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INVITED REVIEW

Greenhouse gas emissions in response to nitrogen fertilization in managed forest ecosystems

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Abstract Nitrogen (N) fertilizer use in managed forest ecosystems is increasing in the United States and worldwide to enhance social, economical and environmental services. However, the effects of N-fertilization on production and consumption of greenhouse gases (GHGs), especially carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) in managed forest ecosystems are poorly understood, unlike in agriculture where effects are well documented. Therefore, a review of the available literature was conducted to comprehend the effects of N-fertilization on CO_2, CH_4 and N_2O emissions in managed forest ecosystems to summarize sources, sinks, and controlling factors, as well as potential mitigation strategies and research gaps to reduce GHG emissions. This review clearly identifies the importance of N-fertilizer management practices on CO_2, CH_4 and N_2O emissions. Potential N management practices to mitigate GHG emissions in managed forest ecosystems include improving N uptake efficiency, identifying and managing spatial variation in soil fertility, using the right fertilizer source at the right time, adopting appropriate methods of N-fertilizer application, and introducing nitrification/denitrification inhibitors. Nitrogen-fertilizer response is affected by soil physical (e.g., moisture, drainage, bulk density, and texture), chemical (e.g., nutrient availability, labile carbon, soil pH, and C/N ratio) and local climatic factors (e.g., temperature, relative humidity, and rainfall). Therefore, the interactions of these factors on GHG emissions need to be considered while evaluating N-fertilizer management practices. Existing studies are often limited, focusing primarily on temperate forest ecosystems, lacking estimation of net emissions considering all three predominant soil-derived GHGs, and were often conducted on a small scale, making upscaling challenging. Therefore, large-scale studies conducted in diverse climates, evaluating cumulative net emissions, are needed to better understand N-fertilization effects on GHG emissions and develop mitigation strategies. Mitigation strategies and research gaps have also been identified, which require the collaborative efforts of forest

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owners, managers, and scientists to increase adoption of N-fertilization best management practices and understand the importance of N-fertilizer management strategies in reducing emissions and enhancing the net GHG sink potential for managed forest ecosystems.

Keywords Carbon dioxide · Methane · Nitrous oxide · Biosphere–atmosphere exchange · Soil gas flux

Introduction

Greenhouse gases (GHGs) affect Earth's climate by absorbing solar infrared radiation in the lower atmosphere. Concentrations of atmospheric GHGs, specifically carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), drive large scale and long-term climate patterns that have both environmental and economic implications. Although GHGs occur naturally in the atmosphere, anthropogenic activities have elevated their concentrations. For example, the global atmospheric concentration of CO_2 increased from a pre-industrial (before 1750) value of 280–379 ppm in 2005, which spiked to above 400 ppm several times in April of 2013 (IPCC 2007a; Monastersky 2013). Similarly, pre-industrial global atmospheric concentrations of N_2O increased from 270 to 319 ppb and CH_4 from 715 ppb to 1,774 ppb by 2005 (IPCC 2007a). The global warming potentials (GWPs, over a 100-year time frame) for N_2O and CH_4 are 310 and 21 times higher than that of CO_2, respectively (IPCC 2007b).

Forests have received much interest in mitigating climate change (McKinley et al. 2011). They cover about 30 % of the global land area and consequently can play a significant role in regulating global climate through their capacity to be a significant sink for CO_2 and CH_4 as well as a source for N_2O (Pilegaard et al. 2006; Smith et al. 2003). The world's forests absorb 8.8 Pg of CO_2 each year with an average sequestration rate of 3.81 Mg CO_2 ha^{-1} $year^{-1}$. The current carbon (C) stock in the world's forests is estimated at 861 ± 66 Pg, with approximately 44 % in soil (to 1 m depth), 42 % in live biomass (above and below ground), 8 % in deadwood, and 5 % in litter (Pan et al. 2011).

In this context, forest ecosystems in the U.S. can play a critical role in the global C cycle because forested areas are estimated to cover 323 million ha and represent a high capacity for C storage (152 Pg of CO_2) in biomass and soil (USDA Forest Service 2010; FIDO 2011). This has a major impact on regional and global sources of CO_2, absorbing as much as 25–50 % of CO_2 emitted annually from fossil-fuel combustion in the US (US Climate Change Science Program 2008). Forests of the US (including vegetation, soils, and harvested wood) accounted for approximately 86 % of total 2010 net CO_2 flux (USEPA 2012). Land use, land-use change and forestry activities between 1990 and 2010 resulted in a net C sequestration of 1.07 Pg CO_2 Eq in the US (USEPA 2012). This is a 22 % increase from 1990, primarily due to an increase in the rate of net C accumulation in forest C stocks including aboveground and belowground tree biomass and harvested wood pools. This sequestration represents an offset of 19 % of CO_2 emissions and 16 % of total GHGs emissions (McKinley et al. 2011; USEPA 2012). This illustrates the potential for C-sequestration in managed forests and overall reduction of GHG emissions. For example, forest management activities such as replanting, thinning, fertilization, drainage, species or genotype selection, and optimizing rotation lengths can enhance C-sequestration (Liski et al. 2001; IPCC 2006).

Increasing use of N fertilizer in agriculture and managed forest ecosystems has caused a large perturbation to the global N cycle since the industrial revolution, thereby significantly increasing net N_2O emissions (Grip and Jansson 2012; Pinder et al. 2012). The IPCC has reported that 10 kg N_2O–N is emitted for every 1,000 kg of N-fertilizer applied in a managed system (IPCC 2006). Although managed forests have a large C sink potential, the impacts of N-fertilizer management on GHG emissions have not been well documented. Nitrogen added through fertilization, which is a common practice in managed pine plantations (Albaugh et al. 2007), has been shown to increase N_2O emissions (Papen et al. 2001; Brown et al. 2012), while simultaneously decreasing the sink strength of CH_4 (Sitaula and Bakken 1993; Mochizuki et al. 2012). Judicious application of N-fertilizer and use of newer fertilizer technologies could decrease future N_2O emissions without impacting productivity and other ecosystem services. Thus, optimizing N-management has a significant potential to reduce net GHG emissions from managed forest ecosystems, including the regeneration phase of forestry.

Extensive studies have been conducted on use of N-fertilizer and its impacts on growth in managed forest ecosystems (e.g., Elliot and Fox 2006; Fox et al. 2007; Haase et al. 2007; Fujinuma et al. 2011; Ring et al. 2011; Vallack et al. 2012). However, the effects of N-fertilization on production and consumption of GHGs (CO_2, CH_4, and N_2O) in managed forest ecosystems are poorly understood despite the fact that: (1) forest fertilization is used to stimulate tree growth, reduce rotation age, and sequester C, and (2) acreage under forest fertilization has increased from 0.8 million ha in 1990 to 4.9 million ha in 2004 in southeastern United States (Albaugh et al. 2007). This calls for a detailed review and synthesis of available studies on the impacts of N-fertilizer management on GHG emissions to strengthen the knowledge in this field and identify critical knowledge gaps. Therefore, the main objective of this paper is to review and synthesize literature related to inorganic N-fertilization (typically $\sim$200 kg N ha^{-1}) impacts on greenhouse gas emissions in managed forest ecosystems, and present the best current knowledge available and future research needs.

Carbon dioxide emissions in forest ecosystems

Source and sink of carbon-dioxide

Forests, especially in the northern hemisphere, are a significant sink of CO_2 (Ciais et al. 1995). Forest ecosystems in the US accounted for 86 % of total 2010 net CO_2 flux (USEPA 2012). They contribute to the C budget by releasing and removing CO_2 from the atmosphere (Schlesinger and Bernhardt 2013). Trees constantly remove CO_2 from the atmosphere through photosynthesis, which converts atmospheric C to organic C. Subsequently, CO_2 can leave forest ecosystems through respiration (both autotrophic and heterotrophic), the major mechanism for CO_2 flux from forest ecosystems (Fig. 1).

Nitrogen-fertilizer applications alter the emissions of CO_2 by regulating plant and microbial activities that are related to soil processes (e.g., N and C cycling) (Fig. 1). Especially in N limited forest ecosystems, N-addition plays a critical role in stimulating plant growth and consequently enhancing CO_2 uptake (Fleischer et al. 2013; Law 2013; Bowden et al. 2004). Usually, N fertilization increases tree growth resulting in an increase in the C sink and net C balance of forest ecosystems (LeBauer and Treseder 2008; Liu and Greaver 2009; Tyree et al. 2013). For example, Albaugh et al. (2012) reported that the application of 224 kg N ha^{-1} with 28 kg of P ha^{-1} sequestered 19.2 Mg ha^{-1} of CO_2 equivalent as additional stem growth. Increase in plant N content due to N-fertilization also

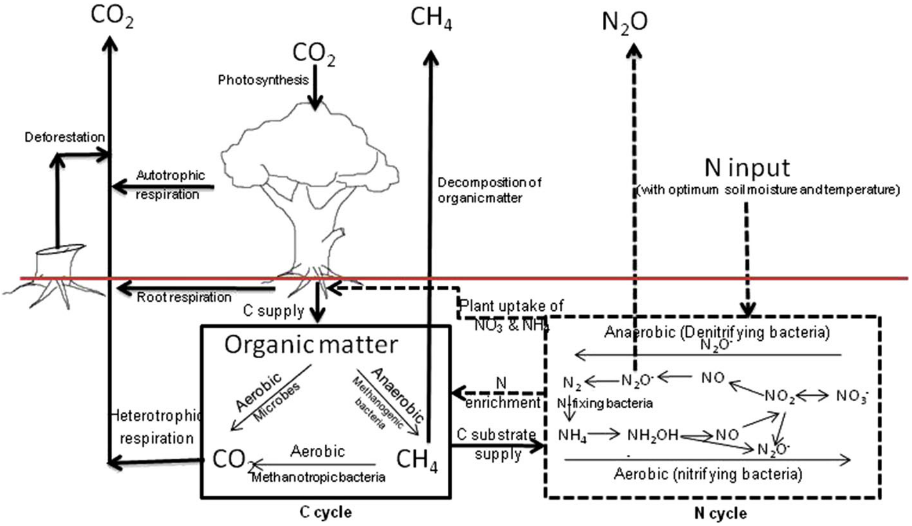

Fig. 1 Effects of nitrogen inputs on the processes that regulate the fluxes of N_2O, CO_2 and CH_4 in forest ecosystems (Conrad 1996; Schimel 2000; Thirukkumaran and Parkinson 2000; Philips and Fahey 2007; Liu and Greaver 2009; Canfield et al. 2010; Watanabe and Ortega 2011; Bernal et al. 2012; Brown et al. 2012)

enhances autotrophic respiration (Griffin et al. 1997; Reich et al. 2008) and litter decomposition (Berg and Laskowski 2006).

At early stages, in the first year, N application has shown to increase soil respiration in hardwood stands compared to pines due to increased productivity of the former. However after 13 years of continuous N-fertilization, soil respiration was observed to be suppressed by 41 % in both stands due to reduced microbial activity (Bowden et al. 2004). Mo et al. (2008) and Janssens et al. (2010) also reported decreases in soil microbial and root biomass due to N-fertilization resulting in a decrease in decomposition rates and increase in soil C sequestration. Application of N-fertilizer at the rate of 50 kg N ha^{-1} $year^{-1}$ to the tropical forest ecosystems of Puerto Rico with high background N availability did not change litter fall productivity. However, mineral-associated soil C concentrations increased significantly, although live fine roots and labile soil C fractions did not (Cusack et al. 2011). Nitrogen-fertilization can also lead to soil acidification, affecting soil microbial activity, and thereby litter decomposition and heterotrophic respiration (Tonon et al. 2010). Decrease in live fine roots and labile soil C fractions resulted in a decrease in CO_2 emissions in fertilized plots.

Nitrogen-fertilizer application has been reported to increase (Gough and Seiler 2004), decrease (Haynes and Gower 1995, Maier and Kress 2000), or have no effect (Lee and Jose 2003) on CO_2 emissions from the soil surface, owing to different controlling factors. Therefore, the effects of N fertilization on soil respiration are equivocal, showing varied responses on autotrophic (roots) and heterotrophic (soil organisms) components of soil respiration, the balance of which determines the overall change in soil respiration.

Controls of CO_2 emissions

Various factors, alone or in combination, can affect CO_2 emissions from forest soils. Carbon dioxide emissions from soils are controlled by (1) soil or litter-soil interface temperature (Butnor et al. 2003; Wang et al. 2003; Vose and Bolstad 2007; Peichl et al. 2010), (2) soil moisture (Bouma and Bryla 2000; Qi and Xu 2001), (3) the amount and quality of soil organic matter (Abbas and Fares 2009), (4) N-fertilization or tissue N content (Pregitzer et al. 1998; Maier and Kress 2000; Burton et al. 2002; Bowden et al. 2004; Philips and Fahey 2007; Jassal et al. 2010; Albaugh et al. 2012), and (5) C:N ratio (Klemedtsson et al. 2005).

The impacts of forest N-fertilization on CO_2 emissions have varied widely. Of the 12 studies of the effect of N-fertilization on soil CO_2 flux in managed forest, especially under high rates of fertilizer addition (e.g., ~ 150 kg N ha^{-1}), six studies reported that N-fertilization decreased soil CO_2 emissions by 15–46 % (Table 1). These studies do not indicate any strong relationship of CO_2 flux with climate or forest type, but in general pine forests have shown to decrease CO_2 emissions and hardwood to increase CO_2 emissions, especially in the first year of application. This increase in CO_2 emissions is attributed to increased productivity and microbial communities (Brumme and Beese 1992; Bowden et al. 2004). Nitrogen fertilization at 167 kg N ha^{-1} $year^{-1}$ of hardwood stands in New York and New Hampshire resulted in a decrease in both decomposition of SOM and heterotrophic (microbial) respiration in the rhizosphere (Philips and Fahey 2007). Similarly, Tyree (2005) reported lower heterotrophic respiration (produced during the decomposition of organic matter by soil organisms found within the soil profile) but increased autotrophic (root) respiration in fertilized loblolly pine ecosystems (*Pinus taeda* L.). Increases in the GHG sink due to N addition in forest ecosystems are mostly due to stimulated large CO_2 uptake from increased primary productivity (Liu and Greaver 2009), reduction in microbial activity and biomass (Fog 1988; Thirukkumaran and Parkinson 2000; Wallenstein et al. 2006), and decrease in root and/or mycorrhizal growth or respiration (Haynes and Gower 1995). Nitrogen fertilization can have the greatest influence on the sink potential in nutrient poor soils, as reported from studies conducted at eleven Swedish coniferous forests (Arnebrant et al. 1996).

In contrast, other studies have been shown to increase CO_2 emissions by 6–41 % after N-fertilizer application (Table 1). Increase in CO_2 emissions was observed in varied climatic conditions from temperate forests in Germany and the U.S. to subtropical forests of China, especially at an early stage of plantation (Table 1; Brumme and Beese 1992; Deng et al. 2010). Jassal et al. (2010) observed an increase in forest floor CO_2 emissions due to an increase in autotrophic (including rhizospheric) soil respiration after application of N-fertilizer until four months. Lastly, there were a few studies that reported no N-fertilizer effect on CO_2 emissions in pine plantations (Table 1; Castro et al. 1994; Maljanen et al. 2006). These observations may be attributed to acidic soil with pH ranging from 3.6 to 4.3, which might have limited the decomposition process and root activity, responsible for the CO_2 production from soil (Maljanen et al. 2006).

Responses to N fertilization are varied, but appear to have some trends when constrained by stand type, age class, soil properties, etc. More studies are required to better understand the variability within each of these settings, particularly where forest management can be used to mitigate GHG emissions.

Management practices to increase CO_2 sink

Our ability to manage forest resources sustainably is an important priority for the environment, ecosystem services and the economy. The use of long-lived wood and wood products as an energy source has been shown to be a low-cost C capture (Birdsey et al. 2000; Lippke et al. 2011). About 2 tonnes of wood produced can sequester 1 tonne of C from the

Table 1 Effect of nitrogen-fertilization on CO_2 emissions in managed forest ecosystems

Forest type	Location	Fertilizer rate (kg N ha^{-1} $year^{-1}$)	Fertilizer source	CO_2 emission increase (+) or decrease (−) due to fertilization over control (%)	References
Temperate-mixed hardwood stand and red pine	Massachusetts	150	NH_4NO_3	+41	Bowden et al. (2004)
Temperate-beech	Germany	140	NH_4SO_3	+16	Brumme and Beese (1992)
Subtropical-hardwood and pine	China	100	NH_4NO_3	+8.2	Deng et al. (2010)
Temperate-Douglas-fir	East coast of Vancouver Island, Canada	200	Urea	+6.0	Jassal et al. (2010)
Temperate-Hardwood forest	New York and New Hampshire	167	As NO_3	−16 to 46	Philips and Fahey (2007)
Temperate-Hardwood forest	England	81	Compound fertilizer	+7	Mojeremane et al. (2012)
Temperate deciduous: Black cherry-sugar maple	Pennsylvania	100	NH_4NO_3	−19.0	Bowden et al. (2000)
Mature tropical forest-evergreen broadleaf and pine	China	150	NH_4NO_3	−15.9	Mo et al. (2008)
Temperate-Red pine	Wisconsin	150	NH_4 and NO_3	−15.8	Haynes and Gower (1995)
Temperate-Hardwood-sugar maple	Michigan	30	$NaNO_3$	−15	Burton et al. (2004)
Humid subtropical-Loblolly pine	North Carolina	112	Urea	−14.5	Butnor et al. (2003)
Tropical-Slash pine plantation	Florida	180	Urea	No impact	Castro et al. (1994)
Boreal forest: mostly *Picea abies*	Finland	200	NH4NO3	No impact	Maljanen et al. (2006)

atmosphere, assuming 50 % C in wood (Karchesy and Koch 1979). Furthermore, life-cycle analyses reveal that the wood products used in construction store more C and use less fossil energy than steel, concrete, or brick, whose manufacture is energy intensive and produces substantial GHG emissions. Although wood products do not permanently sequester C from the atmosphere, they do sequester it for the life of the product, which can be >100 years (Lippke et al. 2011).

Another management factor that needs consideration is forest fertilization (Albaugh et al. 2012). Fertilizer application rates should be reduced where wood is used for paper production because of high life cycle GHG emissions and short half-life of paper products (Gan et al. 2012). Nitrogen-fertilizer application usually enhances C sequestration in forest ecosystems, ranging from 20 to 70 kg C sequestered per kg of atmospheric N deposited, with most C gains occurring in aboveground biomass (Shan et al. 2001; Hyvönen et al. 2008; Pinder et al. 2013). However, the impacts on belowground C are complex and varied, but usually small (Nave et al. 2009). Intensive management and fertilization of young forest plantations can convert forested ecosystems from a C source to a C sink by increasing C sequestration potential through enhanced primary productivity compared to natural forests (Maier and Kress 2000, Albaugh et al. 2012). For example, N-fertilization in temperate deciduous forests can reduce soil CO_2 emissions by 19 % (Table 1; Bowden et al. 2000), but must be considered in the context of other soil-derived GHG emissions.

Methane emissions in forest ecosystems

Source and sink of CH_4

Methane is the second most abundant GHG and is 21 times more potent than CO_2 (IPCC 2007b). Production of CH_4 by methanogenic archaea in wetlands is a major source while consumption by CH_4 oxidizing bacteria in upland soils is a major sink. The average annual CH_4 emission can be 10 times larger at undrained sites relative to drained (Arnold et al. 2005). Since 1750, atmospheric CH_4 concentrations increased by 8.5 ± 1.3 Gt CO_2-eq. year^{-1} (Montzka et al. 2011). Forest soils can act as both sources and sinks of CH_4. Well-drained upland forest soils are one of the most important global biological sinks for CH_4 (Smith et al. 2000; Dutaur and Verchot 2007; Megonigal and Guenther 2008), where methanotrophic bacteria assimilate CH_4 into the microbial biomass as organic C and oxidize it to CO_2 to gain energy (King 1997; Xu and Inubushi 2004; Dutaur and Verchot 2007; USEPA 2010). Methanotrophic bacteria play a vital role in GHG budgets, especially for forest ecosystems, because they are the only biological sink for CH_4. Although methanotrophic bacteria are the only organisms using CH_4, they also emit CO_2, usually in proportion to about 50 % of the CH_4 consumed. Methane sink strength of soils depends on oxidation by methanotrophic bacteria, which is influenced by environmental factors that control oxidation rates. Globally, forest soils are a net sink of atmospheric CH_4, with estimates from 1.8 to 11.8 Tg CH_4–C yr^{-1} (IPCC 2007b). Specifically, tropical forest soils contribute 28 % (6.2 Tg yr^{-1}) to the global CH_4 sink (Veldkamp et al. 2013). Methane consumption rates for aerobic temperate forest soils can be up to 3.17 mg CH_4–C m^{-2} day^{-1} (Steudler et al. 1989).

On the other hand, it has been demonstrated that flooded forest soils are CH_4 sources (McKenzie et al. 1998; Rice et al. 2010). Major sources of CH_4 include wetlands, ruminant animals, rice paddies, biomass burning and fossil fuel production (IPCC 2007b). Although CH_4 is emitted mostly in wetland soils, it can also be found in tropical and upland soils during high rainfall or wet seasons (Keller and Reiners 1994). For example, Frankenberg et al. (2005), using space-borne near-infrared absorption spectroscopy, mapped the global CH_4 distribution and observed unexpectedly high CH_4 concentrations over tropical rainforests, revealing that emission inventories considerably underestimated forest CH_4 sources. They reported that model simulation indicated a tropical CH_4 source of around 120 Tg year^{-1}. Another study by Crutzen et al. (2006) reported a CH_4 source of 78 Tg year^{-1} from tropical forests, again suggesting that CH_4 sources cover a wider array of soil types, biomes, and climates than previously thought.

Controls of CH_4 emissions

The major factors regulating production or consumption of CH_4 from soils are: (1) water-filled pore space (Adamsen and King 1993; Castro et al. 1995; Brumme and Borken 1999; Smith et al. 2000; Borken and Brumme 2009; Jassal et al. 2011; Aronson et al. 2012; Gundersen et al. 2012; Veldkamp et al. 2013), (2) N status or N-fertilization of soil (Steudler et al. 1989; Castro et al. 1995; Sitaula et al. 1995a; Smith et al. 2000; Papen et al. 2001; Fender et al. 2012; Gundersen et al. 2012; Mochizuki et al. 2012; Wood and Silver 2012), and (3) soil temperature (Castro et al. 1995; MacDonald et al. 1997; Smith et al. 2000; Veldkamp et al. 2013). Other factors shown to affect CH_4 production or consumption in forest soils include pH (Sitaula et al. 1995a; Sparks 1995; Brumme and Borken 1999; Smith et al. 2000; Xu and Inubushi 2004; Borken and Brumme 2009), bulk density (Smith et al. 2000; Teepe et al. 2004), soil texture (Boeckx et al. 1997), forest type (Hudgens and Yavitt 1997), and stand age (Gundersen et al. 2012).

One of the major factors influencing CH_4 oxidation is soil N status, which is often manipulated through forest N-fertilization. N limitation of CH_4 oxidation is common in both wetland as well as upland soils (Bodelier and Laanbroek 2004). The effects of forest N-fertilization on CH_4 production or consumption has been studied intensively (Table 2). Out of the 12 studies we found that focused on CH_4 responses to forest N-fertilization, nine were conducted in temperate forests and only three in tropical forests. Ten studies reported inhibitory effects of forest N-fertilization on CH_4 uptake, ranging from 5 to 95 % reduction, and two studies did not observe any effect. Inhibitory effects of N-fertilization on CH_4 oxidation have been observed in temperate coniferous forest (Castro et al. 1994; Schnell and King 1994; Sitaula et al. 1995a; Xu and Inubushi 2004), temperate deciduous forest (Chan et al. 2005), mixed deciduous woodland (Dobbie and Smith 1996), and tropical forests (Castro et al. 1994).

Zhang et al. (2008a), in a rehabilitated tropical mixed pine-broadleaf forest and a disturbed tropical pine forest, reported that N-fertilization had no inhibitory effect on CH_4 uptake. However, in a fertile, mature tropical broadleaf evergreen forest, N-fertilization at rates of 50, 100, and 150 kg N ha^{-1} inhibited soil CH_4 uptake by 6, 14, and 32 %, respectively (Zhang et al. 2008a). Thus, the response of N-fertilization to soil CH_4 uptake also depends on the land-use history and on the soil N status of the forest. However, an earlier study by Castro et al. (1995) reported that high-fertility sites can have 2–3 times greater CH_4 uptake rates than low-fertility sites, which may be due to CH_4 oxidizers being N-limited for growth, especially in unfertilized forest soils (Papen et al. 2001). This shows the variability of N-fertilization on the site productivity and capacity of land for CH_4 oxidation potential.

These impacts of forest N-fertilization on CH_4 consumption and production are controlled by various factors that affect N availability as depicted in Fig. 2. The main mechanism by which N fertilizers inhibits CH_4 oxidation, as well as toxic effects resulting from hydroxylamine and nitrite, is ammonia, either from ammonification or ammonium

Table 2 Effect of nitrogen-fertilization on CH_4 uptake in managed forest ecosystems

Forest type	Location	Fertilizer rate (kg N ha^{-1} $year^{-1}$)	Fertilizer Source	Reduction in CH_4 uptake (%)	References
Temperate deciduous	Pennsylvania	100	NH_4NO_3	24	Bowden et al. (2000)
Temperate-Douglas-fir	Vancouver Island, Canada	200	Urea	50	Jassal et al. (2011)
Temperate	Norway	30	NH_4NO_3	85	Sitaula et al. (1995b)
Scots pine		90		62	
Temperate old growth forest-Pine and broadleaf mixed	China	45	$(NH_4)_2$ SO_4 or NH_4CL	5–34	Xu et al. (2011)
Temperate deciduous forest-Black cherry-sugar maple	Pennsylvania	100	NH_4NO_3	24	Bowden et al. (2000)
Temperate forest-Red pine mixed with black oak red/maple	Massachusetts	120	NH_4NO_3	33	Steudler et al. (1989)
Temperate-Wet meadow (mostly *Carex scopulorum*) Dry meadow (mostly *Kobresia myosuroides*)	Colorado	100	Slow release urea	No effect 52	Neff et al. (1994)
Temperate-Red pine, mixed hardwood	Massachusetts	37–150	NH_4NO_3	15–64	Castro et al. (1995)
Temperate-Hardwood forest	England	81	Compound fertilizer	45	Mojeremane et al. (2012)
Temperate-Black cherry and sugar maple	Pennsylvania	100	NH_4NO_3	35	Chan et al. (2005)
Tropical- Slash pine plantation	Florida	180	Urea	80–95	Castro et al. (1994)
Tropical forest	Panama	125	Urea	No effect	Veldkamp et al. (2013)
Tropical forest-Mature evergreen broadleaf Rehabilitated forest Disturbed forest	Guangdong Province, China	50 100 150 50 100 50 100	NH_4NO_3	6 14 32 No effect No effect No effect No effect	Zhang et al. (2008a)

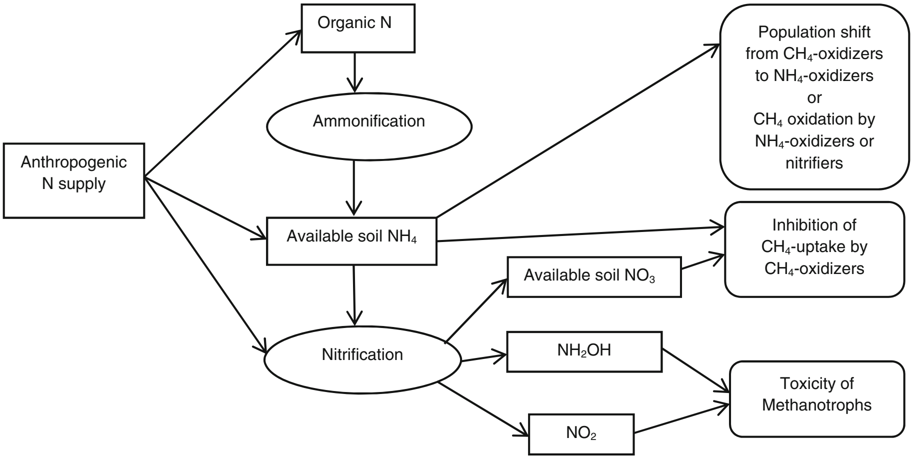

Fig. 2 Conceptual model showing N factors that control CH_4-oxidation in forest soils (modified from Bedard and Knowles 1989; Castro et al. 1994; Schnell and King 1994; Willison et al. 1995; Bowden et al. 2000; Bodelier 2011; Mochizuki et al. 2012; Konda et al. 2012)

based N-fertilization, which competes with CH_4 for the methane monooxygenase activity for reduction (Schnell and King 1994; Bodelier 2011; Alam and Jia 2012).

Different anthropogenic or N-cycling soil processes that increases NH_4^+ such as N-mineralization of organic matter, application of N-fertilizer containing NH_4–N (e.g., ammonium sulphate) (Saari et al. 2004), or release from clay particles (Green et al. 1994) can inhibit CH_4 oxidation and reduce CH_4 uptake through methanotrophic oxidation of NH_4^+ instead of CH_4 (Harmsen and Van Schreven 1955; Steudler et al. 1989; Green et al. 1994, Neff et al. 1994; Konda et al. 2012). Reduction in CH_4 uptake is also possible through changes in the N-cycle after fertilization, which induces a shift in the microbial population of CH_4-oxidizing bacteria to NH_4-oxidizers, thereby reducing CH_4 uptake (Castro et al. 1994). Inhibitory effects on CH_4 consumption can be detected shortly after N-fertilization and can persist for long periods (Smith et al. 2000; Papen et al. 2001). Additionally, high rates of nitrification can cause toxic effects on methanotrophs by producing NO_2^- and NH_2OH (Schnell and King 1994; King and Schnell 1994). Therefore, nitrification rate is negatively correlated with CH_4 uptake (Sitaula and Bakken 1993). As such, earlier studies (Hutsch et al. 1994; Willison et al. 1995) have also shown a slight inhibitory effect of nitrate (NO_3^-) on CH_4 oxidation. More recent studies further support these claims, reporting that NO_3^- addition can have a greater inhibitory effect on CH_4 oxidation in managed forest soils (Wang and Ineson 2003; Reay and Nedwell 2004), especially to the sites with low soil N (Mochizuki et al. 2012). Few studies (Wang and Ineson 2003; Xu and Inubushi 2004) have reported strong inhibitory effects of NO_3^- additions on CH_4 uptake compared to NH_4 addition in coniferous forest soils. In contrast, Wood and Silver (2012) in humid tropical forest observed the positive relationship between CH_4 consumption and soil NO_3. Ammonium and NO_3 stimulate CH_4 uptake by Methylomicrobium (a gamma-proteobacterium) under high methane concentration (10,000 ppm), but on the contrary inhibit CH_4 uptake by Methylocystis (an alpha-proteobacteirum) in forest soils (Mohanty et al. 2006). However, at a lower CH_4 concentration (1,000 ppm), only NO_3^- showed an inhibitory effect. In an incubation study, CH_4 uptake was observed after 27 years of N-fertilization with a high rate of 600 kg ha^{-1} (Borjesson and Nohrstedt 1998). This indicates that in the long-run, increased C retention in forest soils after N-fertilization can lead to increases in CH_4 uptake.

Management practices to increase CH_4 uptake

One of the major factors that can counteract atmospheric increases of CH_4 is the consumption of CH_4 by soils. Forest soils can exhibit a highly dynamic pattern in terms of CH_4 flux rates, with a net uptake shifting to net emission when soil becomes submerged (McNamara et al. 2006). Human activities have reduced the soil sink for atmospheric CH_4 by converting undisturbed forests to agricultural and urban land (Dobbie et al. 1996).

Type of fertilizer applied, or form of available N, can have significant impacts on CH_4 oxidation potential in forest soils. For example, applying KNO_3, which has low potential to inhibit CH_4 oxidation, can increase CH_4 uptake (Xu and Inubushi 2004). Other means to enhance CH_4 uptake are by reducing flooded condition, reducing acidity, decreasing compaction, and increasing C retention. Soil hydrology controls the direction and magnitude of CH_4 flux rates (Hutsch et al. 1994; Borjesson and Nohrstedt 1998; Sitaula et al. 2000; Christiansen et al. 2012). Therefore, improving soil drainage has been proposed as a mitigation measure that can reduce CH_4 emission from wet soils (Castro et al. 1995; Papen et al. 2001; Dalal and Allen 2008; Guckland et al. 2009; Christiansen et al. 2012) and even convert it to a CH_4 sink (Von Arnold et al. 2005; Guckland et al. 2009; Christiansen et al. 2012); however, in the process, it has been shown to release soil C (Smith et al. 2003) and should be considered in the context of all soil-derived GHG fluxes. Practices that increase C retention in forest soils could lead to increased CH_4 oxidation in the long-term (Borjesson and Nohrstedt 1998). For example, wood ash application in boreal spruce forest soil has been shown to increase CH_4 consumption (Maljanen et al. 2006). Soil acidity adversely affects atmospheric CH_4 consumption (Smith et al. 2000; Benstead and King 2001; Borken and Brumme 2009). Therefore, lime application in forest soil can increase CH_4 consumption by 25–560 % due to improvement in the chemical, biological, and physical condition of the soils (Borken and Brumme 1997).

Nitrous oxide emissions in forest ecosystems

Source and sink of N_2O

Nitrous oxide is a powerful GHG in the Earth's atmosphere, 310 times more potent than CO_2, and is involved in the destruction of the stratospheric ozone layer (Ehhalt et al. 2001; IPCC 2007b). Globally, soils are the largest source of N_2O, accounting for 65 % of total global emissions (Dalal et al. 2003; Smith and Conen 2004; IPCC 2007a, b). It is important to note that N_2O is increasing in the Earth's atmosphere by ≈ 0.3 % $year^{-1}$ in the recent past (Denman et al. 2007; Davidson 2009). Although N_2O emissions have decreased by 3.2 % in the US, direct emission from forest fertilization increased by 455 % since 1990 (USEPA 2012). The application of synthetic fertilizers to US forest soils in 2010 resulted in direct N_2O emissions of 0.4 Tg CO_2 Eq. (USEPA 2012). This increase is likely to continue in the absence of mitigation efforts (Fisher et al. 2007; Galloway et al. 2008). Ecosystem losses of N from 24 to 53 % are attributed to denitrification (Houlton et al. 2006), which is induced by high soil moisture when sufficient substrate (i.e., NO_3) is available (Dobbie et al. 1999). N_2O and nitric oxide (NO) are emitted by nitrification and denitrification. Depending on the soil saturation in water, nitrification or denitrification may contribute more to the production of N_2O with only denitrifying bacteria able to reduce N_2O to N_2 under complete anoxia. Under well-drained conditions, nitrification would be the main source of N_2O (Fig. 1).

Several studies have reported that significant amounts of N_2O are emitted from forest ecosystems, with estimates ranging from 2.4 to 5.7 Tg N_2O–N year^{-1} (Brumme et al. 1999; IPCC 2007b). Forest fertilization has become a common silvicultural practice where soil available-N is deficient (Chapin et al. 2002; Fox et al. 2007) or annual atmospheric N deposition is <2 kg N ha^{-1} (Galloway et al. 2004). A mature forest which is rich in N and SOC and has a low C:N ratio, has been observed to be a hotspot for N_2O emissions after application of N-fertilizer (Zhang et al. 2008b). Usually, a positive effect of N-fertilization has been reported on N_2O production from N- and P-limited forest soils via both nitrification and denitrification, two separate pathways for N_2O production (Fig. 1) (Matson et al. 1992; Castro et al. 1994; Sitaula et al. 1995b; Billore et al. 1996; Hall and Matson 1999; Ventera et al. 2003; Jassal et al. 2008, 2010; Zhang et al. 2008b). However, the peak N_2O emissions induced by N-fertilization only last for a short period from 2 to 3 weeks after fertilization in the summer and also in soil thawing periods in early spring for the first year. But in the second year, the elevation of N_2O emissions is significant for N-fertilizer application of >100 kg N ha^{-1} yr^{-1} (Peng et al. 2011). Elevated precipitation increases soil water content and thereby increases N_2O efflux, especially in combination with added N-fertilizer (Niboyet et al. 2011; Brown et al. 2012). However, studies have reported that N-fertilizer applications in temperate forest do not necessarily result in large emission of N_2O (Bowden et al. 2000).

A significant N_2O sink has been observed in managed forests of Canada (Kellman and Kavanaugh 2008) and Europe (Goldberg and Gebauer 2009; Inclán et al. 2012), as well as in natural forests of South Korea (Berger et al. 2013). Nitrogen-limited dry forest ecosystems can usually serve as a sink for atmospheric N_2O (Castro et al. 1993; Bowden et al. 2000; Papen et al. 2001; Rosenkranz et al. 2006; Goldberg and Gebauer 2009). The finding of soil as a sink for atmospheric N_2O can be explained by the presence of denitrifier population in soil, which in the absence of soil NO_3, can use atmospheric N_2O as an electron acceptor for the production of N_2 (Papen et al. 2001). It is important to understand these findings, and the controlling factors or processes involved, in order to manage forest ecosystems to mitigate increases in atmospheric N_2O.

Controls of N_2O emissions

Effects of N-fertilizer application on soil N_2O emissions are primarily controlled by soil N status, and thus practices that affect N mineralization or the form and amount of N-fertilizer applied (Corre et al. 1999; Ambus et al. 2006; Liu and Greaver 2009: Konda et al. 2012). However, fertilization effects are greatly dependent on soil conditions and local climate. Water filled pore space (WFPS), or soil water content, is a secondary controller of N_2O emissions (Corre et al. 1999; Garcia-Montiel et al. 2001; Kellman and Kavanaugh 2008; Koehler et al. 2009; Berger et al. 2013). However, in a tropical forest with a pronounced dry season, soil moisture might be the primary factor controlling N_2O emissions (Koehler et al. 2009). The magnitude of N_2O emissions has been shown to increase with increasing WFPS (Abbasi and Adams 2000). In a silt loam soil, for example, autotrophic nitrification is the predominant source of N_2O at 35–60 % WFPS, and denitrification predominates above 70 % WFPS (Davidson 1991; Bateman and Baggs 2005). Heterotrophic nitrification accounted for 20 % of N_2O emitted at 50 % WFPS in arable soils where the main source of available N is NH_4^+. A WFPS around 60 % offers optimal conditions for nitrification accounting for 81 % of N_2O emitted. Nitrous oxide emissions during heterotrophic nitrification would be expected to be greater in acidic soils such as in conifer forest, where autotrophic nitrification is often inhibited (Pedersen et al. 1999; Laverman et al. 2000). Other factors controlling N_2O emissions are labile C availability (Szilas et al. 1998; Davidson et al. 2000; Galloway et al. 2008; Goldberg and Gebauer 2009; Weslien et al. 2009; Goldberg et al. 2010; Brown et al. 2012; Christiansen et al. 2012; Wood and Silver 2012), soil temperature (Kellman and Kavanaugh 2008; Ullah and Moore 2011), soil pH (Sitaula et al. 1995b; Weslien et al. 2009; Peichl et al. 2010; Christiansen et al. 2012), soil texture (Berger et al. 2013), amount of throughfall (Borken and Beese 2005; Zona et al. 2013), C:N ratio (Ambus et al. 2006; Ullah and Moore 2011), land use or forest composition (Brumme et al. 1999; Ambus et al. 2006; Eickenscheidt et al. 2011), and age of forest stand (Gundersen et al. 2012).

The reviews on the effect of forest N-fertilization on N_2O emissions have been summarized in Table 3. Our review indicates that, in general, forest N-fertilization increases N_2O emissions ranging from 20 to >500 % compared to unfertilized controls. A large increase in N_2O emissions of >fivefold after N-fertilization in tropical forest of Panama is due to the presence of an organic layer in which nitrification increased significantly following N application (Koehler et al. 2009). Nitrogen fertilization in combination with increased precipitation and temperature can encourage soil N_2O emissions (Brown et al. 2012). Where precipitation is more variable, N_2O and NO emissions shift in importance, with N_2O flux dominating in wet months (Verchot et al. 1999). Yet, increases in soil temperature alone may (Malchair et al. 2010) or may not increase soil N_2O emissions (Niboyet et al. 2011; Brown et al. 2012).

Few studies (Bowden et al. 2000; Steudler et al. 2002; Maljanen et al. 2006) fail to observe significant increases in N_2O emissions after forest N-fertilization. This may be possible because: (1) trees may be better competitors for available N, (2) smaller amounts of N (33.3 kg N ha^{-1} per fertilization) were applied compared to other studies, and (3) WFPS was well below the optimum of 60–70 % for maximum denitrification (Garcia-Montiel et al. 2001; Steudler et al. 2002). Additionally, lack of fertilization responses on N_2O emissions in these studies were also due to rapid N cycling, reducing N_2O release especially in fine textured soils (Bowden et al. 2000) and N-rich forest soil (Maljanen et al. 2006).

Nitrogen-limited forest ecosystems can function as a sinks for atmospheric N_2O (Papen et al. 2001). In these ecosystems, N_2O uptake from the atmosphere into the soil can be catalyzed by soil denitrifiers which use N_2O from the atmosphere instead of NO_3^- as an electron acceptor for denitrification. On the other hand, forest N-fertilization can alter these dynamics and change soils from a net sink to a net source of atmospheric N_2O (Papen et al. 2001). Usually, the effects of N-fertilization on N_2O emission is triggered by rainfall events (Borken and Beese 2005; Zona et al. 2013) as soil moisture is one of the major drivers of N_2O emissions. Drought has been observed to decrease soil N_2O emissions and resulting soil N_2O consumption. Berger et al. (2013) observed N_2O consumption in sandy-loam soils of temperate deciduous forests in South Korea during early summer drought which then switched to a N_2O source during the monsoon season especially when WFPS was >36 %. Soils consume N_2O with varying magnitude, most likely in anoxic microsites throughout the soil profile, but the potential is larger in organic than in mineral forest soils, possibly due to higher organic C levels and C:N ratios (Ambus et al. 2006; Frasier et al. 2010; Ullah and Moore 2011).

Table 3 Effect of nitrogen-fertilization on N_2O emissions in managed forest ecosystems

Forest type	Location	Fertilizer rate (kg N ha^{-1} $year^{-1}$)	Fertilizer source	Increase in N_2O emissions (%) over unfertilized control	References
Temperate-scots pine	Norway	90	NH_4NO_3	82	Sitaula et al. (1995a)
Tropical-Slash pine plantation	Florida	180	Urea	87–100	Castro et al. (1994)
Tropical-Red pine	Massachusetts	150	NH_4NO_3	126	Magill et al. (1997)
Temperate-Douglas fir forest	Vancouver Island, Canada	200	Urea	101	Jassal et al. (2008)
Temperate-Douglas-fir	East coast of Vancouver Island, Canada	200	Urea	20	Jassal et al. (2010)
Temperate deciduous-black cherry-sugar maple forest	Pennsylvania	100	$(NH_4)_2SO_4$	No increase	Bowden et al. (2000)
Temperate-beech stand	Germany	140	$(NH_4)_2SO_4$	39	Brumme and Beese (1992)
Temperate-hardwood pine	northeastern United States	150	NH_4NO_3	210 376	Bowden et al. (1991)
Temperate-Hardwood forest	England	81	Compound fertilizer	49 fold increase	Mojeremane et al. (2012)
Semideciduous tropical forest	Panama	125	Urea	553 (275)[a,b]	Koehler et al. (2009)
Tropical evergreen broadleaf	Guangdong Province, China	150	NH_4NO_3	37	Zhang et al. (2008b)
Tropical rainforest, old growth	Panama	125	Urea	374 (96)[a,b]	Koehler et al. (2009)
Boreal forest: *Picea abies*	Finland	200	NH_4NO_3	No increase	Maljanen et al. (2006)
Humid tropical forest-palms	Brazil	100	NH_4Cl & $NaNO_3$	No increase	Steudler et al. (2002)
Temperate-Wet meadow (dominated by Carex scopulorum)	Colorado	100	Slow release urea	45 fold increase 22 fold increase	Neff et al. (1994)
Dry meadow (dominated by Kobresia myosuroides)					

[a] Average of 2 years

[b] Transitory effects—within 6 weeks of fertilization (in the parenthesis, long-term effects—after 6 weeks of fertilization)

Management practices to reduce N_2O emissions

Nitrous oxide emissions are affected by forest management, and particularly N management. But, scaling up prediction of these GHG exchanges in time and space from plot level studies remains a great challenge (Schulze et al. 2009). This is because of the fact that hot spots (small but reactive areas) and hot events (brief periods of high fluxes) frequently account for much of the N_2O gas exchange (Groffman et al. 2009). Improving N uptake efficiency (NUE) is important for increasing forest productivity while maintaining environmental quality and the economic efficiency of fertilization. Increasing NUE is possible by applying the right amount at the right time and adopting the best method of application through the right type or source of N-fertilizers. Selecting and applying an appropriate N source, which leads to low N_2O emission potential, such as NO_3^- compared to NH_4^+, can reduce nitrification driven N_2O emissions (Sitaula and Bakken 1993). However, Ambus et al. (2006) reported that NO_3^- (in addition to NO_2^-) is the dominant substrate for denitrification driven N_2O emissions. Thus, the type of fertilizer must be considered in the context of soil drainage/moisture and other obvious environmental quality concerns. Enhanced efficiency fertilizers such as slow-release fertilizers, coated urea fertilizers, controlled-release fertilizers, and stabilized fertilizers (nitrification or urease inhibitors) can minimize the potential for nutrients loss to the environment (McCarty and Bremner 1989; Hall 2005; Cahill et al. 2010). These fertilizers are commonly used for lawn care, golf courses, and agriculture (Shaviv 2001; Obreza and Rouse 2006; Morgan et al. 2009) but need to be explored for use in forest ecosystems. Nitrapyrin (2-chloro-6-trichloromethyl-pyridine) has been the inhibitor mostly used (Taylor 1983; Menéndez et al. 2012; Burzaco et al. 2013) although allylthiourea (Hall 1984; Jäntti et al. 2012) has also been used. These inhibitors block the NH_4^+ to NH_2OH step of ammonia oxidation (Campbell and Aleem 1965). Effectiveness of these inhibitors depends on concentration of inhibitors, strains of nitrifiers, substrate added, and loss of nitrapyrin through volatilization or degradation (Bedard and Knowles 1989).

Additional factors that can interact with fertilizer-N causing N_2O emissions are soil acidity, moisture level and forest management. Soil acidification can have a strong effect on N_2O emission in forest soils (Sitaula et al. 1995b). Liming of acidic forest soils has been shown to reduce N_2O emissions by 9–62 % (Borken and Brumme 1997; Klemedtsson et al. 1997) in addition to reducing soil acidity and increasing growth and vigor of forest trees especially sugar maple (Long et al. 1997). Management of the water level can also be effective in reducing N_2O emissions. However, there may be a trade-off with an increase in CH_4 emissions with soil water management (Gundersen et al. 2012). Applying small amounts of N enough for plant N uptake when WFPS is < 60 % can reduce N_2O emissions (Garcia-Montiel et al. 2001). Nitrous oxide emission measurement solely from inter-row soil can underestimate N_2O emissions by 44–67 % (Cai et al. 2012), because annual N_2O emission can decrease with distance from the tree (Pang et al. 2009). Therefore, a better understanding of the N-fertilizer effects on soil and forest management is vital for the mitigation of GHGs.

Forest regeneration strategies

Increased global demand for timber and other forest products will increase the need of forests to produce enough raw materials to satisfy societal needs. This could result in increased deforestation and, consequently, higher soil GHG emissions, primarily as CO_2. Generally, little to no change in soil C is observed when land is quickly regenerated following harvesting, regardless of intensity (Johnson and Curtis 2001). This illustrates the importance of developing the next cohort of trees as quickly as possible given that CO_2 emissions can be high the longer the soils are subjected to the increased temperatures and microbial activity associated with diminished canopy cover. Winjum et al. (1992) estimated that land available for reforestation and regeneration (natural and managed) could sequester 2.2–5.6 Gt C in the high-latitude zone over a 50-year period.

The magnitude of GHG emissions following harvest varies from site to site, geographical location (e.g., tropical vs. temperate forests), and management. For example, tropical forests may emit more GHGs immediately following harvest due to high decomposition rates, but these losses are quickly offset by rapid stand development. Additionally, the amount of C sequestered in wood products during a 50–100 year period is also much higher. On the other hand, the magnitude of loss in temperate regions is largely dependent on relative latitude. If we examine and compare the two highest producing temperate forests in the world, Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) in the Pacific Northwest (PNW) and loblolly pine (*Pinus taeda*) in the southeast US, Douglas-fir and western hemlock forests have slower decomposition rates than loblolly pine forests due to relative latitude. However, sawtimber rotation lengths are 40–50 years for Douglas-fir versus 25–30 years for loblolly pine so a few extra rotations are possible with loblolly pine over a 100 year period. This can lead to significantly more C accretion vs. a single rotation of 100 years (Johnson et al. 2003).

Sound sustainable forest management practices, specifically through proper and rapid forest regeneration (natural or artificial) immediately following harvest and debris management (i.e., forest residues and forest floor), is key to minimizing soil GHG emissions. For example, PNW stands are primarily cable logged, leaving the majority of debris on site. In combination with exceptional soil/site productivity and minimal site preparation (i.e., tillage), large GHG losses are likely low. If compared to loblolly pine in the southeastern US, decomposition rates are accelerated due to the hot humid conditions but stands fully occupy sites more quickly and have shorter rotation lengths. These forests often have a lot of soil disturbance from site preparation (i.e., bedding, ripping, and tillage); however, unlike agriculture where 100 % of land is cultivated/disturbed, only about 20–25 % of land is site prepared, leaving large amounts of debris and the majority of forest floor intact. Silvicultural systems that promote natural regeneration such as variable retention of seed tree regimes can also enhance C stocks through quick regeneration (Archambault et al. 2008). Whether natural or artificial regeneration strategies are deployed, rapid forest regeneration can increase C absorption capacity thereby reducing the amount of GHG emissions to the atmosphere (IPCC 2006; Johnson and Curtis 2001).

Mitigation strategies

Without substantial mitigation efforts, emissions of GHGs will continue to increase as need for food, fiber, and energy increases. Land use and its management practices impact a variety of ecosystem processes (e.g., photosynthesis, respiration, nitrification, methane oxidation, ammonification, denitrification, decomposition, and combustion), which can result in significant GHGs emissions. Reversing forest losses through restoration, improvement, and conservation of forest land is a vital step for GHG mitigation (Jackson and Baker 2010). Improved N-fertilizer management practices, restoration of previously cleared forest land, and sustainable harvesting can result in an increase in net GHG sink potential. These mitigation strategies require the collaborative effort of forest owners, managers and scientists.

Based on the findings of this review, GHG mitigation strategies in managed forest require an integrated management plan taking into account several of the following:

1. *Nitrogen uptake efficiency* Future GHG mitigation strategies need to involve the identification of fertilizer management practices that increase N uptake efficiency. Nitrogen uptake efficiency can be increased by managing N-fertilizer with appropriate form, rate, timing, placement, and by using controlled or slow release N-fertilizers.
2. *Nitrogen fertilization*, especially in nutrient poor soils, increases cumulative GHG sink potential in forest ecosystems, mostly due to stimulated productivity.
3. *Identification of temporal and spatial variability*: Special attention should be given to variation of soil fertility. Soil and foliar analysis provides an estimate of N-availability, which helps guide N-fertilizer rate decisions. Tools such leaf area index (LAI) using remote sensing are best to optimize nitrogen uptake efficiency and reduce gaseous loss.
4. *Use of inhibitors* Nitrification and denitrification processes can be reduced using inhibitors.
5. *Carbon sequestration* Best management practices that include avoiding deforestation or conversion of forest to other land uses, encouraging afforestation, increasing harvest interval, promoting urban forestry, and storing C in long lived forest products, need to be adapted to increase C retention in forest ecosystems.
6. *Managing soil acidity for optimal nutrient uptake* Managing soil acidity by avoiding the use of acidic fertilizers and by liming of acidic forest soils has shown to reduce N_2O emissions and increase CH_4 consumption.
7. *Consideration of forest type* Coniferous forest soils can have N_2O fluxes three times lower than deciduous forest soils. Methane uptake is also higher from sites dominated by coniferous species.
8. *Soil moisture/drainage* Improving soil drainage has been proposed as a mitigation measure to reduce CH_4 emissions from wet soils, but more work is needed to determine concomitant contributions to N_2O emissions.
9. *Extension education* Forest N-fertilizer management practices need to be transferred to those that own and manage forest lands in order to encourage adoption of fertilizer best management practices for the benefit of reducing emissions of GHGs.

Conclusions

Nitrogen-fertilizer use in managed forest ecosystems is increasing in order to provide social, economic and environmental services. This review has indicated several direct and indirect contributions of N-fertilization on the emissions of CO_2, CH_4 and N_2O GHGs. Most significantly is the need to improve N uptake efficiency through better N-fertilizer management practices to minimize GHG emissions. Nitrogen-fertilizer responses on soil production or consumption of GHGs are mainly affected by biological processes. They are also affected by different physical (i.e., temperature, moisture, rainfall, drainage, bulk density, and texture) and chemical (i.e., nutrient availability, labile C, soil pH, and C/N ratio) factors within managed forest ecosystems. Therefore, the multiple factors affecting N-fertilizer response on GHG emissions need to be simultaneously considered while evaluating N-fertilizer management practices for a particular site.

Additionally, managing one GHG (e.g., CH_4) might increase emissions of other GHGs (e.g., N_2O). Therefore, managing GHG emission effects of N-fertilization in managed forest ecosystems should evaluate cumulative net emissions considering all three major GHGs (CO_2, CH_4 and N_2O) in order to estimate the net benefit (Law 2013). The effects of fertilization on emissions of individual gasses are often evaluated, but studies evaluating impacts on all three GHGs are needed to understand the net impact on global warming potential (GWP). If all three major GHGs are considered in the budget calculations, proper N-fertilization may decrease GWP by stimulating a biological CO_2 sink (Albaugh et al. 2012), despite contrary effects on other GHGs. Consideration of the trade-offs are critical.

Limited studies are available on N-fertilizer response to GHGs emissions from managed forest ecosystems and the available studies are of small scale. The prediction and upscaling of GHG exchange is challenging as the findings from small scale studies have shown a wide variation in terms of N fertilizer use and GHG emissions. Generalizing these findings at higher scales can be erroneous. Therefore, large-scale studies or surveys are needed to better estimate GHG budgets and GWP implications of forest fertilization. Furthermore, GHG fluxes are often governed by high magnitude, short-term fluxes that are highly variable spatially. Special attention should be given to this spatial and temporal dynamic in order to increase the accuracy of GHG estimates.

Furthermore, methane uptake inhibition effects of N-fertilizer application have been well documented for the temperate managed forest ecosystems but only one study was available for a tropical climate. Therefore, there is a need for better understanding of N-fertilization effects on CH_4 uptake outside of temperate latitudes.

In conclusion, these identified gaps in our scientific understanding require the collaborative attention of forest owners, managers, and scientists. Recent worldwide increases in the use of N fertilizers to enhance ecosystem services from managed forest have also increased concerns about potential increase of GHG fluxes from managed forests. This review identifies strategies, cumulatively considered as best management practices for N fertilization, that have the potential to enhance the CO_2 and CH_4 sink capacity and simultaneously mitigate N_2O sources from managed forest ecosystems as an important GHG mitigation option for climate change strategies.

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