 and a constant N retention fraction of 0.70, and an average C/N ratio in stem wood of 500 and in forest soils of 30, then additional N deposition of 1 kg ha^{-1} $year^{-1}$ leads to a C sequestration of 46 kg ha^{-1} $year^{-1}$, of which 25 kg ha^{-1} $year^{-1}$ is retained in stem wood (0.05×500) and 21 kg ha^{-1} $year^{-1}$ in soil (0.7×30) (Nadelhoffer et al. 1999; see also de Vries et al. 2009). For European forests, de Vries et al. (2009) synthesized the ranges in C sequestration per kg N addition in above-ground biomass and in soil organic matter. They found that the results from various studies were in close agreement and showed that

Table 3 Approximate estimates of nitrogen-induced carbon sequestration in forest biomass throughout China

	1990s	2000s
Nitrogen deposition in throughfall (kg N ha^{-1} $year^{-1}$)	21.0	23.1
Forest area (10^6 ha)	132–143[a]	195[b]
C sequestration in forest trees (Tg C $year^{-1}$)[c]	~72	113

[a] Data from Fang et al. (2007)

[b] Forest area in 2008 based on the 7th national forest inventory in China, http://www.022net.com/2009/11-17/50706327323043.html

[c] Estimate assuming that deposition of 1 kg N sequesters 25 kg C from the atmosphere in additional biomass (de Vries et al. 2009)

above-ground accumulation of carbon in forests is generally within the range 15–40 kg C per kg N, and that the uncertainty in C sequestration per kg N addition in soils is larger than for above-ground biomass and varies on average between 5 and 35 kg C per kg N. For Europe, de Vries et al. (2009) suggested approximately 25 kg C per kg N as likely C sensitivity to N for both biomass and soil.

In China, average DIN fluxes in throughfall were 21.0 and 23.1 kg N ha^{-1} $year^{-1}$ in the 1990s and 2000s, respectively (Table 3). If we assume a C sensitivity of 25 kg C per kg N in the biomass, as suggested for European forests (de Vries et al. 2006), this corresponds to forest C sequestration of 72 Tg $year^{-1}$ for China in the 1990s (Table 3). Similarly, the C sequestration estimate was 113 Tg $year^{-1}$ for the 2000s, owing to 2.1 kg N ha^{-1} $year^{-1}$ more N input and expansion of the forest area (Table 3). On the basis of forest inventory data it was estimated that forest biomass absorbed 92 Tg C $year^{-1}$ in China from 1994 to 2003 (Fang et al. 2007; Piao et al. 2009). Thus, this observed C sequestration could almost alone be explained by N deposition (with forest expansion accounting for ~30 Tg $year^{-1}$ of the calculated change from the 1990s to the 2000s, Table 3). It should be noted that the greatest uncertainty in the calculations, however, is the C response to N deposition for Chinese forests. Most likely this C sensitivity to N deposition will be lower than 25, because the observed forest growth increase in China (92 Tg C $year^{-1}$) is probably caused by several factors, for example improved forest management, climate change, and CO_2-fertilisation, and not alone by N deposition as suggested by our approximate calculation (Table 3).

Following the evidence reviewed by de Vries et al. (2009) an additional 5–35 kg C per kg N could be sequestered in the soil. A soil organic C stock increase of 540–690 kg C ha^{-1} $year^{-1}$ was observed over two decades at a site in south China (Zhou et al. 2006) with a throughfall DIN input at 35 kg N ha^{-1} $year^{-1}$ (Fang et al. 2008). If this soil C increase is caused by the elevated N alone, the soil C sensitivity is ~15 kg C per kg N. Using this rate, Chinese forests roughly sequester an additional 70 Tg C $year^{-1}$ in soil organic matter. Altogether N induced C sequestration in China could be of the order of 100–200 Tg C $year^{-1}$. Considering this large potential effect of N deposition on C sequestration, much more detailed studies are urgently required in order to explore the effect of N deposition on C sequestration of Chinese forests.

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