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Forests on drained agricultural peatland are potentially large sources of greenhouse gases – insights from a full rotation period simulation

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Abstract

The CoupModel was used to simulate a Norway Spruce forest on fertile drained peat over 60 years, from planting in 1951 until 2011, describing abiotic, biotic and greenhouse gas (GHG) emissions (CO_2 and N_2O). By calibrating the model against tree ring data we obtained a "reference" model by which we were able to describe the fluxes and controlling factors over the 60 years. We discuss some conceptual issues relevant to improving the model in order to better understand peat soil simulations. However, the present model was able to describe the most important ecosystem dynamics such as the plant biomass development and GHG emissions. The GHG fluxes are composed of two important quantities, the forest carbon (C) uptake, $405 \, \mathrm{g \, C \, m^{-2} \, yr^{-1}}$ and the decomposition of peat soil, $396 \, \mathrm{g \, C \, m^{-2} \, yr^{-1}}$. N_2O emissions contribute to the GHG emissions by $0.5 \, \mathrm{g \, N \, m^{-2} \, yr^{-1}}$, corresponding to $56.8 \, \mathrm{g \, C \, m^{-2} \, yr^{-1}}$. The 60-year-old Spruce forest has an accumulated biomass of $164 \, \mathrm{Mg \, C \, ha^{-1}}$. However, over this period $208 \, \mathrm{Mg \, C \, ha^{-1}}$ GHG has been added to the atmosphere, which means a net addition of GHG emissions. The main losses are from the peat soil and, indirectly, from forest thinning products, which we assume have a short lifetime. We conclude that after harvest at an age of 80 years, most of the stored biomass carbon is liable to be released, the system having captured C only temporarily and with a cost of disappeared peat, adding CO_2 to the atmosphere.

1 Introduction

Peatlands contain around one third of the carbon (C) stored in global soils, which is equivalent to almost half that present in the atmosphere (FAO, 2012; IPCC, 2013). Undisturbed peatlands accumulate C as partially decayed vegetation, and the decay processes emit C in the form of carbon dioxide (CO_2) and methane (CH_4). Overall, the net greenhouse gas (GHG) balance of the photosynthesis and decomposition is generally positive, thus peatlands are considered to be C sinks contributing to an attenuation

of climate change (Gorham, 1991). However, when peatlands are drained for intensified land use, i.e. agriculture or forestry, the stored peat starts to decompose aerobically. The accelerated soil decomposition emits large amounts of CO_2, and, in contrast to CH_4 emissions, are greatly reduced, possibly even accounting for a net uptake of atmospheric CH_4 (Limpens et al., 2008). The decomposition also releases nitrogen, while another powerful GHG, nitrous oxide (N_2O), could also be produced, primarily through microbial nitrification and denitrification processes (Firestone and Davidson, 1989). Globally, peatlands cover only 3 % of the Earth surface among which 10–20 % of the total peatlands have been drained for agriculture or forestry, mainly in the boreal and tropical regions (FAO, 2012). However, these small areas emit around 6 % of the global annual anthropogenic GHG emissions (IPCC, 2013).

To date, a number of studies have investigated the size of GHG fluxes from managed peatlands with different land uses, together with their interactions with environmental factors e.g. (Kasimir-Klemedtsson et al., 1997; von Arnold et al., 2005a, b; Alm et al., 2007; Beek et al., 2010; Lund et al., 2010; Lohila et al., 2011; Ojanen et al., 2013). Several factors have been found to influence the size of the emissions, including the groundwater level (GWL), land use intensity, climate zones, and soil fertility (Klemedtsson et al., 2005; Drösler et al., 2008; Leppelt et al., 2014). In general, nutrient rich fens with deep GWL are larger GHG sources than ombrotrophic bogs with shallow GWL, while intensive land use in tropical/temperate regions have much higher emissions than extensive land use in boreal regions (Byrne et al., 2004). Peatlands in Europe used as grassland, agricultural land, peat cuts, and abandoned peat are generally found to be net GHG sources (Byrne et al., 2004; Drösler et al., 2008). However, forested drained peatland can be everything from a source to a small GHG sink due to the growing forest, where the net primary production (NPP) of trees and understorey vegetation balances the soil emissions (Drösler et al., 2008; Klemedtsson et al., 2008; Hommeltenberg et al., 2014). Previous flux measurement studies have also shown contradictory results. Measurements from Scandinavia and Great Britain have shown the NPP to compensate for the soil CO_2 release, and thus the forests to act as net

sinks (Hargreaves et al., 2003; von Arnold et al., 2005a; von Arnold et al., 2005b; Oja-nen et al., 2013). Hommeltenberg et al., (2014) also reported an afforested drained bog in southern Germany to be a net GHG sink; however, if the 44 year history of the forest were included in the analysis, then the so-called "long-term carbon balance", showed the forest to be an overall GHG source. von Arnold et al. (2005a) showed that account-ing for N_2O in the greenhouse budget calculation could shift drained birch peatlands from being minor GHG sinks into sources. This was also shown by Meyer et al. (2013) for a drained former agricultural peat soil with spruce forest, where soil N_2O emissions, in terms of global warming potential (265 times of CO_2 in a 100 year perspective, IPCC, 2013), offset half the net ecosystem exchange (NEE). Large N_2O emissions are most pronounced for fertile soils like former agricultural peatlands (Klemedtsson et al., 2005). So far most studies have only covered a few years at most. Consequently we still lack an understanding of the full GHG balance when viewed over the full forest rotation (Maljanen et al., 2010).

In the present study we aim to address this knowledge gap by exploring the GHG bal-ance for a Norway Spruce (*Picea abies*) forest on drained agricultural peatland (Skog-aryd Research Site: http://www.fieldsites.se/en/field-research-stations) over a full rota-tional time period. Since measurements are mostly short-term, and because it is not possible to directly upscale the measured fluxes to the entire forest rotation period (Drösler et al., 2008; Hommeltenberg et al., 2014), we chose a modeling approach based on emission data over five years and data on forest growth rate over 45 years for a Spruce forest on former agricultural peatland. This study forms a continuation of that by He et al. (2016), in which the process-based model "CoupModel" (Jansson, 2012) was calibrated to simulate the water, heat, and major C and N processes for the Skogaryd Research Site.

1.1 Conceptual model of drained peatland for forestry

When peatlands are drained for forestry or agriculture, resulting in a lower GWL, the aerobic soil volume increases (Fig. 1a). The previously water-logged peat soil then

decomposes aerobically, losing soil C stock and also causing a lowering of the soil surface (surface subsidence) (Eggelsmann, 1976; Hooijer et al., 2012). During the first few decades after planting, the development of plant roots and leaf area cover increases the transpiration rate, so deepening the GWL (Fig. 1b). In other words, a growing forest will, in part, help to keep the soil drained. However, drainage becomes less efficient with time due to subsidence and ditches becoming filled with litter and moss, all of which can lead to an increased GWL (Fig. 1c), which is why ditch maintenance is performed. For instance in Finland, forest peatlands used for timber production normally undergo drainage maintenance every 40 years (Minkkinen et al., 2002). After ditch maintenance the forest ecosystem restarts at the well-drained state (Fig. 1d), until the final clear-cutting when re-drainage has to be conducted. The entire cycle then starts again and can continue until all the peat is gone.

2 Material and methods

2.1 Site description

Data used for the present study were obtained from the Skogaryd research site, located in southwest Sweden (58°23′ N, 12°09′ E), which is part of the Swedish Infrastructure for Ecosystem Science (SITES, www.fieldsites.se). The drained peat area at Skogaryd was previously a fen, classified as mesotrophic peat with a peat depth of more than 1 m, according to the soil classification scheme suggested by Karlsson (1989). It was initially drained by ditches in the 1870s and then used for agriculture until 1951. Norway Spruce (*P. abies*) was then planted and the stand is now a mature mixed coniferous forest dominated by Norway Spruce (95 % by stem volume), with a sparse presence of Scots pine (*Pinus sylvestris*) and Silver birch (*Betula pubescens*) (Klemedtsson et al., 2010). The site has been intensively measured and monitored since 2006, providing abiotic and biotic data including CO_2 and N_2O fluxes that could be used to validate the

long-term model predictions. More detailed site description can be found in He et al. (2016), Klemedtsson et al. (2010), Meyer et al. (2013) and Ernfors et al. (2010).

2.2 Brief introduction to the CoupModel

The CoupModel (coupled heat and mass transfer model for soil-plant-atmosphere systems) is an updated version of the previous SOIL and SOILN model (Jansson and Moon, 2001). The main model structure is a one-dimensional, layered soil depth profile, in which the water, heat, and C and N dynamics are simulated based on detailed descriptions of soil physical and biogeochemical processes. The model is available at http://www.coupmodel.com/. A detailed description of the model, its parameterization and setup is given in He et al. (2016); here only the variables and parameters with different values are reported.

2.3 Model approach and design

The CoupModel conceptualizes the soil profile into a number of soil layers, where the soil's physical structure (defined by the measured water retention characteristics) and the drainage depth are assumed to be fixed over time (Fig. 1e and f). The subsidence of the soil surface and any variation in drainage (Fig. 1a–d) during the plant development years (1951 to 2011) cannot therefore be explicitly simulated. We thus make the following assumptions to simplify the system:

First, we assume the soil physical characteristics in 1951 to be the same as measured in 2006; the drainage depth to be constant during the simulated 60 years; and there to have been a continuous lowering of drainage ditch depth as the soil subsides, so keeping the distance between soil surface and GWL constant (Fig. 1e and f), with the drainage depth set to 0.5 m as in He et al. (2016). However, a range between 0.3 and 0.8 m was used to assess the model's sensitivity to long-term processes (Fig. 1e, see also Table 1).

Second, in order to define the initial soil C content in 1951, we use the soil C measurements made at Skogaryd in 2007 and back-calculated to 1951 by adding an annual peat loss of $2.6\,\mathrm{MgC\,ha^{-1}\,yr^{-1}}$ from 1951 to 2007. This annual loss was taken from the recent IPCC wetland supplement (IPCC, 2014), where it represents the emission factor for forest on drained nutrient-rich peatlands in the temperate region. The model's sensitivity to this initial condition can be assessed by varying IPCC emission factors (EF's) between 2 and $3.3\,\mathrm{MgC\,ha^{-1}\,yr^{-1}}$ when calculating total soil C in 1951. In addition, an extremely large initial soil C is also used in the sensitivity analysis which was back-calculated using the highest peat decomposition rate of $6.3\,\mathrm{MgC\,ha^{-1}\,yr^{-1}}$ (Meyer et al., 2013) measured at Skogaryd during 2008 (Table 1).

Third, the soil C/N ratio in 1951 was assumed to be the same as that measured in 2006, and the N deposition rate was also assumed to be constant as in He et al. (2016) during the entire simulated period. The model's sensitivity to this was tested by varying the initial soil C/N ratio between 20 and 45, the latter being the value measured at a nearby un-drained peatland near Skogaryd.

Fourth, similarly to the previous calibration study, the model only simulates the C and N dynamics in the uppermost 1 m depth of soil.

The model was initially run with the calibrated single parameter representation using the same mean parameter values as used by He et al., (2016). However, each calibrated parameter has a range of possible values, its so-called posterior distribution, which we varied in order to fit the model results to the 45 year (1966–2011) tree-ring-derived biomass data and extended abiotic data (2006–2011). We call the model parameterized to fit those data the "reference" model simulation, and used it for sensitivity analysis by varying the drainage depth, initial soil C, as well as the initial soil C/N ratio.

2.4 Tree ring sampling and data processing

The previous calibration of the CoupModel mainly focused on the soil processes while plant development was less emphasized (He et al., 2016). In order to calibrate the model results of the plant biomass development, we acquired incremental core sam-

ples from the Spruce trees in Skogaryd during spring 2013, to estimate forest biomass. In total, 25 samples were obtained from randomly chosen trees. The cores were taken at breast height (1.3 m above ground). The annual growth rings in the tree cores were cross-dated according to standard dendrochronological methods (Stokes and Smiley, 1968) to assign an exact calendar year of formation to each ring. Tree ring width data were obtained by analysis of scanned images of carefully surfaced cores using the software CooRecorder (cybis.se). The annual variation in height growth was modeled with the Korf's function using cumulative radial growth during the previous years, cal-ibrated by extensive inventory data, collected in 2010 (Meyer et al., 2013). Since the inventory data lacked information concerning trees with a diameter smaller than 10 cm, and because the sample depth of trees decreases back in time, the forest biomass calculations were only considered to be valid from 1966 (a date when all trees had a diameter above 10 cm and the sample replication was complete). The forest biomass was calculated for stem, living branches, dead branches, stumps and roots including fine roots, following the allometric equations (Marklund, 1988) for Spruce in Minkkinen et al. (2001) and Meyer et al. (2013), using the inputs of measured annually resolved radial growth and modeled annual longitudinal growth. The total biomass of the tree stands was calculated as a sum of the average biomass of the individual trees, where the planting density was assumed to be 3000 trees ha^{-1}, which was a typical planting density during the 1950s in Sweden (Drossler et al., 2013). A thinning was conducted by the land owner in 1979 when the number of trees was reduced to a ca. 1000 trees ha^{-1}, according to the survey data presented in Meyer et al. (2013). Using these tree ring biomass data, the thinning management was estimated to have removed 72 % of the plant biomass, which we assumed to have been taken only from the Spruce for-est. The proportion of the thinning for each part of the forest was made according to general Swedish forest management guidelines and the model parameter set was thus selected according to Svensson et al. (2008). In addition, a heavy storm hit Skogaryd forest in 2010 and blew down 10 % of the tree biomass. The fallen trees were removed

from the experimental site after the storm event. Therefore an additional harvest was included in the CoupModel to simulate this removal of storm-fallen biomass.

2.5 Data for model forcing

To drive the model, we used daily mean meteorological data (1961–2011) from Såtenäs station (58°44′ N, 12°71′ E), (www.smhi.se) situated approximately 60 km east of Skogaryd. Precipitation, air temperature, wind speed and relative humidity data from Såtenäs were strongly correlated ($R^2 > 0.8$) with those from Skogaryd from 2006 to 2011, and were of similar magnitude. Another driving variable needed in CoupModel is the global short wave radiation. As these data are not available from Såtenäs station, they were deduced by the model from the potential global radiation and atmospheric turbidity, using the measured total cloud-cover fraction (for more details see http://www.coupmodel.com). Since meteorological data were only available from 1961, the meteorological data from 1961 to 1971 were duplicated to represent the climate between 1951 and 1961.

2.6 GHG budget compilation

For a total GHG budget of the system we include harvest removal and products. We assumed that the material removed due to the thinning management in 1979 and the storm harvest in 2010 was used for paper production, as is common practice in Sweden (Swedish Forest Agency, 2005). We therefore used the emission factors suggested in the IPCC guidelines (IPCC, 2006), in which paper is assumed to decay exponentially with a half-life of 2 years.

3   Results

3.1   Model performance

3.1.1   Plant and soil development from 1951 to 2011

The simulated tree biomass dynamics during the 60 years agrees well with the estimated tree biomass from radial growth observations beginning in 1966. After an initial phase of slow growth during the establishment of the Spruce trees' leaf area, growth increased almost linearly (Fig. 2d). The slow establishment of the Spruce in the first decade was probably due to competition from grasses and other field vegetation. The Spruce plants gradually increased their leaf cover until a closed canopy formed in the 1980s with a maximum leaf area index (LAI) of around 6, which was similar to field measurements (Fig. 2b). The simulated annual average plant growth over the whole period is $405\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ with the maximum growth rate of $820\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ in 1974 (Fig. 2c). However, the reference model generally shows underestimation of the plant growth during the first 20 years, which is probably due to the model's difficulty in precisely simulating the competition between the Spruce layer and the understorey layer. The underestimation of Spruce tree growth for the first 20 years suggests an overestimation of the understorey layer in the reference model. The LAI and the NPP of Spruce generally follow the dynamics of the plant's ability to intercept radiation (Fig. 2a); however, the model slightly overestimates annual tree growth from the 1970s to the 1990s, and underestimates it from 1996 until 2011 (Fig. 2c). Furthermore, the large increase of simulated plant growth observed in 2006 was not observed in the tree ring data. The total tree biomass in 2011 is modeled to be $16.4\,\mathrm{kg\,C\,m^{-2}}$, which is very similar to the biomass estimated from the tree ring data, $16.2\,\mathrm{kg\,C\,m^{-2}}$ (Fig. 2d). The thinning conducted in 1979 removed $6.8\,\mathrm{kg\,C\,m^{-2}}$ plant biomass, and the storm in 2010 caused an additional removal of $1.8\,\mathrm{kg\,C\,m^{-2}}$; these quantities were used for indirect emission calculations (Fig. 2d). The modeled amounts of leaf and root biomass in 2007 also match estimations using allometric equations reported by Meyer et al. (2013).

The modeled and estimated values for leaf biomass were 0.95 and 1.06 kg C m^{-2}, respectively, and the values for total roots were 2.9 and 3.0 kg C m^{-2}, respectively. The modeled value for Spruce stem biomass was 13.1 kg C m^{-2}, which was higher than the estimated 11.2 kg C m^{-2}. This discrepancy may be explained by the estimated total plant biomass by Meyer et al. (2013) being smaller than that estimated from tree ring data. The maximum biomass of understorey vegetation was simulated to be around 2 kg C m^{-2} 10 years after planting, but it decreased gradually thereafter (Fig. 2e).

The total soil C within the top 1 m depth at the end of the simulated period generally matches the observed data and only accounts for 39 % of initial values estimated for 1951 (Fig. 3). Over the whole of the simulated 60 years, the accumulated soil litter respiration almost equaled that of the soil peat (which was treated as humus in the model). The resistant litter (called humus formation in the figure) only accounts for ca. 20 % of the total plant litter input to the soil, and the total losses of the soil C are mostly due to losses from respiration of soil peat. Before 1965, the total soil C losses are mainly through soil peat respiration, which is larger than the total plant litter input, indicating there to be a large soil C imbalance at the beginning of the rotation (Fig. 3).

3.1.2 Comparing reference model output with observational data from 2006 to 2011

The simulation beginning in 1951 using the reference model, fitted to the tree ring data, showed a good fit for the period 2006 until 2011 when comparing simulated and measured GWL, total net radiation and soil temperature data. The linear correlations between the simulated and measured data were all above 0.7 with the mean errors close to zero (Fig. 4). Discrepancies were found in May 2010, when the measured GWL peaked (high GWL) and which the model underestimated (Fig. 4c), and during summers and autumns when the model overestimated both radiation and soil temperature (Fig. 4a and b). Besides being a reasonable description of abiotic factors, the model results were also similar to observed data from 2007 to 2008 on NEE flux, both

in terms of seasonal pattern and magnitude (Fig. 4d). However, the simulations seemed to slightly underestimate the CO_2 uptake during summertime and overestimate the flux in the autumn (Fig. 4). The model performance for N_2O emissions was generally similar as in the previous calibration study (He et al., 2016).

3.2 GHG balance

3.2.1 Annual NEE and N_2O from 1951 to 2011

The annual 60 year NPP for the Spruce forest, including biomass and litter, was on average $683\,g\,C\,m^{-2}$ with less than $100\,g\,C\,m^{-2}$ during the first 10 years after planting, and with a value that fluctuates around $1000\,g\,C\,m^{-2}$ over the last 40 years (Fig. 5b). In the first decade the peat decomposed at its maximum rate of around $700\,g\,C\,m^{-2}$. However, this decreased to about $150\,g\,C\,m^{-2}$ at the end of the simulated period, with an annual average of $396\,g\,C\,m^{-2}$ (Fig. 5c). NPP and peat decomposition are the two major components of NEE, in which the system showed itself to be both a sink and a source during the first 29 years (1951 to 1980), but thereafter to be a continuous CO_2 sink (Fig. 5d).

According to our simulation, the forest and soil system emits $316\,g\,C\,m^{-2}$ CO_2 to the atmosphere in the first year after planting, quickly changing to an uptake of $150\,g\,C\,m^{-2}$ in the following year, which coincides with the colonization of the understorey vegetation as shown by the modeled biomass development (Fig. 2e). From 1953 to 1962, the forest system behaves as a major CO_2 source, emitting $343\,g\,C\,m^{-2}\,yr^{-1}$. However, during 1963 to 1965 the system switches and acts as a minor CO_2 sink with an average flux of $61\,g\,C\,m^{-2}\,yr^{-1}$ before it again returns to being a CO_2 source from 1966 to 1971, with an average flux of $78\,g\,C\,m^{-2}\,yr^{-1}$, except for another brief period as a minor sink of $86\,g\,C\,m^{-2}$ in 1968. From 1972 to 1979, the forest ecosystem acts as a sink of CO_2 with an average flux of $172\,g\,C\,m^{-2}\,yr^{-1}$. The thinning management in 1979 had a large impact on the NEE which changed the system to that of a source of $825\,g\,C\,m^{-2}\,yr^{-1}$

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for the following year. After 1981, the forest ecosystem was a continuous sink of CO_2 with an average NEE of $327\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ (Fig. 5d).

Surprisingly, the model does not predict the largest N_2O emissions in the early period when the peat decomposition was high. Instead it predicts most of the N_2O to be emitted from 1966 to 1988, a period concomitant with the rapid increase of Spruce NPP and the period of thinning. Over the 60 years, the simulated annual N_2O emission varied between less than 0.01 to $4\,\mathrm{g\,N\,m^{-2}\,yr^{-1}}$, with an average of $0.5\,\mathrm{g\,N\,m^{-2}\,yr^{-1}}$ (Fig. 5e).

3.2.2 Overall GHG balance from 1951 to 2011

Over the full 60 year time period the forest trees acted as a C sink and the soil as a source, of fairly similar size (Fig. 6). This could be viewed as a relocation of C from the soil to the trees, since our model predicts the total soil C loss to be $590\,\mathrm{Mg\,CO_2}$ $\mathrm{ha^{-1}}$ over the 60 years, while plant growth (including spruce forest and understorey vegetation) sequesters $602\,\mathrm{Mg\,CO_2\,ha^{-1}}$. The accumulated NEE shows the young forest ecosystem to be a net CO_2 source, and it is not until 1990, 39 years after the forestation, that the ecosystem reaches zero CO_2 emission before becoming a continuous sink. If the N_2O emissions during the 60 year rotation period are included, and we take the most commonly used 100 year time horizon from IPCC for calculating N_2O greenhouse gas potential ($1\,\mathrm{g\,N_2O} = 265\,\mathrm{g\,CO_2}$ equivalent global warming potential, IPCC, 2013), the source strength of the forest ecosystem increases and the system first reaches GHG neutrality in 1998.

However, if the fate of the biomass removed as thinnings, which usually goes into paper production, is included, these indirect CO_2 emissions switch this extended system (from the production site to the fate of the products) from a GHG sink to a GHG source of $162\,\mathrm{Mg\,CO_2\,ha^{-1}}$ by the end of the simulation (Fig. 6). Soon, the whole forest will be harvested releasing a large part of the captured carbon, $16.4\,\mathrm{kg\,C\,m^{-2}}$ (Fig. 2d), into

the atmosphere again; and if everything were released from these soils there would be 763 Mg CO_2 ha^{-1} released over a period of 60 years.

3.3   Model sensitivity

The simulated soil peat decomposition is generally more sensitive than the accumulated plant biomass if only the single factor of either the initial soil C or increasing drainage depth is changed (Table 1). On the other hand, the accumulated plant biomass is more sensitive to changes in the soil C/N ratio or shallower drainage depth (Table 1). The NEE and N_2O sizes are also very sensitive to these variations, the NEE becoming a CO_2 source when using the largest initial soil C, when the peat decomposition rate becomes larger than the accumulated plant biomass. The model sensitivity also shows higher N_2O emissions under shallower rather than deeper drainage (Table 1). When these various factors were combined, the peat decomposition varied by −53 % to +74 %, being largest when the combination was deep drainage with the largest initial soil C, and a low initial soil C/N ratio. The accumulated biomass varied between −63 and +9 %, being smallest when the combination was shallow drainage with a low initial soil C and a large soil C/N ratio. However, the overall total GHG emissions, including the harvested biomass and its associated CO_2 losses, when compared to the reference model, increased by 88 to 267 % (Table 1), suggesting that the total GHG balance was still a source to the atmosphere.

4   Discussion

4.1   Comparison of our simulated results with observational and published data

The GHG balance over a rotational period for forestry on drained peatland is mainly determined by two large numbers *viz.* those important quantities relating to plant growth

and peat decomposition. We therefore first discuss the validity of these two variables by comparing our simulated results with values published in the literature.

4.1.1 Plant growth

Our simulated Spruce growth at $405\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ was higher than the normal growth rate of 162 to $270\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ in southwest Sweden, but lower than the potential growth rate of 472 to $607\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ under experimentally optimal nutrient conditions (Bergh et al., 2005). This high growth rate can be explained by the fertile soil at the Skogaryd site, which was drained fen before it had been used for agriculture, and then forestry. The high rate of nitrate leaching, estimated at $3.6\,\mathrm{g\,N\,m^{-2}\,yr^{-1}}$ also suggests that nutrients are not likely to be limiting. That the forest growth at this site is close to maximum has also been demonstrated in a modeling study by Tarvainen et al. (2013) who showed that if canopy N content was increased by 30 % canopy C uptake would only increase by 2–4 % and none of the 37 nutrients tested would directly limit photosynthesis. The very small increase of plant growth (+9 %) in our model sensitivity analysis, which obtained when more deeply drained soil, a larger initial soil C, and a small C/N ratio were assumed, can also be explained by the already high fertility at the site, so any extra nutrient availability would have a negligible impact. Our simulated understorey vegetation was small during most of the simulated years; however, it dominated the organic matter dynamics and GHG fluxes in the first two decades after plantation, a finding similar to that of Laiho et al. (2003).

4.1.2 Soil CO_2 and N_2O fluxes

Our simulated peat decomposition rate of $396\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ during the period 1951 to 2011 is lower than the value measured in 2008, which was $630\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ (Meyer et al., 2013). However, this high peat decomposition rate could be attributed to an inter-annual weather variation, which is corroborated by the high plant growth measured in 2008, at 830 ($\pm$390) $\mathrm{g\,C\,m^{-2}\,yr^{-1}}$. Our simulated N_2O emission, 0.65 ($\pm$0.1) $\mathrm{g\,N\,m^{-2}\,yr^{-1}}$

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during 2007 to 2009 is similar to the observed data, at 0.71 (±0.59) $g\,N\,m^{-2}\,yr^{-1}$. However, an extended measurement period at Skogaryd (2006–2011) reveals the possibility of lower annual N_2O emissions of 0.38 (±0.12) $g\,N\,m^{-2}\,yr^{-1}$ (Holz et al., 2015). During these years, our predicted level of emissions was 0.50 (±0.17) $g\,N\,m^{-2}\,yr^{-1}$. Our simulated CO_2 and N_2O fluxes are therefore generally comparable with the measured data.

When we compare our simulated peat decomposition and N_2O emissions with measured flux data from afforested drained peatland published in the literature, they are generally within the reported ranges (Table 2). However, when compared with the IPCC EF's for temperate drained nutrient-rich forest soil, which are given as 260 (200 to 330) $g\,C\,m^{-2}\,yr^{-1}$ for CO_2 and 0.28 (−0.06 to 0.61) $g\,N\,m^{-2}\,yr^{-1}$ for N_2O (IPCC, 2014), our simulated values were found to be larger. This could be explained by the higher soil fertility at the Skogaryd site and also a deeper GWL (mean of 0.55 m during the simulated 60 years), compared to what pertained at those sites used for constructing the IPCC EF's. That the GWL is of crucial importance for emission levels for drained peat soils has also been shown by Couwenberg et al. (2011) and Leppelt et al. (2014). This could justify our assumption that our somewhat high estimates were due to deep and long-lasting drainage. However, the unexpectedly low simulated N_2O emission in the first years after planting could be explained by a high N uptake by the understorey vegetation, probably dominated by grasses, making less N available for nitrification and denitrification.

4.2 Challenges of modeling long-term dynamics of an organic soil

Overall our modeling application indicates, given a few assumptions, that the Coup-Model is generally able to simulate the decadal-scale dynamics of the drained organic soils used for forestry. However, our modeling exercise also reveals that there are some issues which still need to be more explicitly accounted for when simulating organic soils and which require further model development.

4.2.1   A need for explicitly specifying the nature of soil organic matter

The CoupModel conceptually divides the soil organic matter (SOM) into two pools called soil litter (fresh plant detritus) and humus, which constitute a fast and a slow decomposing pool, respectively (Johnsson et al., 1987). The model assumes soil litter to decay, so producing both CO_2 and adding a resistant fraction into the humus pool (Johnsson et al., 1987). This conceptual model was developed from mineral soils and has been shown to work well for forest mineral soils e.g. (Svensson et al., 2008; Wu, 2013). However, for organic soils, because there is no explicit peat pool in the model, we have had to assume the peat to comprise an unknown mixture of the fast and the slow pool. In the present study we have assumed the initial values of SOM as only representative of the slow pool. Thus our simulated slow pool may be composed differently at the end of the simulation compared to that at the beginning when the Spruce was planted, since there is a continuous addition of resistant organic matter in the form of Spruce litter, which changes the substrate quality over the simulation period. However, the quality of a resistant litter fraction differs from that of historically stored peat, both in terms of quality of the organic matter and the decomposition kinetics. Although most existing models do not explicitly specify the nature of the organic matter (Smith et al., 1997), they can still simulate the total organic matter dynamics fairly well over a relatively short period, a situation which we also found here (Fig. 3). Metzger et al. (2015) found that the CoupModel could capture major C fluxes and the ecosystem dynamics when applied to five European treeless peatlands. However, they also pointed out that the total C flux was mainly determined by the decomposition coefficients of the total SOM. The decomposition coefficients for the fast and slow pool can be obtained by calibrating the model coefficient against the measured fluxes as we did in our previous study (He et al., 2016). However, if the decomposability of the substrates, i.e. the humus, changes over time due to the continuous addition of resistant litter, then the decomposition coefficient must also change over time accordingly. However, this is seldom accounted for. In order to understand the long-term dynamics of organic mat-

ter, which might differ in origin and components, a more precise consideration of the changes of soil organic matter characteristics would be helpful.

4.2.2 Modeling physical changes of peat soil

For mineral soils in which the physical structure of the soil does not normally change over time, the CoupModel works well by assuming the soil layer profile to be fixed over time (Jansson and Karlberg, 2011; Jansson, 2012). However, this is not the case for organic soils where the soil structure is mainly built by soil organic matter, which gradually disappears through decomposition. Thus the soil's physical characteristics change over time, e.g. the pore structure, which could change the soil hydraulic conductivity and preferential flows (Kechavarzi et al., 2010). Moreover, decomposition makes the topmost meter of soil almost disappear after a forest rotation time of eight decades, resulting in surface subsidence (Minkkinen and Laine, 1998; Leifeld et al., 2011; Hooijer et al., 2012). This causes the GWL distance to the soil surface to diminish and so requires further drainage to be created. This process has not so far been implemented in the CoupModel, which cannot currently account for surface subsidence, mainly due to the model lacking a feedback coupling between the soil's biological and physical properties. The model simulates the water and heat flow in its soil physical subroutine and then links this to the biochemical processes by response functions of water moisture and soil temperature. Hence, there is no feedback to the soil physical processes arising from any decomposition or other change of the soil organic matter.

Root development also follows a simple static approach. The disappearance of the soil and the deepening of the root system are therefore not considered in the simulation as it is currently configured. All these processes remain a major challenge when applying the CoupModel to the long-term dynamics of a forest ecosystem on drained peatland. In the present study, the system has been simplified by assuming a fixed drainage depth, whereas a range of values was used to quantify the model's sensitivity. The lower end of the range was chosen to be a drainage depth of 0.3 m, since this has been suggested to be the minimum requirement to sustain forest productivity

on drained peatlands (Sarkkola et al., 2010; Ojanen et al., 2013). The higher drainage level, 0.8 m, was set according to general forest management practices and also took into consideration the fact that our simulated soil depth only reaches a maximum of 1 m. The variation of the drainage depth had a considerable impact on the soil peat decomposition, as shown by the model sensitivity analysis, which in turn highlights the need, when developing future models, to explicitly account for these processes when performing long-term simulations.

4.2.3 Initial soil C, N and soil C/N ratio

A major difficulty in the simulation was the unknown initial soil conditions. We chose to use the EF's from the IPCC wetland supplement (IPCC, 2014), which compiles up-to-date observational data from similar sites under temperate climate conditions. Another alternative could be to use the subsidence rate to calculate the soil C losses, which has been applied in other published studies e.g. (Leifeld et al., 2011; Hommeltenberg et al., 2014). By taking the subsidence rate of $0.01 \, \mathrm{m\,yr^{-1}}$, suggested for German and Swedish drained fens (Berglund and Berglund, 2010; Leifeld et al., 2011; Hommeltenberg et al., 2014), the subsidence from 1951 to 2007 is 0.56 m. Assuming the same bulk soil density of $0.23 \, \mathrm{g\,cm^{-3}}$ as measured in 2006 and analogizing the measured total soil C in the upper 0.5 m in 2007, which was $11.6(\pm1.6) \, \mathrm{kg\,C\,m^{-2}}$ (Meyer et al., 2013), the estimated soil losses during the period from 1951 to 2007 would be $13 \, \mathrm{kg\,C\,m^{-2}}$, which is equivalent to a loss of $230 \, \mathrm{g\,C\,m^{-2}\,yr^{-1}}$, and falls within the range of the IPCC EF's. The convergence of the C losses calculated by these two methods suggests the back-calculated initial C in the present study to have been acceptable. However, the variation of the initial soil C in our sensitivity analysis shows there to be a direct impact both on peat decomposition and on plant growth. Compared to the reference model, the combination of a small initial soil C, a large soil C/N ratio, and a shallow drainage, gives a larger reduction in plant growth than in peat decomposition, which is why the overall emissions of GHG increase. These initial settings can probably

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be attributed to a smaller N availability than in the reference model, resulting in a lower rate of plant growth.

4.3   GHG balance for the forest ecosystem

Our modeling indicates forest on drained agricultural peatland to be a strong net CO_2
source for the first 39 years of the forest rotation which changes into a CO_2 sink thereafter due to a large amount of tree growth (Fig. 6). This means that, despite soil respiration being high, the high growth rate of forest over 60 years compensates for most C losses. Meyer et al. (2013) also showed the forest ecosystem in Skogaryd to be an overall GHG sink (4.1 Mg CO_2 eq ha^{-1} yr^{-1}) in 2008, a year when the plant growth rate was at its maximum, thus offsetting the high rate of peat decomposition. Our findings are also generally in line with the few previous field investigations conducted on afforested drained agricultural peatlands where Mäkiranta et al. (2007) and Lohila et al. (2007) found a 30-year-old Scots pine forest on drained agricultural bog to be, overall, a small source of CO_2 (50 g C m^{-2} yr^{-1}), which was explained by a small leaf area index (varying between 0.7 and 2 during the observational period). Another study by Hommeltenberg et al. (2014), reported an afforested drained bog in Germany, previously used for agriculture, to emit 500 g C m^{-2} yr^{-1}. By combining eddy covariance measurements and biometric estimation, they concluded it to be a major CO_2 source, emitting a total of 134 Mg C ha^{-1} over the last 44 years. However, their short-term measurements (2010–2012) also indicated that forest growth offsets peat decomposition, a result similar to our own reported in the present study. However, since they found a higher rate of peat decomposition, their site was a C source over a longer period (44 years) than our study (39 years).

Growing forests on drained peat is done at the cost of the soil peat, which has generally accumulated slowly during the last millennia (the last four thousand years in Skogaryd). The soil loss and the forest gain can be viewed as a "relocation" of the peat carbon into timber carbon, and when the forest growth term is larger than the soil loss, the system has been interpreted as being an overall sink (Meyer et al., 2013; Hom-

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meltenberg et al., 2014). However, when we combine our simulation with the fate of the forest products, we can show the forest system to be a net GHG emitter during a 60 year period, and more so if the full standard forest rotation period is considered, which in this part of Sweden extends to 80 years (Fig. 6). After 80 years, there are only two options: either continue to keep the land forested or harvest it. Keeping the forest will probably result in a declined growth rate over time, as has already been shown in the simulated period from 2011 to 2031 (Fig. 6) and the carbon stock would keep increasing, albeit slowly, but certainly not for thousands of year, as it does during the formation of peat. Sudden fires would also release the C stored in the forest biomass into the atmosphere. The peat soil will continually decompose as long as it is kept aerated by a living transpiring forest. If, however, the forest is harvested, the bulk biomass will soon release its C when burnt as fuel for energy production. A better alternative would be the use of the timber to construct wooden buildings (Gustavsson et al., 2006). However, even then, most buildings do not last more than a century and only a few buildings are functional for longer periods. Most of the harvest products will also soon be burnt, similarly releasing their stored C as we have shown occurs with paper products. These larger indirect emissions would shift the system from an overall sink of GHG into a large source (Fig. 6). One alternative use of the biomass could be as biochar in agricultural soils (Ojanen et al., 2013), which potentially could shift the system into an overall GHG sink. However, we think this alternative to be somewhat peculiar, since it is just moving C around, releasing it from peat and storing it in agricultural soils, and it is not clear for how long char persists. Additionally, there are some other direct and indirect GHG sources that become apparent during the full forest rotation period which we have not accounted for, such as methane emissions in drainage ditches and loss of dissolved organic C or particulate organic C. However, these contributions to the overall GHG balance are in general of minor importance and thus not likely to alter the overall picture (Meyer et al., 2013; Hommeltenberg et al., 2014). In summary, the overall message is that a forest rotation on fertile drained peat

soil has a long-term GHG cost, never reaching a balance, and thus the wood products produced on peat soil cannot be regarded as renewable products.

5   Conclusions

Our simulation study shows that the GHG fluxes in a forested drained peatland are composed of two important quantities: C uptake by forest growth, and C losses from the soil. By fitting the CoupModel to growth in tree biomass, up-scaled from radial tree-growth observations, we obtained a "reference" model by which we were able to describe the C and N fluxes over 60 years. We show that the forest C growth is tightly coupled to soil C losses, and if the forest is harvested and used, there will be only losses. Our model sensitivity conducted by this study also provides evidence that a wide range of drainage depth, site fertility and initial soil C does not change the results. Of course, there are many other factors not considered in our study, which could potentially further influence the results. Thus, forests on drained agricultural peatlands are potentially large sources of GHG to the atmosphere. This conclusion, however, comes with some caveats regarding the model configuration and its need for further improvement.

In Sweden, forests on drained peatland cover 1.7 Mha (Maljanen et al., 2010; von Arnold et al., 2005a) of which 0.4 Mha has a level of fertility comparable to the soil in the present study. These forests emit around 10 Mg CO_2 eq ha^{-1} yr^{-1}. Thus these fertile, drained peat soils emit 4 Mt CO_2 eq yr^{-1}, which is equivalent to 10 % of the emissions coming from all other sectors in Sweden except LULUCF sources. From a climate change perspective, forested drained peatlands can be seen to play an important role, so highlighting the need for alternative actions, such as the rewetting of peatlands or other nature conservation measures. However, these measures need support from policy makers since landowners often only recognize revenues from forest production, not the cost of GHG emissions.

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Table 1. Model sensitivity: comparisons with the reference model.

Variables	Drainage depth (m)		Initial soil C ($g\,C\,m^{-2}$)			Initial C/N ratio (–)		Combination 1	Combination 2
	−0.3 (1)	−0.8 (2)	$22\,770^{1}$ (3)	$30\,050^{2}$ (4)	$46\,850^{3}$ (5)	20 (6)	45 (7)	(1) + (3) + (7)	(2) + (5) + (6)
Accumulated plant biomass	−23 %	−2 %	−4 %	1 %	6 %	3 %	−24 %	−63 %	9 %
Peat decomposition	−16 %	4 %	−14 %	12 %	65 %	3 %	−17 %	−53 %	75 %
NEE[4]	−67 %	−25 %	11 %	−32 %	−202 %	9 %	−86 %	−113 %	−229 %
N_2O emission	379 %	−69 %	−22 %	13 %	92 %	21 %	−59 %	60 %	−7 %
Indirect CO_2 emission	−18 %	−4 %	−3 %	1 %	6 %	3 %	−12 %	−54 %	11 %
NEE + N_2O + indirect CO_2 emissions	303 %	−24 %	−32 %	46 %	298 %	9 %	39 %	88 %	267 %

[1,2,3]: Back-calculated initial soil C using the reported range of IPCC EF's 2, 3.3, and 6.3 $Mg\,C\,ha^{-1}\,yr^{-1}$ respectively.

[4]: Positive change of NEE means the forest ecosystem sequesters more atmospheric CO_2 than the reference model; negative change means sequencing less atmospheric CO_2 or a possible source to the atmosphere.

Table 2. A comparison of soil peat CO_2 and N_2O emissions in the present study with values published in the literature.

Soil CO_2 flux ($gCm^{-2}yr^{-1}$)	Soil N_2O emissions ($gNm^{-2}yr^{-1}$)	Ecosystem type	Country	References
190–1000		Forestry-drained boreal peatland	Finland	Ojanen et al. (2013)
109–1200	0–1.9	Forest soils and other vegetated sites on deep peat	UK and other European Countries	Morison et al. (2012)
125–260[1]		Forestry-drained peatland	Finland	Minkkinen et al. (2007)
700		Grassland on agricultural fen peat	Germany	Kluge et al. (2008)
1405	1.94 (0.67)	Highly fertile drained peatland for forestry with low soil pH	Sweden	Weslien et al. (2009)
452	0.3	Afforested drained lowland raised peat bog	UK	Yamulki et al. (2013)
123–259[1]	0.02–0.57	Drained organic soils for deciduous and coniferous forests	Sweden	von Arnold et al. (2005a, b)
396	0.5	Drained forested agricultural peatland	Sweden	This study

[1]: Calculated by assuming 50 % of measured soil respiration to have originated from root-based activity.

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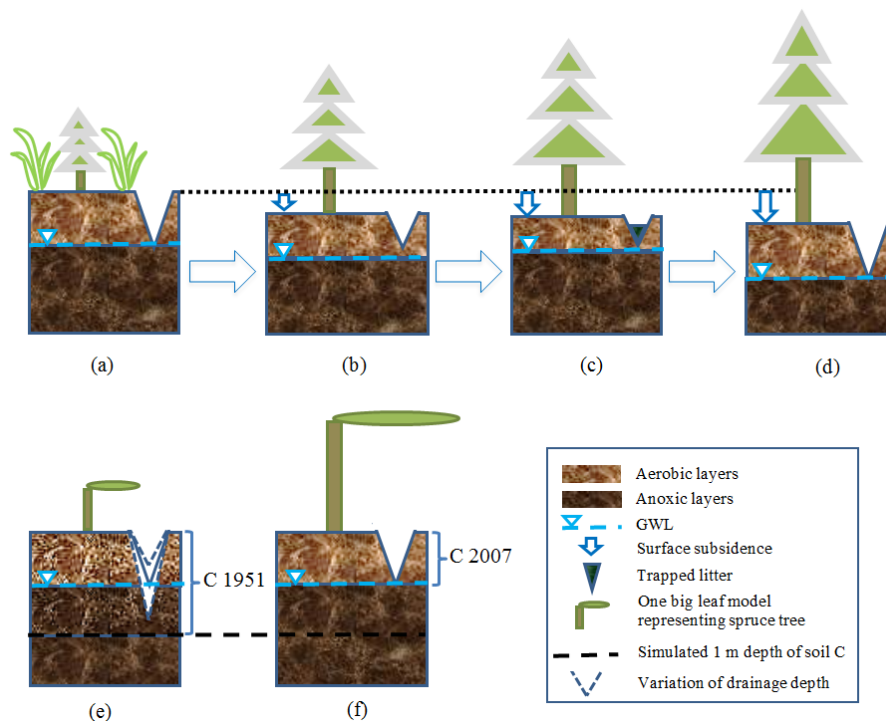
Figure 1. Conceptual representation of the dynamics of plants and peat soil development over a forest rotation period. The upper figures **(a–d)** represent the conceived reality and **(e)** and **(f)** represent the CoupModel conceptualization. For all the figures, Spruce tree and understorey vegetation, e.g. grasses are considered but for clarity, understorey vegetation is only shown in **(a)**. "C 2007" in **(f)** represents the measured total soil C in the upper 0.5 m of the soil profile in 2007, and "C 1951" is the total soil C in the upper 1 m of the soil profile, as back-calculated from the equation: "C 2007" + (2007–1951) × IPCC EF's. Any variation of climate during the forest development in this conceptual figure is not considered.
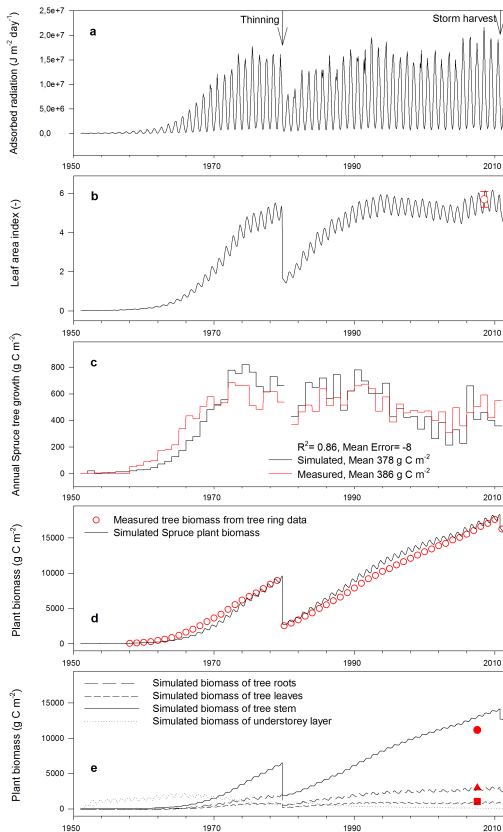
Figure 2. (a) Simulated (black line) Spruce adsorbed radiation; **(b)** simulated and measured (red hollow circle) leaf area index; **(c)** annual Spruce tree growth rate; **(d)** total plant biomass; **(e)** plant biomass for different components. In **(e)**, the solid red symbols show the calculated plant biomass of leaf biomass, root and stem biomass using the allometric function given by Meyer et al. (2013).
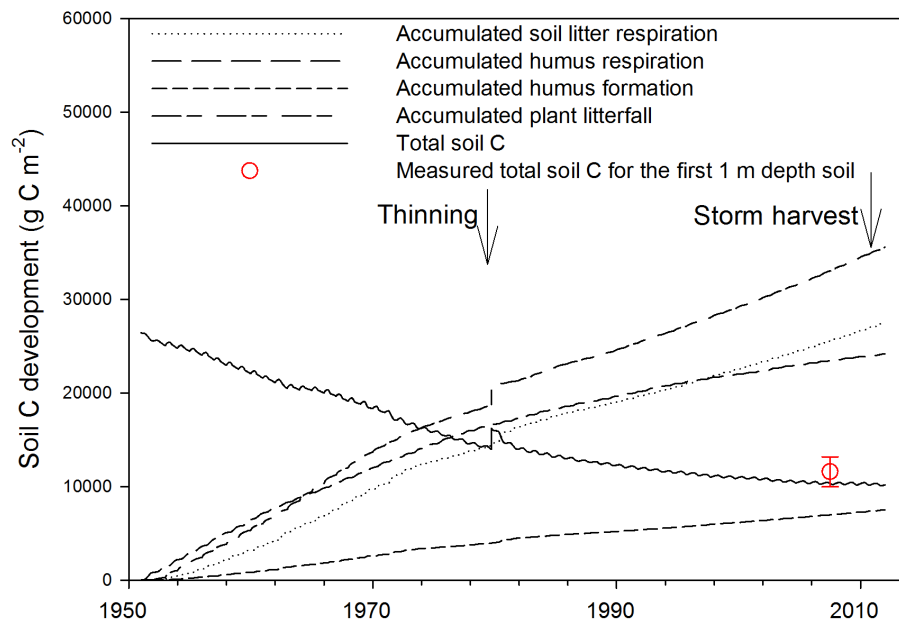
Figure 3. Simulated major soil C pool development from 1951 to 2011. The red circle shows the measured total soil C in 2007 (±95 % confidence intervals) by Meyer et al. (2013).

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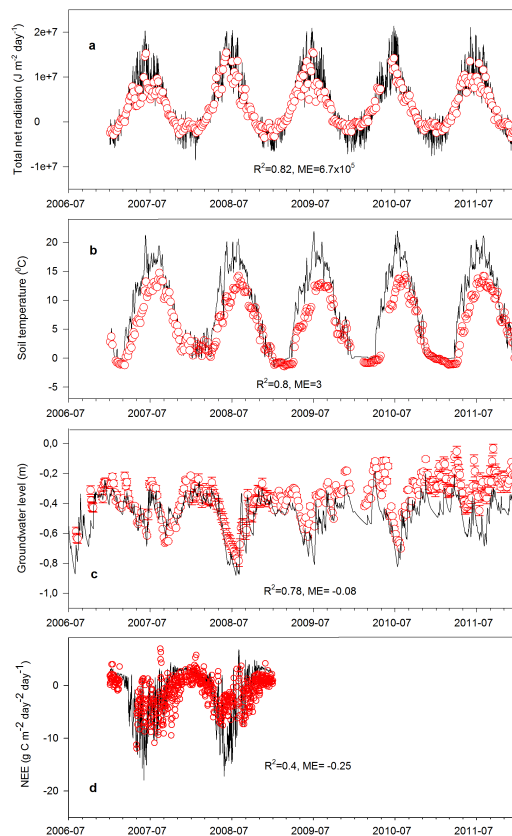
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Figure 4. (a) Simulated (black line) and measured (red hollow circle) total net radiation; **(b)** soil surface temperature (0–5 cm depth); **(c)** GWL; **(d)** NEE. Measured data used to create these plots are 5 day averages, except for NEE where daily averages have been used.
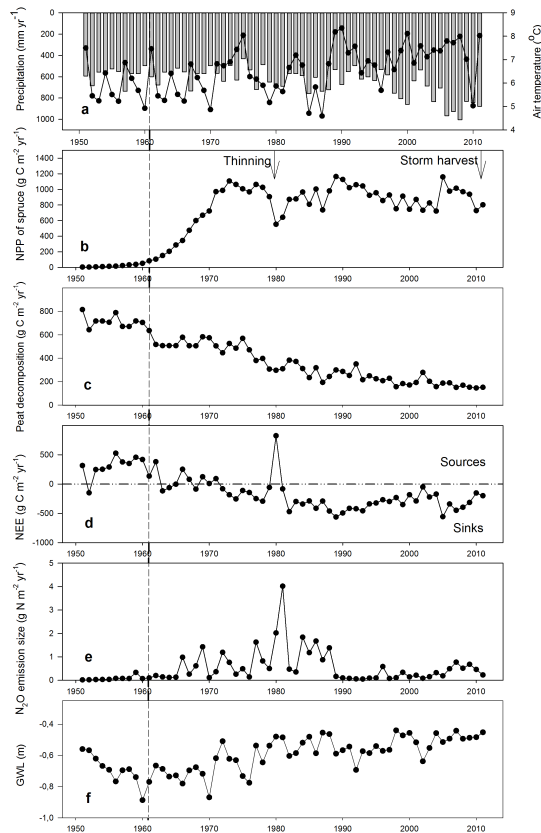
Figure 5. For the period 1951–2011: **(a)** Annual precipitation and air temperature; **(b)** the simulated annual NPP of Spruce trees; **(c)** annual peat decomposition rate; **(d)** annual NEE; **(e)** annual N$_2$O emissions; **(f)** annual GWL. The dashed reference line separates the duplicated 1951–1961 and real climate 1961–2011. The source or sink is based on the atmospheric perspective, e.g. the soil emissions are sources, and plant uptakes are sinks.

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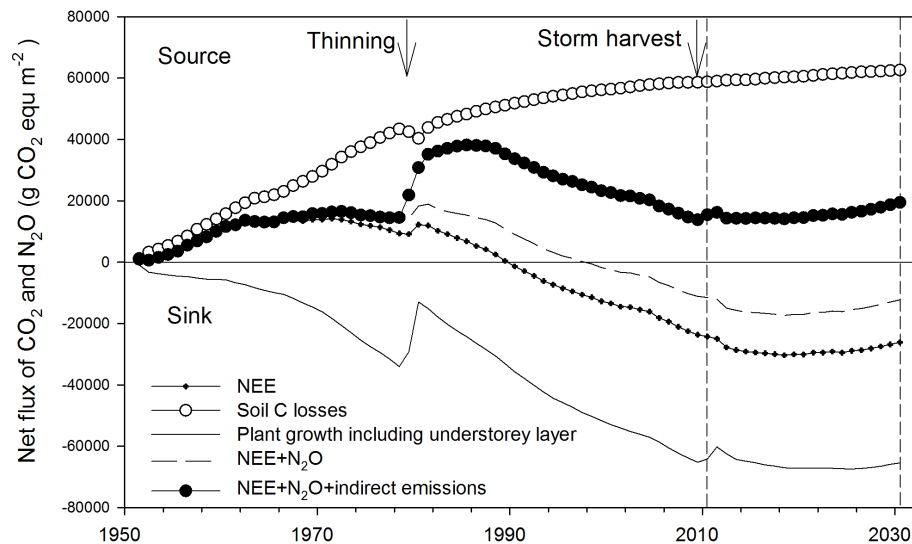
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Figure 6. Simulated total GHG balance for the forest ecosystem from 1951 to 2011 and extended to 2031. The simulated results of 2011–2031 are obtained by running the reference model with extended meteorological files using meteorological data from 1991–2011 duplicated to represent the climate of 2011–2031. It should be noted that the GHG balance presented in this figure assumes no final harvest.