Biogeochemistry (2012) 109:203–218
DOI 10.1007/s10533-011-9620-0

Time-dependent radiative forcing effects of forest fertilization and biomass substitution

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Received: 30 September 2010 / Accepted: 5 July 2011 / Published online: 17 July 2011

Abstract Here we analyse the radiative forcing implications of forest fertilization and biomass substitution, with explicit consideration of the temporal patterns of greenhouse gas (GHG) emissions to and removals from the atmosphere (net emissions). We model and compare the production and use of biomass from a hectare of fertilized and non-fertilized forest land in northern Sweden. We calculate the annual net emissions of CO_2, N_2O and CH_4 for each system, over a 225-year period with 1-year time steps. We calculate the annual atmospheric concentration decay of each of these emissions, and calculate the resulting annual changes in instantaneous and cumulative radiative forcing. We find that forest fertilization can significantly increase biomass production, which increases the potential for material and energy substitution. The average carbon stock in tree biomass, forest soils and wood products all increase when fertilization is used. The additional GHG emissions due to fertilizer production and application are small compared to increases in substitution benefits and carbon stock. The radiative forcing of the 2 stands is identical for the first 15 years, followed by 2 years during which the fertilized stand produces slightly more radiative forcing. After year 18 the instantaneous and cumulative radiative forcing are consistently lower for the fertilized forest system. Both stands result in long-term negative radiative forcing, or cooling of the earth system. By the end of the 225-year simulation period, the cumulative radiative forcing reduction of the fertilized stand is over twice that of the non-fertilized stand. This suggests that forest fertilization and biomass substitution are effective options for climate change mitigation, as climate change is a long term issue.

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Keywords Radiative forcing · Climate change mitigation · Forest fertilization · Biomass substitution · Greenhouse gas · Wood products

Introduction

The forest sector can play an important role in climate change mitigation. The portfolio of forest-related mitigation activities includes afforestation, reducing deforestation, maintaining or increasing carbon stocks in forests, and using sustainable forest harvests to substitute for more greenhouse gas (GHG)-intensive fuels and materials. These activities help mitigate climate change, to the extent that they reduce net emissions to the atmosphere of GHGs including CO_2, N_2O and CH_4.

Managing forests so as to maintain or increase forest carbon stocks, while simultaneously producing a yield of usable biomass, is increasingly seen as a forest management strategy with large sustained mitigation benefit over the long term (IPCC 2007b). Using sustainably-produced forest biomass to substitute for fossil fuels provides permanent and cumulative climate benefits by avoiding the emission of fossil carbon into the atmosphere (Schlamadinger et al. 1997). Substituting wood in place of non-wood materials also reduces net GHG emissions through several mechanisms including the reduced fossil energy used to manufacture wood products compared with alternative materials; the avoidance of industrial process carbon emissions from e.g. cement manufacture; the temporary storage of carbon in wood materials; and the use of wood by-products as biofuel to replace fossil fuels (Sathre and O'Connor 2010).

The climate change mitigation potential of energy and material substitution depends on an adequate supply of sustainably-produced biomass. Forest growth on mineral soils in boreal regions is often limited by a low availability of nitrogen (N) (Tamm 1991), and fertilization has shown particular promise in increasing yields in boreal forests (Nohrstedt 2001; Saarsalmi and Mälkönen 2001). In Sweden, increased attention is being placed on balanced fertilization of forest land (Tamm et al. 1999). Beginning with the first field experiments with N fertilization in the 1920s, substantial experience has been accumulated in the effects of fertilization on Swedish forests (Nohrstedt 2001). Experiments have shown that it is possible to more than double the rate of stemwood production in some forest stands by optimising the availability of essential nutrients while avoiding the leaching of nutrients to the groundwater (Bergh et al. 1999).

GHG stocks and flows associated with forestry and wood substitution exhibit a variety of temporal patterns. Cyclical patterns repeatedly accumulate and release carbon stocks in a predictable pattern, such as a forest rotation cycle. Step changes involve a discrete increase or decrease in carbon stock in a given pool, for example the beginning and end of the life span of a wood product. Cumulative changes are flows that accumulate over time, such as avoided fossil emissions due to ongoing biomass substitution. Asymptotic changes are rapid at first and become slower over time, for example a decrease in soil carbon stock due to decay. Different forest management actions and wood product uses will result in different combinations of GHG emissions and removals over time. This heterogeneity over time complicates the comparison of the effectiveness of different GHG mitigation options (Gustavsson and Sathre 2011).

One approach to compare the climate effects over time of different systems is cumulative radiative forcing. Using this approach, the total change in energy accumulated within the earth system due to radiative forcing caused by each system over a given time horizon is calculated. The cumulative radiative forcing is used as a proxy for surface temperature change and hence disruption to physical, ecological and social systems.

Several authors have used cumulative radiative forcing to analyze and compare the climate effects of different systems. Zetterberg (1993) described the computational approach, and provided several examples including the development over time of a forest stand. Korhonen et al. (1993) used a similar approach to compare scenarios of future fossil fuel use in Finland. Zetterberg et al. (2004), Nilsson and Nilsson (2004), and Kirkinen et al. (2007) used the approach to compare the use of fossil fuels and peat fuel, and included different management options for peat land after peat extraction. Holmgren et al. (2007) compared the radiative forcing effects of using fossil fuels or forest residues for energy, considering the time dynamics of biomass decay if residues are left in the forest. Kirkinen et al. (2008, 2009) introduced a "relative radiative forcing commitment" which is the cumulative radiative forcing caused by using a fuel relative to the combustion energy of the fuel, and used the measure to compare several fossil fuels and biofuels. Bird (2009) discussed the issues of timing of CO_2 emissions of biomass systems, and suggested that cumulative radiative forcing is an appropriate measure of climate impacts in life cycle assessments. O'Hare et al. (2009) discussed CO_2 emission timing of crop-based biofuels and fossil fuels, and proposed a "physical fuel warming potential" defined as the cumulative radiative forcing caused by using a fuel, relative to the cumulative radiative forcing of using a reference fuel. Kendall et al. (2009) introduced a "time correction factor" to be applied to CO_2 emissions that occur at the beginning of a defined time horizon but are amortized over the entire horizon, such that the cumulative radiative forcing are equivalent for both. Levasseur et al. (2010) proposed the use of "dynamic characterization factors" for the global warming impact category of life cycle assessment, based on the cumulative radiative forcing of different GHGs over different time horizons.

Previous research using static (i.e. non-time-dependent) methods has shown that the net GHG balance of fertilizing boreal forests is beneficial, i.e. the avoided emissions due to increased substitution potential and increased carbon stock are greater than the additional emissions due to fertilization (Sathre et al. 2010). However, the fertilization occurs first and the substitution benefits and carbon stock increase occur later, so the climate impacts may vary over time. Furthermore, previous research has shown that the net GHG balance of substituting forest biomass in place of fossil fuels (Gustavsson et al. 1995, 2007) and non-wood materials (Sathre and O'Connor 2010) is beneficial. The magnitude of the climate benefit of biomass substitution, however, depends on the time horizon under consideration (Börjesson and Gustavsson 2000; Gustavsson et al. 2006). In the present study we conduct a time-dependent simulation of the cumulative radiative forcing of forest fertilization and biomass substitution, with consideration of the temporal patterns of GHG emissions and removals.

Methodology

We model and compare the climate implications of producing and using biomass from a unit hectare of fertilized and non-fertilized forest land in northern Sweden. Figure 1 shows schematically the GHG flows and stocks that we track annually for each system. These include the CO_2, N_2O and CH_4 from production and application of fertilizer, N_2O emission from fertilized soil, soil C stock change due to fertilization, CO_2 from fossil fuels used for biomass harvest and transport, avoided CO_2 emissions from using the biomass to substitute for materials and fuels, and C-stock changes in living trees, wood products, and in soil and decaying biomass. We calculate the annual net emissions of CO_2, N_2O and CH_4 for each system, and the annual atmospheric concentration decay of each emission. We calculate the resulting annual changes in instantaneous radiative forcing and the cumulative radiative forcing over a 225-year period with 1-year time steps. This time period is used because it allows us to identify long term patterns, and it corresponds to approximately 2 and 3 forest rotations for the non-fertilized and fertilized stands, respectively.

Biomass production and harvest

Year 0 of the 225-year simulation period begins with cleared forest land on which the 2 stands are established. We assume that previous management was the same for both land areas, thus the effects of the previous harvest (e.g. decaying biomass) are the same for both stands and are not considered in the analysis. We use the empirical growth model DT to estimate biomass production (Nilsson et al. 2011). The modelled tree species is Norway spruce

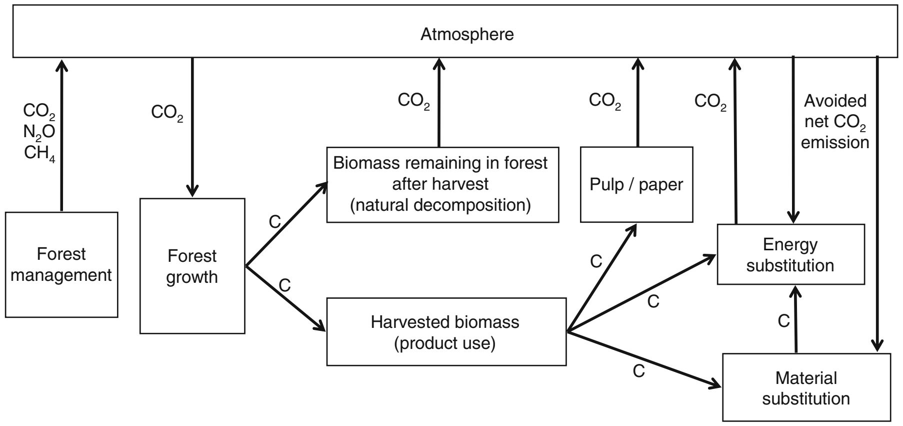


Fig. 1 Schematic diagram of GHG flows and stocks that are tracked on an annual basis

(*Picea abies*). Stand characteristics including growth, height, stem shape and quality, and harvest volumes in thinnings and final harvests are all estimated with a 1-year time step. The DT model forecasts the development of forest stands from a young age (>5 years) to final harvest. Development of young stands is described by functions of height development and statistical relations between diameter and height (Fahlvik and Nyström 2006), with initial conditions based on circular plots with 10 m radius. Development of established stands is described by species-specific diameter growth functions (Elfving 2004), with height and diameter calculated for individual trees. Estimates of total basal area of the whole stand, derived from the ProdMod2 forest generator (Ekö 1985), are used to adjust the diameter and basal area growth of individual trees. Thinning regimes are in accordance with the recommendations of the Swedish Forest Agency (National Board of Forestry 1985). The timing of final harvest is determined by when the current annual increment is about the same level or less than the mean annual increment.

We use biomass expansion factors for appropriately-aged Norway spruce trees (Lehtonen et al. 2004) to disaggregate total biomass production into dry mass of stems, branches and tops, needles, stumps, coarse roots, and fine roots. We assume that 60% (by mass) of stemwood is large-diameter logs ("sawtimber") and 40% is small-diameter logs ("pulpwood"). We further distinguish between tree biomass cut during thinning operations and during final fellings. Based on this breakdown of tree biomass, we assume that 100% of large-diameter stemwood is harvested and used for production of wood construction material, 100% of small-diameter stemwood is harvested and used for pulp production or bioenergy, 75% of branches and tops and 25% of needles (here termed "slash") from both thinnings and final harvest are recovered and used for bioenergy, and 50% of stumps and coarse roots (here termed "stumps") are harvested and used for bioenergy. Primary energy used for forest establishment, thinning, roundwood harvest, and transport is based on data for northern Sweden (Berg and Lindholm 2005). Primary energy used for recovery and transport of slash and stumps is based on Eriksson et al. (2007).

Fertilization

Management of the fertilized and non-fertilized stands is assumed to be identical for the first 15 years, before fertilization commences. The fertilized stand then begins to receive small, frequent applications of N or nitrogen–phosphorous–potassium (NPK) fertilizers. Biannual applications begin when the stand height is 2–4 m and continue until canopy closure (basal area = 25 m^2/ha). Applications are less frequent after canopy closure, and the last application occurs at least 10 years before expected final harvest. Fertilization dosage and frequency will depend on actual need, determined by nutrient analysis of tree needles. Here we assume 10 applications of 125 kg N per hectare, occurring at years 16, 18, 20, 22, 24, 26, 28, 38, 48 and 58 of a 69-year rotation.

GHG emissions associated with the production and use of fertilizers include CO_2, N_2O and CH_4. We assume half of the applied N is in Skog-CAN fertilizer (27-0-0) and half is in Opti-Crop fertilizer (24-4-5). Primary energy use and GHG emissions from the fertilizer manufacture are based on Swedish industry (Davis and Haglund 1999), and are assumed to occur during the year the fertilizer is applied. Aerial application of fertilizer is assumed, because this method is not restricted by wet soil conditions or interference by growing trees. The amount of fossil fuel used for fertilizer application by helicopter is based on Mead and Pimentel (2006), corresponding to 0.022 kg CO_2 emission per kg fertilizer.

The use of N fertilizer may lead to emission of N_2O from the soil, due to the microbiological processes of nitrification and denitrification. The emission of N_2O from forest soils is subject to considerable uncertainty. For example, Crutzen et al. (2008) suggest that 3–5% of N applied globally for agricultural biofuel production is emitted as N_2O, while Maljanen et al. (2006) found no significant difference in N_2O emission from a fertilized and non-fertilized spruce forest site in Finland. Nordin et al. (2009) suggest that between 0.5 and 1% of the N in the fertilizer applied to Swedish forests can be expected to be released as N_2O, and MacDonald et al. (1997) found that 1% of the N deposited on upland spruce forests in Scotland was emitted as N_2O. In this study we assume that 1% of the N in the applied fertilizers is released as N_2O, during the year the fertilizer is applied. We do not consider the potential for N addition to inhibit the oxidation of CH_4 by soil microorganisms, which is expected to be negligible in comparison to soil carbon stock changes due to fertilization (Nordin et al. 2009).

Biomass substitution

We model the material substitution benefits of using large-diameter stemwood to produce wood construction material that substitutes in place of reinforced concrete. Material substitution data are based on a case study comparing a Swedish multi-story apartment building constructed with either a wood structural frame or a reinforced concrete frame (Gustavsson et al. 2006). The substitution benefit is calculated based on the net GHG emission reduction per unit of additional wood needed to make the wood-frame building instead of the concrete-frame building. Emission calculations take into account the differences between the buildings due to the emission from fossil fuels used to manufacture and transport the materials. We assume these emissions occur during the year of harvest, when the building is assumed to be constructed.

Material substitution benefits also include the avoided cement process emissions from concrete substituted by wood material. These process reactions occur at different times during the building life cycle. Calcination emissions from cement manufacture occur when the building is constructed. Then a gradual removal of CO_2 from the atmosphere into the concrete material occurs due to carbonation, equalling about 18% of the initial calcination emission (Dodoo et al. 2009), and is assumed to occur linearly during the 50-year service life of the building. Finally, a rapid carbonation removal of CO_2 at the end of the building's service life occurs when the surface area of the concrete is increased due to demolition and crushing, and equals about 20% of the initial calcination emission (Dodoo et al. 2009). When wood is used instead of concrete these process reactions are avoided. We model the avoided calcination emission as if it were a CO_2 removal, and we model the avoided carbonation removal as if it were a CO_2 emission.

A third material substitution impact is the use of residues from wood processing, construction and demolition as biofuel to replace fossil fuel. Some wood processing residues are used internally as energy for e.g. kiln-drying lumber, and the remainder is available externally for use as bioenergy. We assume that fossil fuel substitution from wood processing and construction site residues occur during the year the trees are harvested. During the assumed 50-year building life span, we account for the carbon stock stored in wood building materials. At the end of the building's service life we assume that demolition wood is recovered and used as bioenergy (Sathre and Gustavsson 2006). Demolition materials are increasingly recovered as efficient management of post-use building materials becomes a priority in many European countries including Sweden (European Commission 2001). Energy used for recovery and transport of processing and demolition residues is based on Eriksson et al. (2007).

Small-diameter stemwood is often used in Sweden for pulp and paper production. In this study we analyse the use of additional quantities of biomass produced through fertilization, but we have not analysed whether there is demand for additional pulpwood in the Swedish pulp industry. We therefore assume a constant demand for pulpwood that is equal to the average annual production of small-diameter stemwood in the non-fertilized stand. Thus, all small-diameter stemwood from the non-fertilized stand is used for pulp production, the same amount of small-diameter stemwood from the fertilized stand is used for pulp production, and any additional small-diameter stemwood from the fertilized stand is assumed to be used as bioenergy. We assume for simplicity that if this excess small-diameter stemwood is not used for bioenergy it will be left in the forest to decay naturally. We acknowledge that this may not be the most economically beneficial use of this wood (Sathre and Gustavsson 2009), though it is analytically consistent because each stand provides the same amount of pulpwood to society. Emissions from processing pulpwood into paper occurs outside the system boundaries of this study and are thus not included in our calculations, but will be the same for both stands. We assume for simplicity that all carbon in paper products will be oxidized and enter the atmosphere during the year the pulpwood is harvested.

When biofuels are combusted, we assume 100% of their carbon content returns to the atmosphere as CO_2. The energy source that bioenergy replaces influences the resulting GHG balance (Schlamadinger et al. 1997). Globally, we are heavily dependent on fossil fuels, with coal, oil, and fossil gas providing 26, 34 and 21% of global primary energy supply, respectively (International Energy Agency 2009). The amount of CO_2 emitted per unit of energy varies between fossil fuels, with coal emitting most and fossil gas emitting least. To show the range of climate impact of replacing fossil fuel in stationary plants, we consider cases where bioenergy replaces either coal or fossil gas with relative conversion efficiencies of 100 and 96%, respectively (Gustavsson et al. 2006). Values of specific CO_2 emission from the combustion of fossil fuels used are 110 kg CO_2 per GJ for coal, 81 kg CO_2 per GJ for oil, and 66 kg CO_2 per GJ for fossil gas, and include emissions during the entire fuel-cycle from the natural resource to the combustion of the fuels (Gustavsson et al. 1995).

Soil carbon stocks

Forest management can cause soil carbon stocks to increase or decrease, which can have significant climate implications. In this analysis we consider soil carbon stock changes from 3 sources: fertilization, clear-cut harvesting, and decay of biomass residues. We estimate each of the 3 effects separately, and do not consider potential interactions between the effects.

Nitrogen fertilization generally causes an increase in soil carbon stock (Johnson and Curtis 2001), due to both an increase in litter input from the enhanced aboveground growth and a decrease in soil microbial activity leading to slower decomposition of soil organic matter (Franklin et al. 2003). The increase in soil carbon stock due to fertilization depends on the dosage of N, dosage of other nutrients, soil type, tree species and climate. Furthermore, it depends on the duration of N application and the time since application, with an initial strong response giving way to continued accumulation at a slower rate and with a gradual decrease toward prior conditions if the treatment is discontinued. The literature contains few studies of the effects of fertilization on carbon stocks in boreal forest soil. Nohrstedt et al. (1989) analyzed 2 Scots pine sites in Sweden and found soil carbon stock to increase by 10–26% in N-fertilized plots over a 15-year time frame. The increase in soil carbon stock was 0.21–0.35 tC/ha-yr over the 15-year period. Mäkipää (1995) studied five Scots pine sites and one Norway spruce site in Finland that were N-fertilized over a 30-year time frame. Carbon stock increased in the humus layer by 14–87% and in the mineral soil by 15–167%. The total increase was greatest at the spruce site where it averaged 0.63 tC/ha-yr over the 30-year period. Hyvönen et al. (2008) analyzed 15 sites in Sweden and Finland that received varying dosages of N and NPK fertilizer over a time frame of 14 to 30 years. They found the fertilized plots to have consistently higher soil carbon stock than the non-fertilized plots. The soil carbon stock of the fertilized plots increased an average of 0.66 and 0.25 tC/ha-yr more than the non-fertilized plots for Norway spruce and Scots pine sites, respectively. Our estimate of soil carbon increase due to fertilization is based on data from Eriksson et al. (2007), who modelled a Norway spruce stand over a 300-year time frame in Sweden and found that NPK-fertilization led to an increased soil carbon stock compared to traditional forest management. Most of that increase was evident during the first rotation period, though with each successive rotation the fertilized stand continued to accumulate more carbon than a non-fertilized stand. Based on this relation, we use a simplified assumption that the soil carbon increases by 0.35 tC/ha-yr during the first 50 years of fertilization, and then continues to increase by 0.027 tC/ha-yr as long as the fertilized regime continues (Eriksson et al. 2007).

Clear-cut harvesting can cause a decrease in soil carbon stock during the decades after harvest, followed by a gradual recovery of soil carbon stocks during the remainder of the rotation period (Jandl et al. 2007). This is thought to result from a reduction in litter inputs to the soil and an increase in the rate of decomposition of soil organic matter (Covington 1981), although other factors may also be involved, for example a movement of carbon from upper to lower soil horizons (Yanai et al. 2003). In a study of Norway spruce sites in Sweden, Olsson et al. (1996) found that carbon content in the forest floor and upper 20 cm of mineral soil had decreased by 10–20 tC/ha during 15–16 years after harvest. In a study of mixedwood forest sites in Canada, Pennock and van Kessel (1997) found the carbon stock in the forest floor and upper 45 cm of soil to be about 14 tC/ha lower in sites 6–20 years after harvest, compared to mature stands. In a modelling exercise based on Finnish conditions, Peltoniemi et al. (2004) estimated that total soil carbon stock reaches a minimum 16–22 years

Table 1 Decay constants assumed for dead biomass remaining in forest

Biomass type	Decay constant (k)	Reference
Small diameter logs	0.033	Næsset (1999)
Stumps and coarse roots	0.046	Melin et al. (2009)
Branches and tops	0.074	Palviainen et al. (2004)
Fine roots	0.129	Palviainen et al. (2004)
Needles	0.170	Palviainen et al. (2004)

after harvest, at a level 15–30 tC/ha less than pre-harvest conditions, followed by a gradual recovery during the remainder of the rotation period. In this simulation we make a simplified assumption of a linear decrease in soil carbon stock of 1 tC/ha-yr for the first 20 years after clear-cut harvest, followed by a linear increase of 0.67 tC/ha-yr for the next 30 years, such that the pre-harvest carbon stock is restored 50 years after harvest.

Carbon stock in forest soil and litter is also affected by the decay of biomass residues that are left in the forest after harvest. We assume that biomass that is not removed from the forest will decay into CO_2 at a negative exponential rate, following Eq. 1

$$M_t = M_0 \times e^{-kt} \tag{1}$$

where M_t is the dry mass remaining at time t, M_0 is the initial dry mass at the time the tree is cut, k is a decay constant specific to each biomass type, and t is the number of years since the death of the tree. Decay constants assumed for each biomass type are listed in Table 1. Logs and stumps decay slowly, while fine roots and needles decay more rapidly. The assumed decay constants are average values for each biomass type, while the actual decomposition rate will vary with, inter alia, ground contact, soil moisture, and aspect (Næsset 1999).

GHG flux and estimates of radiative forcing

Emissions and removals of GHGs from all sources that occur during each year are summed, and the net annual emissions are treated as pulse emissions of CO_2, N_2O and CH_4. The atmospheric decay of each annual pulse emission is then estimated using Eqs. 2, 3 and 4 (IPCC 1997, 2001, 2007a):

$$(CO_2)_t = (CO_2)_0 \times \left[0.217 + 0.259e^{\frac{-t}{172.9}} + 0.338e^{\frac{-t}{18.51}} + 0.186e^{\frac{-t}{1.186}}\right] \tag{2}$$

$$(N_2O)_t = (N_2O)_0 \times \left[e^{\frac{-t}{114}}\right] \tag{3}$$

$$(CH_4)_t = (CH_4)_0 \times \left[e^{\frac{-t}{12}}\right] \tag{4}$$

where t is the number of years since the pulse emission, $(GHG)_0$ is the mass of GHG emitted as a pulse emission at year 0, and $(GHG)_t$ is the mass of GHG remaining in the atmosphere at year t. The decay over time of unit pulse emissions of CO_2, N_2O and CH_4 is shown in Fig. 2. The total atmospheric

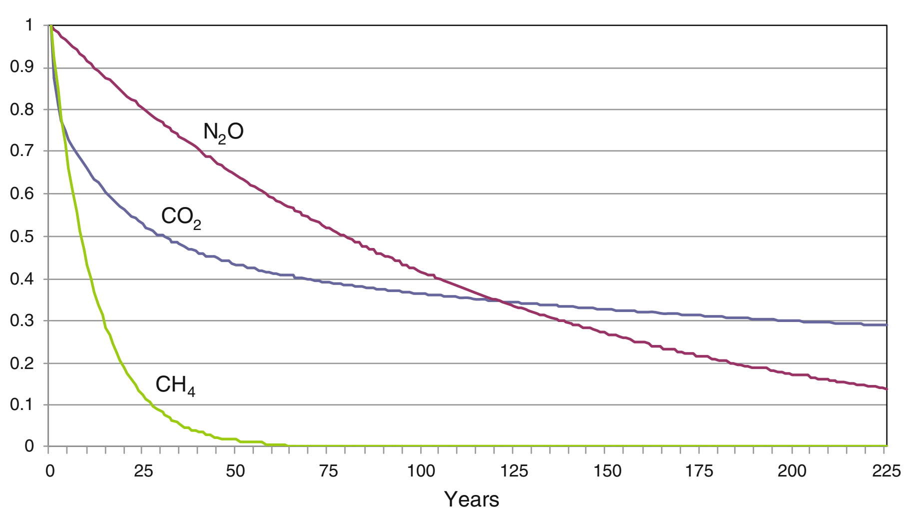


Fig. 2 Atmospheric decay of unit pulses of CO_2, N_2O and CH_4

mass of each GHG for each year of the simulation period is then determined by summing the emissions occurring during that year plus the emissions of all previous years minus their decay during the intervening years.

The change in atmospheric mass of each GHG is then converted to change in atmospheric concentration, based on the molecular mass of each GHG, the molecular mass of air, and the total mass of the atmosphere which is 5.148×10^{21} g (Trenberth and Smith 2005). Annual changes in instantaneous radiative forcing due to the GHG concentration changes are then estimated using Eqs. 5, 6 and 7 (IPCC 1997, 2001, 2007a):

$$F_{CO_2} = \frac{3.7}{\ln(2)} \times \ln\left\{1 + \frac{\Delta CO_2}{CO_{2ref}}\right\} \quad (5)$$

$$F_{N_2O} = 0.12 \times \left(\sqrt{\Delta N_2O + N_2O_{ref}} - \sqrt{N_2O_{ref}}\right) - f(M,N) \quad (6)$$

$$F_{CH_4} = 0.036 \times \left(\sqrt{\Delta CH_4 + CH_{4\,ref}} - \sqrt{CH_{4\,ref}}\right) - f(M,N) \quad (7)$$

where F_{GHG} is instantaneous radiative forcing in W/m^2 for each GHG, ΔGHG is the change in atmospheric concentration of the GHG (in units of ppmv for CO_2, and ppbv for N_2O and CH_4), $CO_{2ref} = 383$ppmv, $N_2O_{ref} = 319$ppbv, $CH_{4ref} =$ 1774ppbv, and *f(M,N)* is a function to compensate for the spectral absorption overlap between N_2O and CH_4 (IPCC 1997, 2001, 2007a).

We then estimate the cumulative radiative forcing occurring each year in units of W·s/m^2, by multiplying the instantaneous radiative forcing of each year by the number of seconds in a year. This operation converts the energy flow per unit of time of the radiative imbalance caused by GHGs into units of energy accumulated in the earth system per year.

Results

Carbon stock in living tree biomass in the fertilized and non-fertilized stand is shown in Fig. 3. The rotation period for the fertilized stand is 69 years, while that of the non-fertilized stand is 109 years. During the 225-years simulation period, 3 rotations occur for the fertilized stand while 2 rotations occur for the non-fertilized stand. The periodic spikes that occur during the growth periods are due to 3 thinnings that are done during each rotation. The average carbon stock is greater in the fertilized stand due to its more rapid initial development and its greater final yield. The average rate of biomass production is about double in the fertilized stand, due to its shorter rotation period and higher yield per rotation. Large-diameter stemwood comprises about 38% of total biomass production. Carbon stock in soil

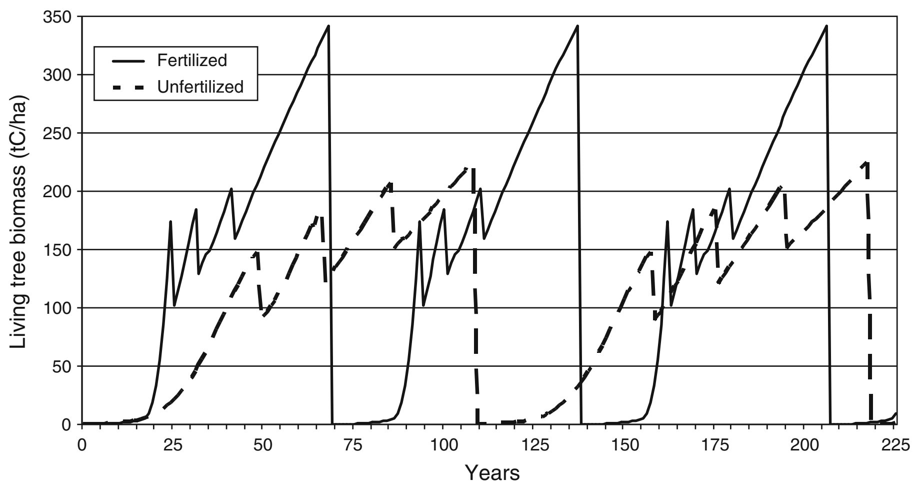


Fig. 3 Carbon stock in living tree biomass in fertilized and non-fertilized stands

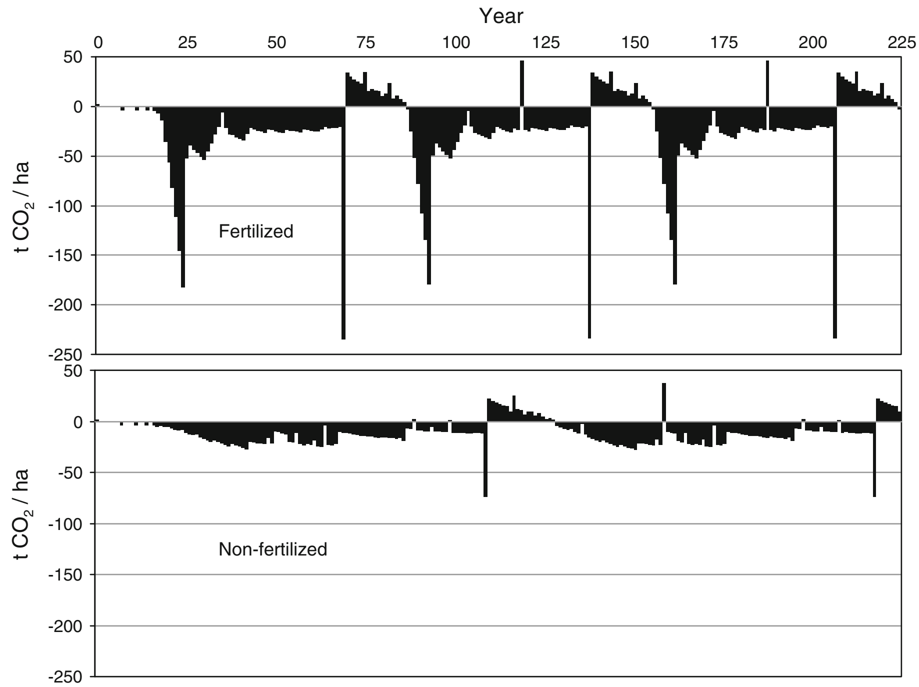


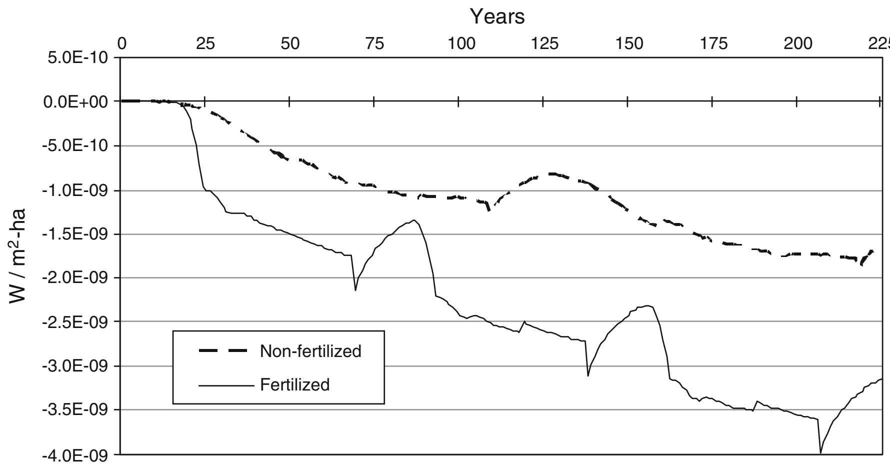
Fig. 4 Annual net CO_2 emissions associated with fertilized and non-fertilized stands. Positive values are emissions to the atmosphere, and negative values are removals from the atmosphere. Substitution effects are based on replacing coal

and wood products is also greater in the fertilized stand than the non-fertilized stand.

Annual net CO_2 emissions over the 225-year simulation period are shown in Fig. 4, with coal as the reference fossil fuel. The annual CO_2 emissions reflect the pattern of the forest rotation periods, showing negative emissions due to carbon uptake by growing trees, avoided emissions due to material and fuel substitution at thinning and harvest events, and positive emissions due to decaying biomass left in the forest after harvests. The positive emission spikes at years 119 and 188 of the fertilized stand, and year 159 of the non-fertilized stand, are due to the combustion of demolition materials from buildings built from forest harvests 50 years earlier. The negative emissions of the fertilized stand are of greater magnitude than the non-fertilized stand. N_2O and CH_4 emissions from the fertilized stand (not shown) are relatively minor, and are associated with fertilization events.

The annual instantaneous radiative forcing associated with the fertilized and non-fertilized stands is shown in Fig. 5, with coal as the reference fossil fuel. The radiative forcing for each year is due to the emissions and removals of GHGs occurring during that year, as well as GHGs emitted or removed in previous years. The instantaneous radiative forcing over the 225-year period reflects the pattern of net annual emissions (Fig. 4), determined by events during the forest rotation periods (Fig. 3), but rapid changes are buffered by the long atmospheric residence time of emissions and removals (Fig. 2). The instantaneous radiative forcing for both stands is slightly positive for the first 6 years due to fossil emissions from stand establishment, but then become negative as the growing trees remove CO_2 from the atmosphere, and as the harvested biomass is used to substitute for fossil fuels and materials. The radiative forcing for both stands is identical for the first 15 years, after which the forcing patterns diverge due to emissions from fertilizer production and application, followed by faster growth of the fertilized stand. For the rest of the 225-year period the negative radiative forcing is greater in magnitude for the fertilized stand than the non-fertilized stand, mainly due to more biomass production resulting in increased carbon stocks and more material and fuel substitution. Negative radiative forcing causes cooling of the earth system, while positive radiative forcing causes warming.

Fig. 5 Annual instantaneous radiative forcing of fertilized and non-fertilized stands. Substitution effects are based on replacing coal



The cumulative radiative forcing of the fertilized and non-fertilized stands is shown in Fig. 6, broken down into contributions from different biomass types and forest management, with coal as the reference fossil fuel. During the first 15 years the cumulative radiative forcing is the same in both stands. It begins slightly positive in sign due to fossil emissions from stand establishment, and by Year 10 becomes negative due to carbon uptake in growing trees. The radiative forcing of the fertilized stand increases during Year 16 due to GHG emissions from fertilizer production and application. By Year 18, however, the CO_2 removal due to the increased biomass growth of the fertilized stand compensates for these emissions, and the cumulative radiative forcing of the fertilized stand becomes less than that of the non-fertilized stand. By the end of the 225-year period, the negative cumulative radiative forcing of the fertilized stand is 2.2 times more than that of the fertilized stand.

The biggest single contribution to the reduced radiative forcing of both stands is the increased biomass substitution from sawlogs. This includes material substitution of concrete building material with wood building material, and fossil fuel substitution with biomass residues from wood processing, construction site, and building demolition. The contribution due to carbon stock in living biomass is also significant. It varies over time reflecting the rotation patterns of the stands, but does not continuously accumulate over time and instead equilibrates around a level determined by the average carbon stock of the forest stands. Emissions due to forest management are relatively minor, and are the only agents that result in net positive radiative forcing. Forest management emissions include fertilizer production and application, N_2O emission from fertilized soil, and fossil emissions from stand establishment, thinning, and harvest.

The radiative forcing effect of CO_2 greatly dominates the radiative forcing of the other GHGs. Of the total cumulative radiative forcing of the fertilized stand after 225 years (-15.8 W·s/m^2·ha), 0.72% is positive radiative forcing from N_2O, 0.04% is positive radiative forcing from CH_4, and the remainder is net negative radiative forcing related to CO_2 flows.

The contributions to radiative forcing reduction of using small diameter logs, slash, and stumps as bioenergy are all greater for the fertilized stand than the non-fertilized stand, due to its higher production rate. Slash contributes about twice as much negative radiative forcing as stumps. Excess small diameter logs that are not used for pulp production are a significant source of biofuel from the fertilized stand. The radiative forcing reduction due to forest biofuel use is a function of 4 factors: (1) the quantity of biofuel recovered; (2) the fossil energy needed to recover and transport the fuel; (3) the carbon intensity of the fossil fuel replaced by the biofuel; and (4) the fate of the stored carbon if the biomass were not used for bioenergy. Biomass left to decompose naturally in the forest releases CO_2 into the atmosphere over a time scale of years to decades. When used as bioenergy, the carbon in the biomass is released immediately into the atmosphere and causes radiative forcing sooner. All other factors being equal, the

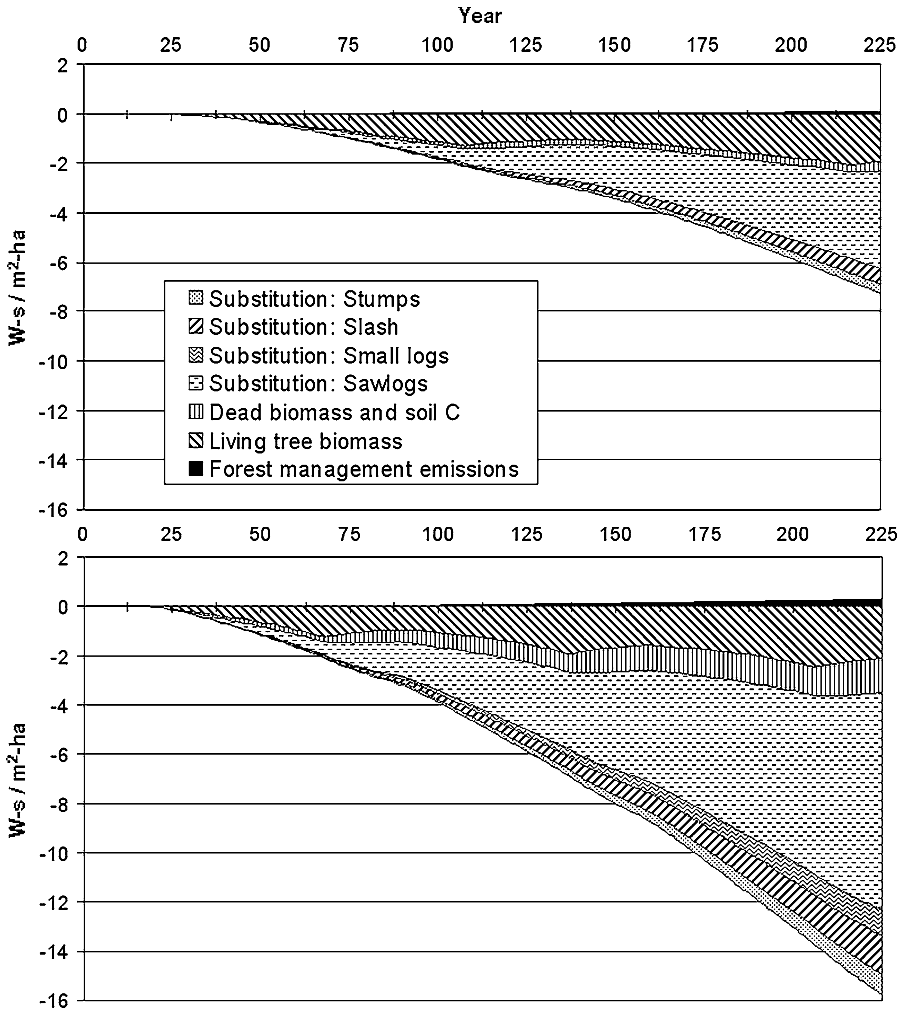


Fig. 6 Cumulative radiative forcing of non-fertilized (*top*) and fertilized (*bottom*) stands, broken down by biomass types and forest management actions. Substitution effects are based on replacing coal

relative radiative forcing benefit of using forest fuel as bioenergy is greater for biomass types that naturally decay quickly.

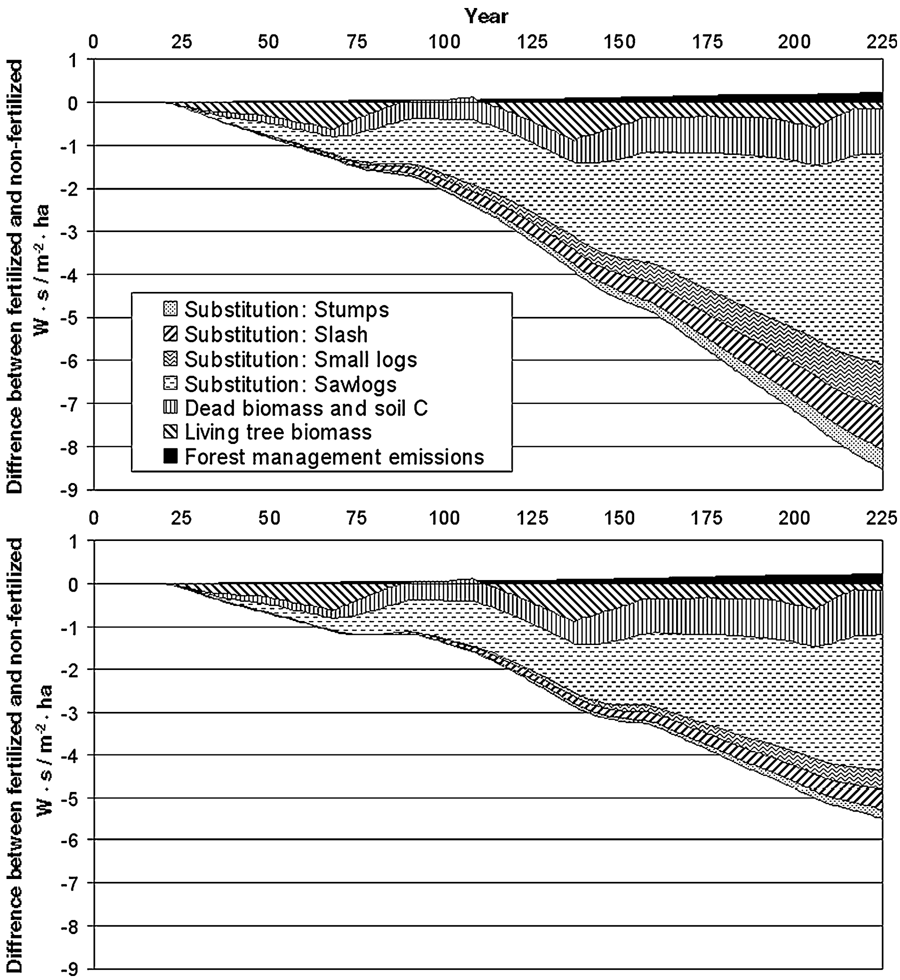
Of the forest fuels analyzed in this study, small diameter logs are the slowest to decay naturally in the forest (Table 1), and also require the least fossil fuel to recover a unit of biomass. Slash is quicker to decay, and require more fossil fuel to recover. Stumps are slow to decay naturally, and require the most fossil energy to recover. Table 2 shows the total radiative forcing and the specific radiative forcing per unit of biomass for these 3 types of forest fuels. The specific radiative forcing reduction is greatest for slash, and is roughly equal for stumps and small diameter logs. The difference in specific radiative forcing of forest fuels produced in the fertilized and non-fertilized stands is a result of the different temporal patterns of biofuel availability during the 225-year period.

Figure 7 shows the difference in cumulative radiative forcing between the fertilized and non-fertilized stands, or in other words, the change in radiative forcing that occurs if fertilization is used instead of non-fertilized forest management. The reference fossil fuel that is replaced by biomass substitution is coal (top) and fossil gas (bottom). The reduction in cumulative radiative forcing after 225 years is about 56% greater when coal use is replaced, because its carbon intensity is greater than that of fossil gas. The reference fossil fuel that is replaced affects the radiative forcing of material substitution (with sawlogs) and energy substitution (with stumps, slash, and small-diameter logs). It does not affect the radiative forcing of living tree biomass,

Table 2 Quantities and radiative forcing impacts of forest fuels recovered from fertilized and non-fertilized stands during 225 years. The forest fuels replace either coal or fossil gas

Biomass type	Biomass quantity (t/ha dry matter)	Cumulative radiative forcing ($W \cdot s/m^2 \cdot ha$)		Specific cumulative radiative forcing ($mW \cdot s/m^2 \cdot$ton dry matter)	
		Coal	Fossil gas	Coal	Fossil gas
Fertilized stand					
Small diameter logs	323	−1.05	−0.45	−6.5	−2.8
Slash	422	−1.57	−0.78	−7.5	−3.7
Stumps	263	−0.84	−0.35	−6.4	−2.7
Non-fertilized stand					
Slash	212	−0.64	−0.31	−6.0	−2.9
Stumps	145	−0.37	−0.15	−5.1	−2.1

Fig. 7 Difference in cumulative radiative forcing between fertilized and non-fertilized stands. The reference fossil fuel that is replaced is coal (*top*) and fossil gas (*bottom*)



decaying biomass and soil carbon, and forest management emissions.

Discussion

In this analysis we have found that increasing biomass production through forest fertilization can significantly reduce cumulative radiative forcing. The emissions from intensified forest management, including manufacture and application of fertilizer, result in very little radiative forcing in comparison to the negative radiative forcing from using the increased forest growth for biomass substitution. The biggest single factor causing radiative forcing reduction is using sawlogs to produce wood material to replace energy-intensive construction materials such as concrete and steel. Another very significant factor is replacing fossil fuels with wood residues from forest thinning, harvest, wood processing, and post-use wood products. The fossil fuel that is replaced by the biofuels affects the reductions in GHG emission and radiative forcing, with carbon-intensive coal being most beneficial to replace. The climate benefits of fertilization are proportional to the increased rate of biomass production, in terms of shortened rotation lengths and increased harvest volumes. Our simulation is based on a typical forest stand in northern Sweden, although the benefits of fertilization will be less significant in central and southern Sweden (Sathre et al. 2010). This suggests that fertilization should be prioritized in areas where its effectiveness will be greatest.

There are uncertainties associated with this analysis. We have estimated separately the soil carbon stock changes due to fertilization, final harvest, and decay of biomass residues, but we have not considered potential interactions between the 3 effects. We have assumed that 1% of the applied N fertilizer will be released from the soil as N_2O, though the actual emission may be much lower (Nordin et al. 2009). Our material substitution analysis is based on a single case study of a wood-framed apartment building (Gustavsson et al. 2006), though GHG benefits vary with the type of wood substitution (Sathre and O'Connor 2010).

In our calculations of radiative forcing we have assumed relatively minor marginal changes in atmospheric GHG concentrations, such that radiative efficiencies and atmospheric decay patterns of the gases remain constant. However, significant increases in the atmospheric concentration of CO_2 can be expected during the 225-year simulation period (IPCC 2000). Increased atmospheric CO_2 concentration will decrease the marginal radiative efficiency of CO_2, but will also decrease the marginal atmospheric decay rate of CO_2. These will have opposite and therefore offsetting effects on radiative forcing, thus we expect this uncertainty to be minor (Moura-Costa and Wilson 2000). Taking into account the expected future trajectories of GHG concentrations would further reduce this uncertainty.

We estimated cumulative radiative forcing ($W \cdot s/m^2$) by simply integrating instantaneous radiative forcing (W/m^2) over time. This is a simplification, as we ignore the feedback effect that the accumulated energy will have on future outgoing radiation. Radiative forcing is a measure of the radiative imbalance given that atmospheric temperatures remain unchanged. In fact, the negative radiative forcing calculated in this study will reduce the heat energy accumulated in the earth system, potentially leading to less outgoing longwave radiation and eventually a restoration of radiative balance. Since instantaneous radiative forcing does not account for this feedback effect, our results may therefore slightly overestimate the actual reductions in cumulative radiative forcing.

In this analysis we do not consider potential radiative forcing due to albedo changes. Albedo is a measure of the reflectivity of a surface, and forested land generally has a lower albedo than cleared land, particularly during snow-covered winter months (Schwaiger and Bird 2010). Fertilization changes the temporal pattern of a stand's tree cover, due to the more rapid initial development and the shorter rotation period of a fertilized stand (see Fig. 3). Further research is needed to understand the potential albedo-related radiative forcing affects of forest fertilization.

Conclusions

In this study we have modelled the cumulative radiative forcing implications of forest fertilization and biomass substitution, with explicit consideration of the temporal aspects of GHG emissions and removals. We have found that forest fertilization can significantly increase biomass production. The potentials for material and energy substitution increase substantially due to the increased biomass production. The average carbon stock in tree biomass, forest soils and wood products all increase when fertilization is used. The additional GHG emissions due to fertilizer production and application are small compared to increases in substitution benefits and carbon stock. The radiative forcing of the 2 stands is identical for the first 15 years, followed by 2 years during which the fertilized stand produces slightly more cumulative radiative forcing. After Year 18 the instantaneous and cumulative radiative forcing are consistently lower for the fertilized forest system. Both stands result in negative radiative forcing, or cooling of the earth system. By the end of the 225-year simulation period, the cumulative radiative forcing reduction of the fertilized stand is over twice that of the non-fertilized stand. This suggests that forest fertilization and biomass substitution are effective options for climate change mitigation, as climate change is a long term issue.

Acknowledgments Authors gratefully acknowledge the support of the European Union, Swedish Energy Agency, Jämtland County Council, Sveaskog, SCA, and Norrskog.

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