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Human footprints in urban forests: implication of nitrogen deposition for nitrogen and carbon storage

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Abstract

Purpose Rising levels of nitrogen (N) deposition are influencing urban forest carbon (C) and N dynamics due to greater human disturbance compared to those in rural areas. N deposition in combination with increased atmospheric carbon dioxide (CO_2) and water limitation may alter C and N storage in urban forests. This review aimed to provide a better understanding of N and C storage under N deposition scenarios in urban forests.

Results and discussion Globally, fuel combustion and biomass burning contribute in approximately 70 and 16 % of the NO_x emission respectively. It is also estimated that NH_y and NO_x are two to four times higher in urban forests compared to rural areas. However, higher N deposition may not always result in increased N and C storage in urban forests. In fact, urban forests may even show early symptoms of N and C losses under climate change. For example, urban forests in fire-prone areas require higher frequency of burning to reduce the threat of wildfires, leading to an acceleration of C and N loss. Additionally, chronic N deposition may result in an early N loss in urban forests due to faster N saturation and soil acidification in urban forests compared to rural forests. Studies of N deposition on urban forests using N isotope composition ($\delta^{15}N$) also showed that N loss from urban forests can occur through the direct leaching of the deposited NO_3^--N. We also noted that using different ^{15}N signal of soil and plant in combination of tree ring $\delta^{15}N$ may provide a better understanding of N movement in urban forests.

Conclusions Although urban forests may become a source of C and N faster than rural forests, N-limited urban forests may benefit from N deposition to retain both N and C stocks longer than non-N-limited urban forests. Appropriate management practices may also help to delay such symptoms; however, the main source of emission still needs to be managed to reduce both N deposition and rising atmospheric CO_2 in urban forests. Otherwise, the N and C stocks in urban forests may further decline when prolonged drought conditions under global climate change increase the frequency of fires and reduce plant photosynthesis.

Keywords Fires · Fossil fuel combustion · Drought · Global climate change · Nitrogen deposition · Nitrogen isotope composition

1 Introduction

Human footprint in urban forests includes creating heat islands and greater nitrous oxide (N_2O), carbon dioxide (CO_2) and other greenhouse gas emissions (Vasishth 2006; Grimm et al. 2008). However, nitrogen (N) deposition and rising CO_2 concentration remain the two most important human-induced disturbances in urban areas. N is one of the crucial elements underpinning function and productivity of forests (Gundersen et al. 1998; Michalzik et al. 2001). In a

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healthy forest, N inputs and outputs need to be balanced, but this state of equilibrium is under threat due to increased atmospheric N deposition caused by increasing agricultural activity, fertilisation, biomass burning and fossil fuel combustion (Krupa 2003; Gruber and Galloway 2008; Fang et al. 2011a). Anthropogenic N deposition is expected to increase between 50 and 100 % by 2030 compared with that in 2000 (Reay et al. 2008), threatening the equilibrium of biogeochemical cycles. Nitrogen dynamics have been closely coupled with carbon (C) cycling (Gruber and Galloway 2008; Reverchon et al. 2012; Bai et al. 2012, 2013; Shen et al. 2014; Wang et al. 2014), and different studies suggest that enhanced N availability results in increased C sequestration in both plant biomass and soil (Hogberg 2007; Xu et al. 2009). It is estimated that for each kilogram of N, between 35 and 65 kg C is sequestrated (Liu and Greaver 2009). However, it has also been argued that N deposition may not always result in increased plant growth due to alteration of element stoichiometry (such as N/P ratio) and N exportation from the system through volatilisation, denitrification and leaching (Elser et al. 2010; Sun et al. 2010; Huang et al. 2012; Xu et al. 2013; Wang et al. 2015). For example, N leaching can be initiated when N deposition rates reach as little as 10 kg N ha^{-1} $year^{-1}$ in soils with low pH and intermediate C/N ratio (MacDonald et al. 2002). However, N leaching may commence with N input of 25–30 kg N ha^{-1} $year^{-1}$ in soils with low pH (MacDonald et al. 2002). Prolonged N leaching will result in N scarcity affecting tree growth (Law 2013) and even species decline or composition change (Emmett 2007).

Of all the forest ecosystems, urban forests may be more vulnerable to the consequences of N deposition because such forests are highly exposed to anthropogenic N deposition, including frequently controlled burning and fossil fuel combustion. For instance, the concentration of NO_3 in dust in urban areas has been shown to be double that of the rural areas (Lovett et al. 2000). Another study undertaken in USA showed that N deposition may be up to 40 % more in urban and suburban areas than in lesser populated areas (Bettez and Groffman 2013) as also presented in Fig. 1. Considering that most N is deposited within a short distance from where it was generated (4 to 45 km) (Lovett et al. 2000; Krupa 2003), the localised effects of N deposition on urban forests might be significant in extended periods of time. This review paper aimed to explore the effects of N deposition on urban forests to improve our understanding of the consequences of frequent human disturbances on C and N storage. In the current review paper, we summarised and synthesised (1) atmospheric N species with their potential sources and (2) the implications of N deposition on N and C storage in urban forests.

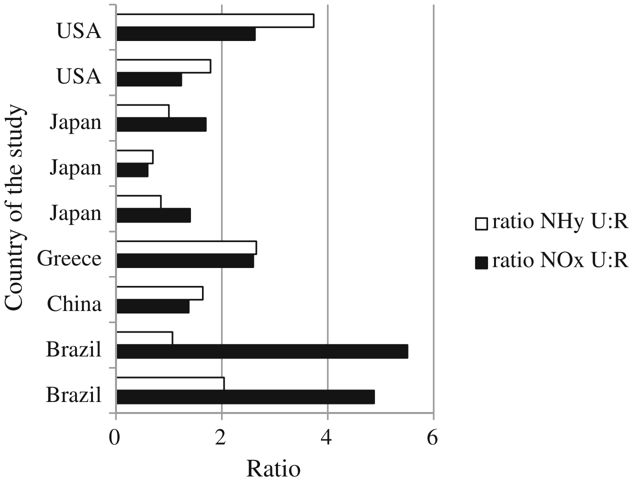

Fig. 1 The ratio of NH_y and NO_x in urban forests (U) to rural forests (R) in different study areas. The data of NH_y and NO_x deposition were derived from Lovett et al. (2000), Michopoulos et al. (2004), Forti et al. (2005), Aikawa et al. (2006) and Fang et al. (2011a)

2 Atmospheric N species, emission sources and deposition

Nitrogen deposition is defined as reactive N transferred from one system to another, which may consist of different components including reduced forms of N (NH_3 and NH_4^+), oxidised forms of N (NO_x, HNO_3, N_2O and NO_3^-) and organic components (Bobbink et al. 2010). NH_3 or NH_4^+ is considered to be mainly produced by the agricultural sector (e.g. fertilisation and biomass burning), and animals, whereas NO_x originates mainly from fossil fuel combustion and biomass burning (Mphepya et al. 2004, 2006; Allen et al. 2011; Paulot et al. 2013). Despite the fact that the origin of organic N is not very well known, it is acknowledged that organic N may originate from both natural and anthropogenic sources (Bobbink et al. 2010). Organic N consists of urea, amines and protein components, and its deposition has been reported from various regions (Singh et al. 2001; Fischer et al. 2002; Neff et al. 2002; Bobbink et al. 2010). Basically, different atmospheric N species have been reported in different studies, and dominance of one N species over the others can be associated with the site characteristics (e.g. savannahs, urbanised, industrial or agricultural areas), season of sample collection, distance from ocean and vegetation type (Mphepya et al. 2006; Allen et al. 2011; Fang et al. 2011a; von Glasow et al. 2013).

Sources of N emission have been extensively studied over the last decades (Pearl and Whitall 1999; Zhang et al. 2008; Paulot et al. 2013). Despite the fact that emission sources may vary significantly with the regions, emission sources can be categorised into two main sectors, human activities (e.g. fossil fuel combustion, biomass burning, domestic fires, atmospheric dust due to constructions and demolishment, livestock and inorganic N fertilisation) and natural sources (e.g. soil and oceanic emissions due to biotic or abiotic phenomena, lightening and wildfires). However, in urbanised areas, human activities are considered to be the dominant emission sources of N, and fires, including control burnings, wildfires and domestic fires, are one of main influential factor (Table 1). In fire-prone countries, management plans have been placed to control wild and un-planned fires, particularly in urban forests, resulting in reduced frequency of wildfires (Guinto et al. 1999). Whilst fuel combustion contributes to approximately 70 % of N emission globally, followed by biomass burning (16 % of global emission), biomass burning remains the dominant source of emission in fire-prone areas (Table 1). In fire-prone countries, the contribution of NO_x emissions due to burning varies between 30 and 75 % of the annual total N emission in that specific region (Table 1). Therefore, management plans need to be implemented to control wild and un-planned fires. The frequency of fires in fire-prone areas, such as Australia, can be as little as 3 years (Russell-Smith et al. 2007), and the chance of wildfire occurrences can be doubled when the area has not been burnt for more than 6 years (Boer et al. 2009). Wildfires result in larger C and N loss compared to prescribed burning because of their greater fire intensity and expansion (Certini 2005; Homann et al. 2011). Wiedinmyer and Hurteau (2010) found that controlled burning can reduce C emissions from forests compared to wildfires; however, they acknowledged that a very high frequency of controlled burning or large areas of burning may increase the forest C emissions. We acknowledge that the previous study was focused on C emissions, but it may have similar implications on N emissions. Therefore, fire intensity, fire frequency, and the extent of the area exposed to fires are likely to determine the extent of the N emissions.

In urban areas, in addition to fires, fuel combustion is another important source of N and accounts for 70 % of total global emission (Table 1). The reported concentrations of NH_y and NO_x in different forms of deposition were used to estimate the ratio of NH_y and NO_x in urban forests versus rural forests (Fig. 1). Both NH_y and NO_x were two to four times higher in urban forests than those of the rural areas. In urban forests, NO_x was the dominant N species. Higher population, anthropogenic inputs, coal-fired power stations, aerosol and gas concentration in urban areas compared to rural areas were accounted for increased emissions, thereby leading to increased deposition of NH_y and NO_x in the urban areas (Lovett et al. 2000; Michopoulos et al. 2004; Forti et al. 2005; Aikawa et al. 2006; Ferguson 2009; Fang et al. 2011a). It has been suggested that the critical threshold for N deposition to change forest dynamics is between 20 and 30 kg N ha^{-1} year^{-1} (Grennfelt and Thörnelöf 1992), and more recently, the threshold of N deposition has been estimated to be as little as 8 kg N ha^{-1} year^{-1} (Fleischer et al. 2013). Fleischer et al. (2013) showed that photosynthesis and C sequestration in forests can be evened out by increased transpiration when N deposition reaches 8 kg N ha^{-1} year^{-1}. The gap between 8 and 20 kg N ha^{-1} year^{-1} is large, and the estimation of N deposition threshold to trigger major changes in forest may have been underestimated; this is an issue which requires to be addressed in future studies.

3 The implication of N deposition on N dynamics

Since most ecosystems are N-limited, especially in boreal and temperate regions, increased N availability due to N

Table 1 Annual emission of NO_x from biomass burning, fuel combustion and other sources

	Annual emission (Tg NO_x year^{-1})	Global emission (%)[a]	Partitioning of NO_x sources at each region		
			Biomass burning (%)	Fuel combustion (%)	Other sources (%)
South Africa	0.36	0.98	8.33	75.0	16.6
Japan	0.54	1.47	0.00	98.1	1.85
Australia	1.0	2.73	33.0	31.0	36.0
Middle east	1.1	3.01	2.72	84.5	12.7
Central America	1.7	4.65	21.7	58.8	19.4
North equatorial Africa	2.2	6.02	35.0	18.1	46.8
South equatorial Africa	2.4	6.57	75.0	7.50	17.5
South America	2.7	7.39	32.2	37.0	30.7
SE Asia and India	4.9	13.4	22.4	65.3	12.2
East Asia	5.3	14.5	2.45	90.5	6.98
Europe	5.5	15.1	1.63	89.0	9.27
United states	6.4	17.5	1.56	98.4	0.04
Global	36.5	100	16.1	69.5	14.2

[a] Contribution of each area into the global emission (%). Data are adapted from Jaeglé et al. (2005)

deposition is likely to alter N cycling rates and losses from the system (Vitousek et al. 1997). Chronic long-term N deposition on urban forests may stimulate N loss due to (a) faster N cycling leading to an early N saturation and acidification in urban forests compared to rural forests (Zhu and Carreiro 2004; Fang et al. 2011a), (b) greater tendency of NH_4 retention in forest canopy than that of NO_3 (Piirainen et al. 1998), and (c) high tendency of deposited NO_3 to be leached directly (Templer and McCann 2010; Curtis et al. 2011; Rao et al. 2014). In this section, we will address the impact of N deposition on N mineralisation, nitrification and leaching and will discuss the use of N stable isotope composition as a tool to study N deposition effects on N dynamics.

As a major driver of forest soil acidification, chronic N deposition has been shown to contribute to the accumulation of NH_4^+ ions (Phoenix et al. 2012) and stimulation of N mineralisation, leading to early N saturation of forest ecosystems. As shown in Fig. 1, N saturation is likely to occur sooner in urban forests than in rural ones due to higher rates of N deposition. The changes in soil pH induced by high N deposition rates will impact microbial activity and composition, which in turn may increase N mineralisation and nitrification rates (Galloway and Cowling 2002; Zhu and Carreiro 2004; Forti et al. 2005; Emmett 2007; Chen et al. 2010; Cusack 2013), thereby reaching N saturation faster. Increased nitrification rates following N deposition have been reported in several studies (Galloway et al. 2003; Isobe et al. 2012). Through soil acidification, N deposition can also change the relative proportion of nitrifying microorganisms, with an increase in the ratio of archeal/bacterial ammonia oxidisers (Schmidt et al. 2007; Nicol et al. 2008). In addition, in the process of N saturation, emissions of NO and N_2O may increase because of the enhancement of nitrification or denitrification processes (Davidson et al. 2000), especially where soils are imperfectly drained.

However, after saturation, a decline in microbial N immobilisation and an increase in nitrification through soil pH changes (Tietema and Verstraten 1991; Shen et al. 2014) may enhance N losses through leaching, mainly in nitrates (Dise and Wright 1995; Fang et al. 2011b). Although the buffering capacity of soil decreases as a result of chronic N deposition, the capacity of forests to retain N may additionally be influenced by forest age, the period of exposure to N deposition and initial N status (Fang et al. 2009; Lu et al. 2009). For example, in China, three different forests, including mature forest, pine and mixed species forests, were exposed to N application and both acidification and N loss occurred faster in the mature forest compared to those of the other two forests (Lu et al. 2009). There was less demand for N uptake in the mature forest, and the pine and mixed species forests were N limited (Lu et al. 2009). Although it seems that N loss through leaching is inevitable under N deposition, urban forests with mature vegetation may be even more vulnerable to N deposition leading to an early N imbalance in their systems.

Early N loss is not always associated with forest N saturation and acidification. An early N loss in urban forests compared to rural forest has been observed to be associated with direct leaching of deposited NO_3 (Templer and McCann 2010; Curtis et al. 2011; Rao et al. 2014). The recovery of N addition to a moorland system was examined using ^{15}N tracers, and no recovery of ^{15}N in plant and soil was observed suggesting that N loss occurred more likely through directly deposited NO_3 (Curtis et al. 2005). Additionally, one of the main pathways to retain N is plant N uptake (Emmett 2007; Bradley 2001). However, some plant species (e.g. conifers) may prefer to take up NH_4^+-N rather than NO_3^--N (Piirainen et al. 1998). Considering that the majority of N deposition in urban forests is NO_x, such urban forests are thus more prone to N loss through NO_3 leaching even before reaching saturation or acidification point.

3.1 Using N isotope composition ($\delta^{15}N$) to study N deposition in urban forest

Traditionally, soil C/N ratio has been used to study N cycling in different systems, and it has been suggested that low C/N ratios (<27) may indicate a N saturation in forest floor (Matson et al. 2002; MacDonald et al. 2002; Aber et al. 2003). However, it has been shown that, sometimes, C/N ratios may not provide robust insights with respect to N deposition because part of deposited N may not be translated in soil C/N ratios (Aber et al. 2003; Emmett 2007). The main reasons are (a) stimulation of plant growth, (b) alteration in favoured N forms for plant uptake and (c) delay between N deposition and changes in soil C/N ratios (Emmett 2007). C/N ratio is also not a good indicator when the turnover of organic matter is fast or the organic matter layer is very insignificant (Fang et al. 2011a) or when deposited N is directly leached (Templer and McCann 2010).

Different ecosystem ^{15}N signals have been used as a useful indicator of N cycling in urban forests (Fang et al. 2011a; Bai et al. 2012, 2013; Falxa-Raymond et al. 2014; Wang et al. 2015). Enriched $\delta^{15}N$ systems suggest an open N cycling with high microbial activity and possibly high N loss from the system when lighter N tends to be lost (Nadelhoffer and Fry 1994; Wang et al. 2015). Even microbial activities involved in N transformations discriminate against heavier N, and their products are $\delta^{15}N$-depleted and hence vulnerable to loss (Nadelhoffer and Fry 1994). Under such conditions, plants would assimilate $\delta^{15}N$-enriched NH_4^+-N and NO_3^--N from soil leading to foliar $\delta^{15}N$ enrichment (Fang et al. 2011a; Falxa-Raymond et al. 2014). It has been shown that soil available N and foliar $\delta^{15}N$ are positively correlated and foliar $\delta^{15}N$ reflects the N status of soil (Ibell et al. 2013a, b, 2014). Thus, both soil and foliar $\delta^{15}N$ enrichment may be used to investigate N saturation and loss in the urban forest.

However, foliar $\delta^{15}N$ enrichment under N saturation and N loss may not always be observed in urban forests. For example, Fang et al. (2011a) revealed that N deposition did not lead to increased foliar $\delta^{15}N$ and also reported a negative correlation between foliar $\delta^{15}N$ and soil nitrification. Different factors may contribute to such observations. In urban forests, NO_x is the main dominant species as shown in different studies (Lovett et al. 2000; Michopoulos et al. 2004; Forti et al. 2005; Aikawa et al. 2006; Ferguson 2009). Therefore, plants may increase their uptake of deposited NO_3^--N, which is significantly depleted in ^{15}N, over that of NH_4^+-N, particularly when NH_4^+-N is scarce (Takebayashi et al. 2010; Fang et al. 2011a). Direct uptake of NH_4^+ by leaves may also influence foliar $\delta^{15}N$ because volatilised NH_4^+ is ^{15}N depleted (Fang et al. 2011a). Foliar $\delta^{15}N$ also provides spatial and temporal information of N cycling in a system (Craine et al. 2009) which may cause an over- or under-estimation of N cycling rates when samples have been collected once or in a short period of time.

More recently, tree ring $\delta^{15}N$ has been used as an indicator of long-term N dynamics (Elhani et al. 2005; Sun et al. 2010; Hietz et al. 2011). Despite the fact that tree ring $\delta^{15}N$ may provide long-term information with respect to N dynamics, it is also subjected to constrains mainly due to translocation of N among adjacent tree rings and differences in the source of N in the soil (Elhani et al. 2005; Hietz et al. 2011). However, it has been argued that tree ring $\delta^{15}N$ still offers valuable information of N dynamics in the past although this method may need to be adapted regionally (Poulson et al. 1995; Zuidema et al. 2013). We also believe that tree ring $\delta^{15}N$ may provide more reliable information of N dynamics than that of tree leaves due to the fact that life span of tree leaves is shorter than that of tree rings.

To evaluate NO_3^--N loss from the forest, $\delta^{15}N$ of NO_3^--N may also provide valuable information with respect to the movement of deposited NO_x in urban forest. Rao et al. (2014) studied $\delta^{15}N$ of NO_3^--N in both urban and rural forests and discovered that the majority of leached NO_3^--N was sourced directly from deposited NO_3^--N rather than N loss after microbial transformation and saturation. However, it might be sometimes difficult to differentiate the source of $\delta^{15}N$ in leached NO_3^--N. It has been suggested that using both $\delta^{15}N$ and $\delta^{18}O$ provides robust and reliable information to identify the source of NO_3^--N because $\delta^{18}O$ of deposited NO_3^--N significantly differs from other sources (Templer and McCann 2010; Curtis et al. 2011). Thus, whilst foliar $\delta^{15}N$ and tree ring $\delta^{15}N$ may provide valuable information with respect to urban forest N cycling and dynamics, the combination of $\delta^{15}N$ and $\delta^{18}O$ of NO_3^--N may provide better insights into the N retention capacity of the forest compared to foliar and tree ring $\delta^{15}N$.

4 The implication of N deposition on forest C storage

Forest C storage is coupled with N status and availability, but the effects of N on C storage are related to other factors, including atmospheric CO_2 and water status. In this section, we will discuss different scenarios, which have also been summarised in Fig. 2. First, implication of N deposition as a single factor on urban forest C storage may be an increased C sink in the short-term due to enhanced plant growth. Stimulation of plant growth and biomass accumulation in urban forests following N deposition may particularly occur in young urban forests or N-limited forests (Reich et al. 2006; Hyvönen et al. 2007) because N cycling and C cycling are closely coupled. N is a crucial component of the enzyme Rubisco, which facilitates CO_2 fixation at photosynthetic carboxylation site (Evans 1989). Increased atmospheric CO_2 may increase photosynthetic capacity leading to enhanced C accumulation in plant biomass (Xu et al. 2009; Reverchon et al. 2012; Xu et al. 2014). Therefore, N deposition may result in increased C fixation and biomass accumulation in forests under increased CO_2 concentration. In their study, Thomas et al. (2010) reported a global increase in tree C storage of 0.31 Pg C year^{-1} following N deposition. However, the effects of N deposition on forest C sink may differ in the long term. Despite reports of increased forest C sink over time, it has been shown that forest productivity may decline over time when trees reach their optimum growth rate and the canopy closes (Pregitzer and Euskirchen 2004). Other reports indicate that in the long term, increased N deposition may limit tree growth and the C storage capacity of forests because N deposition acidifies soil, facilitates soil cation loss, shifts N limitation to P limitation and changes plant diversity (Oren et al. 2001; Phoenix et al. 2006; Huang et al. 2012). These changes in the soil environment may further influence soil microbial extracellular enzymes governing decomposition rates of organic compounds, which partly control soil C storage (Waldrop et al. 2004). Additionally, although increased CO_2 first improves photosynthesis and C sequestration, photosynthesis may reach a threshold in the long term and respiration may also increase (Hyvönen et al. 2007). Thus, it is uncertain to what extent and how long interactive N deposition with rising CO_2 increases C sink in urban forests.

We also argue that other limitations, particularly water scarcity, will influence the interactive effects of both N and CO_2 deposition because water is also one of the driving factors for both soil microbial activity and plant photosynthetic capacity (Fig. 2). Water scarcity limits photosynthetic capacity due to increased stomatal closure of plants and also increases tree mortality (Bréda et al. 2006). Additionally, the rising occurrence of drought-induced fires in the future is likely to enhance tree mortality and decrease rainfall through a reduction of water fluxes (Nepstad et al. 2001). All factors together, in the long term, may suggest (a) faster N loss in urban forests

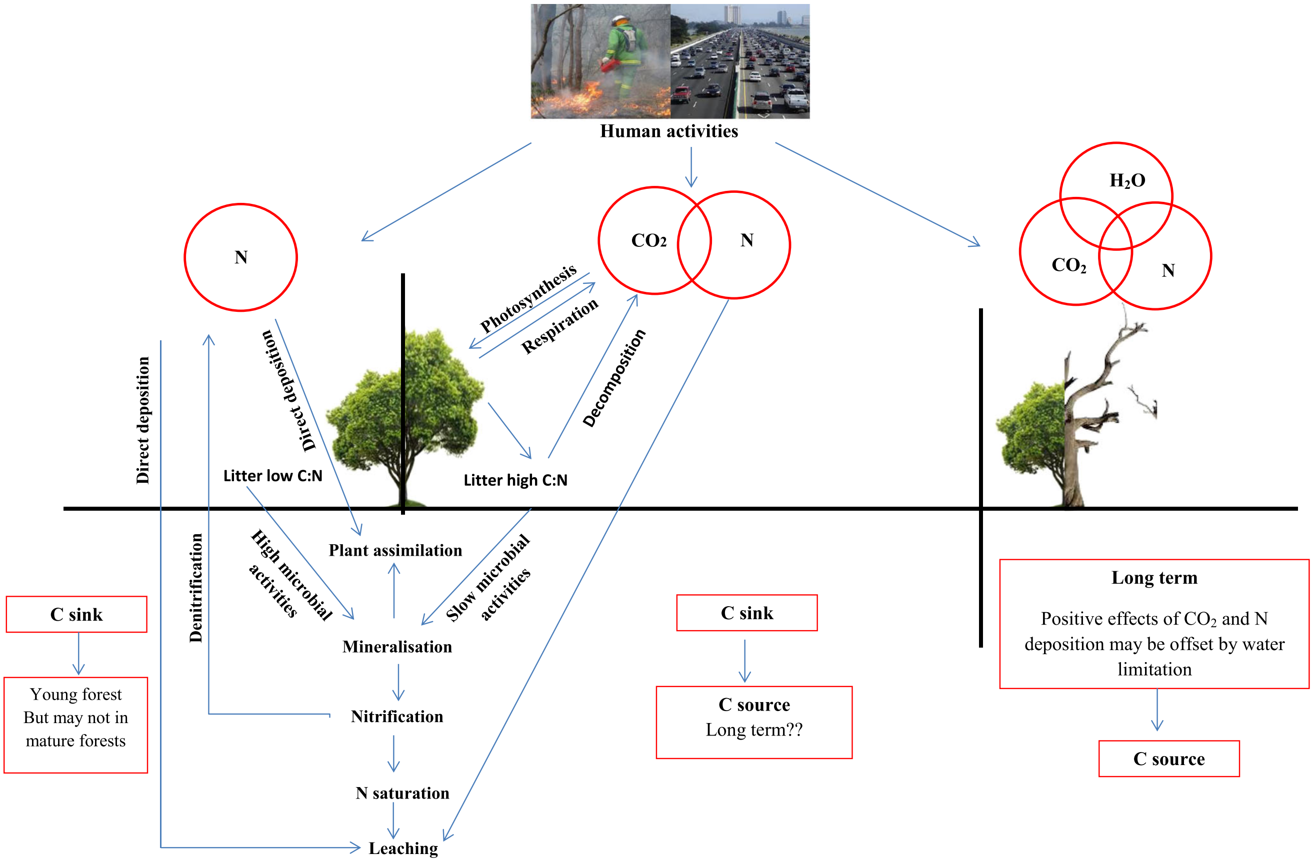

Fig. 2 Schematic model of C storage in urban forests with respect to nitrogen deposition, rising atmospheric CO_2 and water limitation

due to N saturation and (b) early alteration of urban forests from C sink to source when compared to rural forests (Fig. 2).

5 Challenges to retain N in urban forests

Different strategies have been suggested to improve N retention in forests, including decreased nitrification, increased immobilisation of NO_3^--N, NO_3^--N reduction to ammonia, encouraged plant uptake and vegetation management, as presented in a vast array of studies in different forest ecosystems (Aulakh et al. 2000; Davidson et al. 2003; Micks et al. 2004; Templer and McCann 2010; Xu et al. 2013). We think that the same principle can also be applied in urban forests, but the application of such methods may become challenging due to the cost or need for more frequent fires in fire-prone forests. In this section, we review the suggestions to improve the N retention in urban forests.

One of the main suggestions is to build up the soil organic layer of urban forests to improve forest N retention. Recently, this suggestion has received more attention when it has been shown that the deposited NO_3^--N can be directly leached and stimulation of NO_3^--N immobilisation in urban forests can be crucial to prevent direct loss of NO_3^--N. Soil organic layer can immobilise NO_3^--N through abiotic immobilisation within a very short period of time (less than 30 min) (Davidson et al. 2003; Micks et al. 2004). Abiotic immobilisation of NO_3^- occurs as a result of altering nitrate to nitrite in the presence of reactive Fe (II), known as ferrous wheel hypothesis (Davidson et al. 2003). Afterwards, nitrite reacts with dissolved organic C to produce dissolved organic N (Davidson et al. 2003). Dissolved organic N can be rapidly stabilised or immobilised in organic layer of the soil (Micks et al. 2004; Lewis et al. 2014). Improving organic layer of soil in fire-prone urban forests may become challenging where frequent burning is applied to decrease the risk of wildfires. However, some landscapes are ecologically fire dependent to regenerate and remain productive (Vasishth 2008). Therefore, proper strategies are required to be considered in fire-prone urban forests to alleviate wildfires but still maintain the ecological needs of the forest. Lime application in forests has also been shown to contribute to N retention through reduced N mineralisation leading to decreased litter decomposition rates (Melvin et al. 2013). However, Balaria et al. (2014) showed that liming did not reduce litter decomposition in a study undertaken in acidic forest of North USA and concluded that higher lime application rates were required or that the effect of liming did not last long. Nevertheless, large-scale lime applications at high rates are not cost-effective (Bostedt et al. 2010). Biochar, which is a stable C-rich material derived from pyrolysis of organic matter feedstock, may also be applied to alleviate acidification and prevent NO_3^--N leaching from the system (Clough and Condron 2010). Biochar has been shown to improve N retention in soil through different mechanisms including immobilisation and increased resident time of NO_3^--N in different ecosystems (Clough and Condron 2010; Van Zwieten et al. 2010; Bruun et al. 2012). However, DeLuca et al. (2006) found that charcoal in forest soil resulted in nitrification stimulation rather than NO_3^--N immobilisation which may reduce the capacity of biochar to retain N in forest soil. Despite wide indications of N retention when biochar is applied in different systems (Clough et al. 2013; Reverchon et al. 2014; Bai et al. 2015; Reverchon et al. 2015; Xu et al. 2015), the economic viability of biochar application in forests would also need a careful evaluation (Joseph et al. 2013).

6 Conclusions

N deposition and rising atmospheric CO_2 have been shown to enhance C and N storage in forests in the short term, but these effects may not last and forests may become C and N sources rather than sinks. Urban forests may even show earlier symptoms of N and C losses under climate change compared to rural forests. Such early symptoms may be delayed in young or N-limited urban forests, but not in mature or non-N-limited urban forests. Water limitation may also further inhibit urban forest C and N retention and lead to an acceleration of C and N losses from the system. Appropriate management practices may decrease nitrification, increase NO_3^--N immobilisation, reduced ammonia production, and encourage plant uptake, leading to slow down the appearance of such symptoms. However, the main source of emissions needs to be managed to reduce both N deposition and rising atmospheric CO_2 in urban forests. Otherwise, retaining N and C in urban forests will remain challenging.

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