Scandinavian Journal of Forest Research

ISSN: (Print) (Online) Journal homepage: https://www.tandfonline.com/loi/sfor20

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To cite this article: Salar Valinia , Øyvind Kaste & Richard F. Wright (2021) Intensified forestry as a climate mitigation measure alters surface water quality in low intensity managed forests, Scandinavian Journal of Forest Research, 36:1, 15-31, DOI: 10.1080/02827581.2020.1854339

To link to this article: https://doi.org/10.1080/02827581.2020.1854339

Published online: 06 Dec 2020.

SCANDINAVIAN JOURNAL OF FOREST RESEARCH
2021, VOL. 36, NO. 1, 15–31
https://doi.org/10.1080/02827581.2020.1854339

Intensified forestry as a climate mitigation measure alters surface water quality in low intensity managed forests

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ABSTRACT

Climate change has led to a focus on forest management techniques to increase carbon (C) sequestration as a mitigation measure. Fertilisation and increased removal of biomass have been proposed. But these and other forest practices may have undesirable effects on surface water quality. In naturally acid-sensitive areas such as much of Fennoscandia a concern is acidification due to acid deposition in combination with forest practices that increase the removal of base cations and leaching of nitrate (NO_3). Here we apply the biogeochemical model MAGIC to the coniferous-forested catchment at Birkenes, southernmost Norway, to simulate the effects of forest fertilisation and harvest on soil and streamwater. The model was calibrated to the 40-year data for water quality, soil and vegetation and then used to simulate fertilisation and clearcutting of the mature forest by either conventional stem-only harvest (SOH) or whole-tree harvest (WTH). The 5 – 10-year pulse of NO_3 following clearcut was larger with SOH than WTH. WTH causes larger acidification of surface water relative to SOH, due to greater depletion of base cations, N and C from the soil. The use of forestry as a climate mitigation measure should take into consideration the potential effects on soil and surface water quality.

ARTICLE HISTORY
Received 21 February 2020
Accepted 17 November 2020

KEYWORDS
Forestry; acidification; clear cut; water quality; whole tree harvest; forest fertilisation

Introduction

Acidification of soils and surface waters has been a major environmental concern in large parts of Europe and North America. During the 1970s, about 40% of all aquatic ecosystems in Europe exceeded the critical load for acidification by deposition of sulphur (S) and nitrogen (N). For terrestrial ecosystems, the maximum area exceeded was 30% in 1980. Currently, the area with exceeded critical loads for terrestrial and aquatic ecosystems in Europe is 9% and 5%, respectively (de Wit et al. 2015a). Acidification of surface waters is a transboundary air pollution problem. The first signs of adverse effects on fish populations came in Norway as early as 1920 (Dahl 1921). Major reductions of emissions of acidifying air pollutants have been achieved in Europe through the UNECE Convention of Long-range Transboundary Air Pollution (CLRTAP) and its subsequent protocols (UNECE 2014). In Europe, deposition of S has decreased by 70% to 90% since 1980 with an expectation for further decrease towards 2030. Chemical recovery from acidification has occurred in large parts of Europe and North America, yet many acid-sensitive areas in Fennoscandia are still acidified (Stoddard et al. 1999; Skjelkvåle et al. 2005; Futter et al. 2014; Garmo et al. 2014). Although chemical recovery is ongoing, biological recovery is lagging behind (Monteith et al. 2005; Holmgren 2014; Valinia et al. 2014; Garmo et al. 2015). Abatement of acidification from emissions of acidifying compounds is a major environmental and societal success story, although other anthropogenic pressures can re-acidify surface waters and soils.

Climate change is one of the largest societal and environmental challenges of our time. Nature and society are threatened by rising temperatures, changes in precipitation patterns, changing winds, and extreme weather events. Climate change is expected to increase extremes in the hydrological cycle (floods and droughts) and substantially alter biogeochemical cycles and fluxes of greenhouse gas (GHG) emissions from terrestrial and aquatic ecosystems (Parry 2007). Forests play an important role in the global carbon (C) cycle and by sequestering C can contribute to climate mitigation. The northern boreal zone including Fennoscandia has about 22% of the global C pool; most is stored in soil organic matter (Pan et al. 2011). Climate change may increase forest growth and wood production in the boreal region due to higher temperatures and concentrations of atmospheric CO_2 (Lindner et al. 2010). The Intergovernmental Panel on Climate Change (IPCC) identified forests and forestry as crucial in global mitigation (Metz et al. 2007). Reduced deforestation, afforestation of new areas and sustainable forest management can together increase forest C stocks, produce timber, fibre and energy and become a viable climate mitigation tool (Nabuurs et al. 2007). Others argue that afforestation and intensive forestry in the northern boreal region will not contribute to CO_2 mitigation due to the changes in biophysical properties such as albedo (Bala et al. 2007; de Wit et al. 2015b). The European

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Union (EU) has set a target for 2020 of 20% renewable resources in the energy sector and 10% in the transport sector. Here forestry for biomass production is one of the most important substitutes for fossil fuels (EC. 2009). Norway considers intensified forestry as a key strategy for a transition to a "sustainable, low emission society", with its ambitions for "an active, sustainable forestry policy [that] will support the overall climate policy, both nationally and internationally" (KLD 2011).

Forest practices and intensified forest harvesting can affect surface water quality by altering soil C, nutrient and sediment transport (Thiffault et al. 2011; Achat et al. 2015; Clarke et al. 2015), reducing the soil base-cation pool, and thus causing soil and surface water acidification (Binkley and Högberg 2016; Löfgren et al. 2016). Organic pollutants such as methyl mercury can be mobilised (Schelker et al. 2012; Zetterberg et al. 2013; Eklöf et al. 2016; Ledesma et al. 2016). Commercial forestry is known to have a range of negative impacts on aquatic ecosystems and biodiversity (Rundle et al. 1992; Malcolm et al. 2014). The negative impacts are linked to tree species and the manner in which forestry activities such as harvesting are conducted (Kuglerová et al. 2014). Aquatic biodiversity is in turn affected by water quality (Friberg 1997; Thomsen and Friberg 2002).

Models provide tools by which future scenarios of atmospheric deposition, climate change and forestry practices can be evaluated. One such model, MAGIC Model for Acidification of Groundwater In Catchments (Cosby et al. 1985; Cosby et al. 2001), is a process-oriented biogeochemical model developed to simulate long-term changes in stream water and soil water chemistry in response to acid deposition. MAGIC has been applied to a wide variety of sites worldwide and used to evaluate the effects in forest and freshwater ecosystems. For example, MAGIC has been used to assess the effects of forestry practices on Swedish lakes (Moldan et al. 2017). Oulehle et al. (2019) used MAGIC to simulate the acidification and recovery of soil and surface water chemistry following acid deposition, climate stress and insect attack at a lake and its catchment in the Czech Republic.

Here, we conduct a modelling study of the potential effect of several forest practices at Birkenes, a spruce-forested catchment in southernmost Norway. Birkenes is one of the calibrated catchments in the Norwegian national monitoring programme for long-range transported air pollutants (Johannessen 1995). The site is acid-sensitive and has been severely affected by acid deposition. We use the biogeochemical model MAGIC to simulate scenarios of N fertilisation and increased harvest intensity and the effects on water and soil acidification. These scenarios represent potential measures proposed by Norwegian environmental managers to increase CO_2 sequestration and thus mitigate climate change.

Material and methods

Site description

The Birkenes calibrated catchment (0.41 km^2) was selected for this study because of its 44-year data series of precipitation and surface-water chemistry, together with long-term soil and forest monitoring data (Figure 1, Table 1). The site, located 20 km inland from the south coast of Norway, is representative for large areas in Norway and Scandinavia with coniferous forest stands on acid-sensitive soils (Figure 1). Birkenes is heavily exposed to acid deposition and was severely acidified when measurements started in the early 1970s. Substantial recovery has taken place since the S deposition peaked in the late 1980s. There have been no disturbances in the catchment since the onset of monitoring in 1973, except for a small clearcut of 10% of the forest in 1986.

The climate at Birkenes is maritime, with mean annual temperature of 5.9°C, mean annual precipitation of 1490 mm and mean annual runoff of 1127 mm. Snow accumulation during winter is sporadic. The catchment is drained by three small second-order streams, which converge about 150 m upstream a V-notch weir where water flow is measured continuously. Runoff data have been collected at least weekly since 1974 (Garmo and Skancke 2018). The site for precipitation and air sampling is located about 500 m north of the catchment (Aas et al. 2019). Deposition of sea-salts at Birkenes is relatively high due to the proximity to the coast. Present N deposition levels are around 12–14 kg/ha per year. Weekly data on major anions and cations are available since 1974 for both bulk deposition and runoff chemistry.

Soils are characterised by thin podzols and brown earth soils developed on patchy moraine of granitic mineralogy overlying biotite granite. Mineral soils have developed in a shallow layer of glacial till on granitic bedrock. Mineral soil types are acid brown earth and podzols. Peaty deposits have developed on poorly drained sites in the catchment. On the slopes, well-drained thin organic layers on gravel or bedrock are common. Soils have been sampled several years and reported as part of the Norwegian Monitoring Programme for Forest Damage (Timmermann et al. 2019).

The vegetation is mainly Norway spruce (*Picea abies L.*) with some Scots pine (*Pinus sylvestris L.*) and birch (*Betula pubescens L.*) and undergrowth of mosses, blueberry, and fern. The spruce forest is old and slow growing with a mean stand age of 130 years (2015). In 2000, the standing volume was 314 m^3 ha^{-1} with an increment of 3.6 m^3 ha^{-1} yr^{-1} (Kvaalen et al. 2002). In a study by Wu et al. (2010), the above-ground tree litterfall was estimated to 4.4 g m^{-2} $d-^{1}$, with an annual N flux of 1.2 g m^{-2}. The O-horizon depth in the forest stand is 0–12 cm with a C:N ratio of 30, whereas the B-horizon depth is 28–36 cm (Wu et al., 2010).

Model description

MAGIC (Model of Acidification In Catchments) is a process-oriented model of biogeochemical processes operating at the catchment scale (Cosby et al. 1985, 2001) and has been extensively used to simulate the effects of acid deposition, land-use and climate change on freshwater quality. Recent modifications of MAGIC (version 8 dated 1 October 2010)

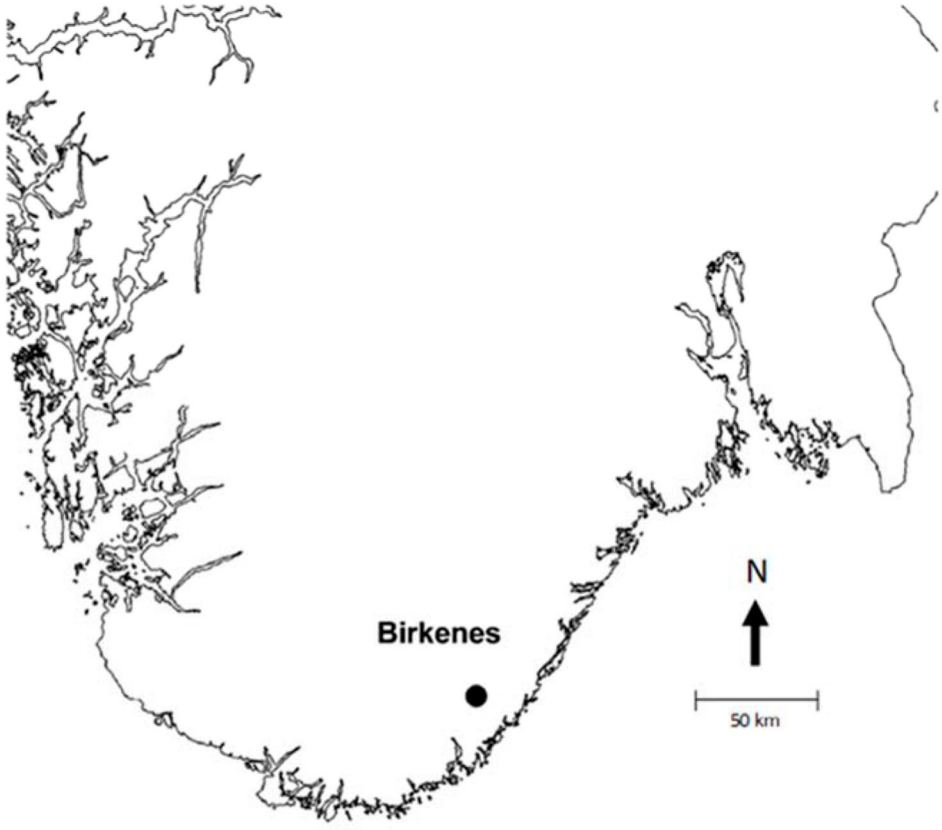

Figure 1. Map showing the location of the study catchment.

include new formulations of N and C processes in forested ecosystems (Oulehle et al. 2012).

The new version of MAGIC attempts to better simulate the short- and long-term changes in the retention and loss of N in catchments. In MAGIC versions 1–5, the fraction of incoming N retained in the catchment was assumed to be constant over time. This assumption was acceptable for sites at which the loss of inorganic N in streamwater was negligible compared to the other strong acid anions, SO_4 and Cl, but it could not be used in applications in which the fraction retained changes over time. In MAGIC version 7 (Cosby et al. 2001), the fraction of N retained was assumed to be a function of the C/N ratio of the soil organic matter (SOM). This was an attempt to model "nitrogen saturation", and link to the empirical evidence that sites with low C/N ratios in SOM leached a larger fraction of incoming N relative to sites with high C/N ratios (Gundersen et al. 1998). Here the problem is that the SOM pool in most soils is very large and thus the C/N ratio changes very slowly over time. Rapid changes in N loss in stream water, such as those following clear cut or other disturbance, cannot be explained alone by changes in the C/N ratio of SOM. The latest version (MAGIC 8) takes the next step in complexity, and describes N retention in soil as the product of microbial processes, which in turn are dependent on the amount and availability of organic matter substrate (litter) (Oulehle et al. 2012). The description does not take into account any changes in soil moisture or temperature due to clear cutting. This version has been used by Oulehle et al. (2019) in an application on the effect of forest dieback by bark beetle attack at Plesne Lake, Czech Republic.

Table 1. Site characteristics.

	Unit	value
Catchment area	km^2	0.41
Latitude	deg N	58.386
Longitude	deg E	8.244
Altitude	m.a.s.l.	200–300
Land cover		
Forest	%	90
Impediment	%	3
Marsh	%	7
Bedrock		Granite, biotite

Our MAGIC application here to Birkenes takes as a starting point the calibration conducted by Larssen (2005) using MAGIC version 7 (dated 14 May 2003). This calibration focussed on the major cations and anions, and the response of the streamwater chemistry to reductions in S deposition during the period 1974–2003. We now include the additional monitoring data for the subsequent period 2004–2017. Our simulations of various forest management scenarios make use of the new formulations of N and C processes in MAGIC 8, largely following the procedures of Oulehle et al. (2019).

The input data requirements for MAGIC comprise catchment-scale fluxes of major ions (Ca, Mg, K, Na, NH_4, SO_4, Cl, NO_3) in atmospheric deposition, soil physical and chemical parameters, and net uptake of nutrients in vegetation, both trees and ground vegetation. For MAGIC 8 specification of soil C pool is also required. We use an annual time step. Net uptake is the annual difference between gross uptake

from the soil and return to the soil via litterfall, both above and below ground. MAGIC simulates stream water and soil chemistry. For the calibration, the simulated annual values are compared with observed soil and stream water data from the monitoring.

Data sources and calibration procedures

The sources of data for the model input and calibration at Birkenes are described by Larssen (2005).

Streamwater

Annual values for the volume and flow-weighted streamwater chemistry (1974–2917) come from the routine monitoring conducted by NIVA as part of the Norwegian national environmental monitoring programmes. Stream flow is monitored continuously by weir and water-level recorder maintained by NVE. Streamwater samples are collected weekly at the weir and analysed for major chemical components at NIVA. Analytical methods have evolved since monitoring began in 1973 and currently entail automated ion-chromatography. For the years prior to the onset of monitoring the annual runoff was assumed to be 1.15 m/yr, the average annual value for the period 1974–2017.

Atmospheric deposition

Annual values for volume and chemical composition of atmospheric deposition (1974–2017) come from the routine monitoring conducted by the Norwegian Institute for Air Research (NILU) as part of the European Monitoring and Evaluation Programme (EMEP) network. The samples are collected daily at the station (NO-1) operated by NILU located about 500 m from the weir at Birkenes. Data and description of sampling and analytical methods are given on www.NILU.no.

The total deposition is comprised of wet and dry deposition. Dry deposition is notoriously difficult to measure accurately to an entire forested catchment such as Birkenes. We thus use several assumptions suggested by Larssen (2005) to estimate dry deposition at Birkenes. First, we assume that the total deposition of chloride (Cl) in deposition equals the observed flux of Cl in streamwater. For the period 1974–2017, the observed flux of Cl in streamwater was 1.48 times the measured wet deposition of Cl. We assume that all Cl deposition comes from seasalt aerosols, and that the deposition of base cations (Ca, Mg, Na, K) and marine sulphate (mSO_4) are in the same proportions relative to Cl as those of seawater. In addition, for Ca some of the deposition was assumed to follow the excess-SO4 deposition. (Ca seasalt factor = 0.037 relative to Cl; Ca excess factor = 0.08 relative to excess SO4*, units meq/m^2/yr) (Table 2).

For SO_4 we also assume that the total deposition equals the observed SO_4 flux in stream water less a possible contribution from weathering of soil or bedrock. SO_4 out = (mSO_4 + SO_4*) + SO_4 weathering, where SO_4* is non-marine (i.e. excess) SO_4 in deposition. For weathering of SO_4 we use the estimate of 20 meq/m^2/yr (Larssen 2005). For the entire period 1974–2017 the SO_4* measured in wet deposition is multiplied by the factor of 1.43 (Table 2).

Table 2. Atmospheric deposition at Birkenes.

Parameter	Unit	1974–1978	1996–2000	2013–2017
Ca	meq/m^2/yr	15.3	10.3	8.4
Mg	meq/m^2/yr	27.2	30.9	36.9
Na	meq/m^2/yr	118.9	134.8	161.1
K	meq/m^2/yr	2.5	2.8	3.4
NH_4	meq/m^2/yr	60.3	53.1	48.3
SO_4	meq/m^2/yr	141.2	72.3	37.6
Cl	meq/m^2/yr	138.9	157.5	188.2
NO_3	meq/m^2/yr	64.7	62.5	50.3

Measured bulk deposition values from NILU adjusted for dry deposition. See text for details.

For N compounds estimates of deposition are more difficult, as N is actively retained in the catchment. Throughfall measurements do not help as N flux in throughfall may be lower than in open field wet deposition due to retention of N in the forest canopy. Thus here we assumed that dry deposition of N is 14% of wet for both NH_4 and NO_3, for all years. This is the average value reported by NILU in the annual reports, for the years 1987–2017 (Table 2).

For the years prior to the onset of monitoring deposition of seasalts and base cations was assumed to be the same as the average annual values for the period 1974–2017. For SO_4*, NH_4 and NO_3 historical deposition (1860–1974) was taken as the modelled deposition for the Birkenes grid square in the EMEP estimates based on historical emissions of air pollutants in Europe (Schöpp et al. 2003) scaled to the estimated deposition amounts for the calibration period 1996–2000. For the forecast period 2018–2100, the deposition of seasalts and base cations was again assumed to be the same as the average annual values for the period 1974–2017. Future deposition of the pollutant components was assumed to follow the European emissions of the CLRTAP current legislation scenario (CLE) for the Birkenes grid square forward to the year 2030 and then held constant to the year 2100. These data were supplied by the Coordination Centre for Effects of the CLRTAP.

MAGIC parameters

Soil data were taken from sampling conducted by the Norwegian Forest Research Institute (NISK) (now Norwegian Institute of Bioeconomy Research NIBIO) also as part of the Norwegian national environmental monitoring programme. These are summarised by Larssen (2005). The net uptake of base cations and N by the forest for the period 1860–1990 was based on the assumption that the long-term forest practice was selective cutting of mature trees, with removal of the boles of the trees only. The rest of the tree biomass was left in the catchment and re-cycled through the soil. The annual net uptake was thus assumed to be equal to the amounts in the boles (stems) of the trees in 1990 (Røsberg and Stuanes 1992) assuming the stand was 95 years old and that the accumulation is constant over the entire life of the tree (Table 3).

The calibration for the base cations entails a trial and error procedure in which for each of the base cations the weathering rate and initial pool (year 1860) of

Table 3. Values of input parameters to MAGIC for Birkenes.

Parameter	Unit	Value
Stream fixed parameters		
Area	% of Catchment	5
KAl	log 10	8.6
H power		3
pK1		3.04
pK2		4.51
pK3		6.46
Temperature	°C	5
pCO_2	atm	0.070
DOC	mmol charge/m^3	15
Nitrification	% of inputs	100
De-nitrification	meq/m^2/yr	0
Soil fixed parameters		
Depth	m	0.4
Porosity	fraction	0.5
Bulk density	kg/m^3	656
CEC	meq/kg	113
SO_4 half saturation	meq/m^3	100
SO_4 maximum capacity	meq/kg	0.1
KAl	log 10	7.8
H power		3
pK1		3.04
pK2		4.51
pK3		6.46
Temperature	°C	5
Moisture	%	100
pCO_2	atm	0.330
DOC	Mmol charge/m^3	30
C and N parameters		
Biomass C/N	mol/mol	10
Carbon efficiency	%	24
Nitrogen efficiency	%	60
Decomp efficiency	%	2.25
Plant uptake NO_3	mmol/m^2/yr	496
Organic output C/N	mol/mol	44
orgC decomp	mol/m^2/yr	24.0
orgC litter	mol/m^2/yr	25.9
Litter C/N	mol/mol	53.3
orgN litter	mmol/m^2/yr	485
Nitrification	% of inputs	100
De-nitrification	meq/m^2/yr	0
Tree uptake (long term)		
Ca	meq/m^2/yr	4.2
Mg	meq/m^2/yr	1.1
Na	meq/m^2/yr	0
K	meq/m^2/yr	2.4

Stream and soil fixed parameters are from Larssen (2005). C and N parameters are estimated here for the microbial N version of MAGIC. See text for details.

Table 4. Values for calibrated parameters of base cations for Birkenes.

Parameter	Unit	Value
Soil base cations		
Weathering Ca	meq/m^2/yr	28
Weathering Mg	meq/m^2/yr	0
Weathering Na	meq/m^2/yr	20
Weathering K	meq/m^2/yr	2
Initial %Ca		18.5
Initial %Mg		13.0
Initial %Na		2.5
Initial %K		3.2
Selectivity coefficient AlCa	log10	−0.93
Selectivity coefficient AlMg	log10	−0.59
Selectivity coefficient AlNa	log10	−1.09
Selectivity coefficient AlK	log10	−6.81

exchangeable cation are specified, then the model is run for the hindcast period (1860–2000) and the simulated values for the concentration of base cation in stream water and the % cation in the soil are compared with the observed. This is repeated until the simulated values agree with the observed (Table 4).

Carbon and nitrogen pools and fluxes in vegetation and soil: hindcast period 1860–2017

In MAGIC8, there is no explicit model for build-up of C in the forest, i.e. there is no forest growth model. This would for example simulate forest growth and the resulting accumulation of C depending on factors such as light, temperature, and nutrient availability. For MAGIC8 it is thus necessary to externally specify the C pools and fluxes and changes in these in order to estimate the supply and loss of other elements – namely, Ca, Mg, K, and N – to the soil. A key parameter is the annual litterfall, both above- and below-ground, as this determines the size of the internal cycling of C (and of nutrients). A second key parameter is the annual increment – i.e. the change in the size of the C pool in tree biomass.

Carbon. Røsberg and Stuanes (1992) give estimates of the total C, N and nutrient cations in biomass and soil at Birkenes based on data from the Norwegian monitoring programme. These estimates are for the year 1990 under 95-year-old spruce forest. The stand density was 1333 trees/ha (Table 5).

The carbon flux in litterfall at Birkenes has been estimated by Fröberg et al. (2011) based on data from the Norwegian monitoring programme. They measured only the aboveground litterfall. They estimated the below ground litterfall using a model of mean-residence time of soil organic carbon (humus layer) based on ^{14}C measurements of archived soil. The litterfall flux of N was calculated from the C flux and measured C/N ratio in needles for the aboveground, and estimates of C/N ratio in roots from Gordon and Jackson (2000) (Table 6).

Nitrogen. Once the litterfall C and C/N were set, the annual amount of N in litter was calculated. For the steady-state forest in 1990, the N in litterfall was assumed to be compensated by an equivalent uptake of N by the trees including the uptake of N incorporated into the annual increment in the boles of the trees. The annual increment was estimated from the total amount of N in the boles of the trees (496 mmol/m^2) divided by the age of the forest (100 years) = 5 mmol/m^2/yr. Thus, the total uptake for all years prior to clearcut was 0.50 mol/m^2/yr. We assumed that both N fixation and de-nitrification were negligible, and that all NH_4 remaining in soil solution was nitrified to NO_3.

Soil compartment. The C pool in the forest floor horizons of the soil was assumed to be higher in 1990 relative to 1895 due to gradual build-up of organic matter in the forest floor and O layer due to ecosystem retention of N deposition. We estimated that 50% of N deposition 1895–1990 went to make new SOM, and the remaining 50% went to lower the C/N ratio of existing SOM. This is consistent with the observations from the experimental additions of N at the long-term whole ecosystem experiment at Gårdsjön NITREX in Sweden (Moldan et al. 2018). Using the measured data from Røsberg and Stuanes (1992) the C/N of total soil including the forest floor in 1990 was 25 mol/mol, but using the measured data from Amlid et al. (1992) C/N of the total soil

Table 5. C, N and nutrient cations in biomass and soil at Birkenes based on data from the Norwegian monitoring programme.

Component	Organic matter	C	Ca	Mg	K	N	C/N
Units	ton/ha	mol/m^2	meq/m^2	meq/m^2	meq/m^2	mol/m^2	mol/mol
Trees							
Foliage	15.3	65	406	107	180	1.1	58
Branch	12.5	54	217	37	100	0.6	91
Bole	47	246	399	103	228	0.5	496
Stump	4.6	2566	57	8	18	0.1	358
Roots	29.5	125	140	45	69	0.4	318
Total	108.8	516	1219	300	595	2.7	193
Understory	2.7	117	57	16	43	0.3	39
Forest floor O horizons	59	252	885	420	271	5.2	49
Soil extractable			913	210	448		
Soil total	116	479				26.7	18

Data from Røsberg and Stuanes (1992).

except the C horizon (data not given) in 1990 was 30 mol/mol. We used the C/N ratio of 30 mol/mol for the year 1990, as this is more realistic for sites northern coniferous forest sites that are nitrogen limited (Gundersen et al. 1998).

The long-term historical rate of decomposition of SOM was thus assumed to be slightly lower than the long-term historical rate of litterfall. This gave a gradual build-up of soil C (Tables 7 and 8).

Finally, the loss of organic C as dissolved organic C in stream water was specified in MAGIC by setting the decomposition efficiency to 2.25%. This gave an annual organic C output (as DOC in runoff) of about 560 mmol/m^2/yr (= 5.4 mgC/l in 1.25 m/yr runoff, average observed values for 1996–2000). With an assumed TOC/orgN ratio of 44 mol/mol, orgN in runoff was thus 13 mmol/m^2/yr. The parameter values used for the microbial N version are summarised in Table 9.

Given these parameter values the starting values for the simulation (initial year 1860) were calibrated (Table 10). For the period 1991–2017, the forest was assumed to continue to grow as before, with the continued accumulation of C and N in the forest floor.

Specification of C, N and base cations following scenarios of forest fertilisation and harvest

In MAGIC8, all the carbon fluxes must be specified as input files, as MAGIC does not have a forest growth component. The inputs (from litter) and outputs (by uptake) to and from the soil are specified for each year in the "Source" and "Sink" files for the base cations, and the "Carbon–Nitrogen" file for C and N.

Table 6. Estimated litterfall at Birkenes.

	C mol/m^2/yr	Method	N mol/m^2/yr	C/N mol/mol
Above ground	11.7	Measured	0.20	58[a]
Below ground	14.2	By difference	0.28	50[b]
Above + below	25.9	Modelled	0.50	52 calc.

[a]C/N value for needles
[b]C/N value for fine roots, from Gordon and Jackson (2000)
Data from Fröberg et al. (2011).

Table 7. Measured (1995) and calculated pools of C and N in soil at Birkenes.

	Org C mol/m^2	C/N mol/mol	Org N mol/m^2
1990	730	30	24.3
1895	605	29.1	20.8

Measured data from Røsberg and Stuanes (1992) and Amlid et al. (1992).

Table 8. Values for the long-term historical rate of litterfall and decomposition of soil organic matter (SOM) at Birkenes.

	Org C mol/m^2/yr	C/N mol/mol	Org N mol/m^2/yr
Litter	25.9	55	0.47
Decomposition	24.0	32	0.76

Table 9. C and N parameter values used to calibrate the microbial N version of MAGIC to Birkenes (calibrated parameter file BIE-M18.par).

	Unit	Value
Biomass C/N	mol/mol	10
Carbon efficiency	%	24
Nitrogen efficiency	%	60
Decomp efficiency	%	2.25
Initial C pool (year 1860)	mol/m^2	548
Initial N pool (year 1860)	mol/m^2	18.8
Soil C/N	mol/mol	29
Plant uptake NO_3	mol/m^2/yr	0.50
Organic output C/N	mol/mol	44
orgC decomp	mol/m^2/yr	24.0
orgC litter	mol/m^2/yr	25.9
ΔC litter-decomp	mol/m^2/yr	1.9
Litter C/N	mol/mol	53.3
orgN litter	mol/m^2/yr	0.49

At the year 2030 after the clearcut (cut assumed December 2029), there was a large one-time input of litter from the logging residues, in addition to the continued decay of normal litter from the previous years (here termed "legacy litter"). The contribution to the decomposition of SOM from previous years' normal litterfall declined from the initial steady-state value of 24.0 molC/m^2/yr over the years after clearcut. The ground vegetation expanded after the clearcut and contributes to the litterfall. And there was a small amount of litter from the new growing forest. We assumed that the total decomposition soil orgC was the sum of these four sources. After 30 years (canopy closure, year 2058) the annual litterfall again approached the steady-state value of 25.9 molC/m^2/yr, and decomposition orgC 24.0 mol/m^2/yr (Figure 2).

The litterfall from the new growing forest was set proportional to the re-growth of the forest. We used the forest growth functions from Plesne Lake (Oulehle et al., 2019), albeit scaled to 0.6, as forest growth in Norway is lower than in the Czech Republic. We assumed canopy closure after 30 years, at which time the litterfall is assumed to

Table 10. Calibrated initial soil C and N pools (calibrated parameter file BIE-M18.par).

	Unit	Value
Initial C pool (year 1860)	mol/m^2	548
Initial N pool (year 1860)	mol/m^2	18.8
Initial soil C/N (year 1860)	mol/mol	29

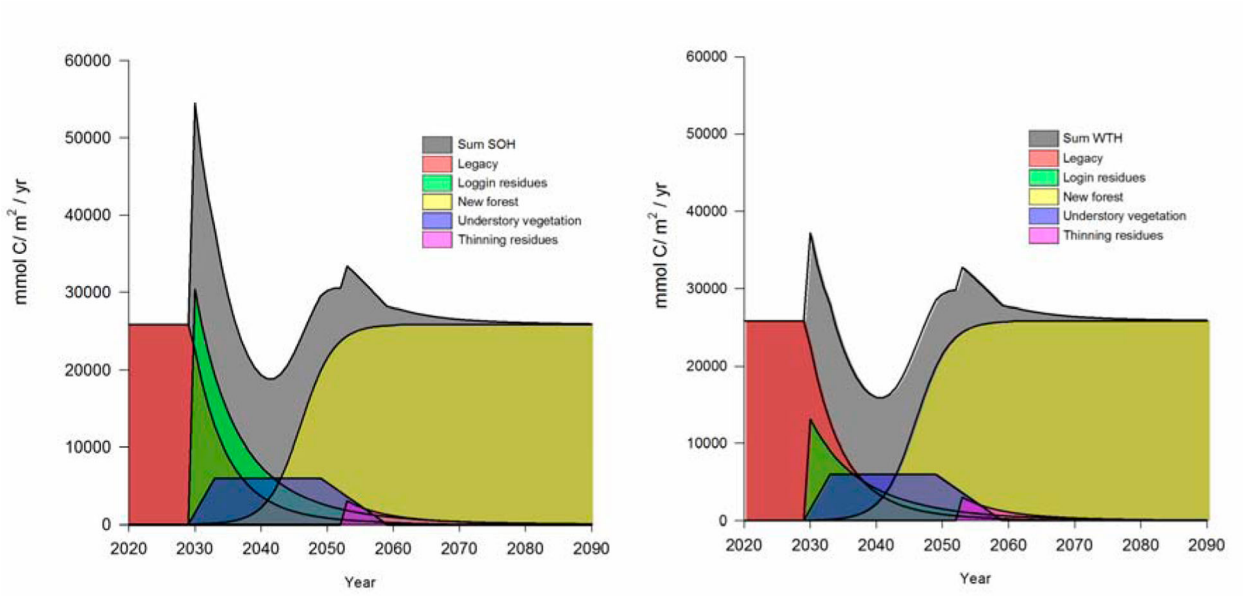

Figure 2. Stipulated inputs of organic C to the soil from tree litter for the stem-only-harvest (SOH) scenario (left-hand panel) and the WTH scenario (right-hand panel). Clearcut in year 2029, re-growth of the forest to canopy closure in 2050, and thinning of 30% of the trees in 2053.

reach "steady-state", i.e. the values measured by Fröberg et al. (2011) in the 1990s for the 95-year old forest.

For the logging residues after clearcut we relied on Hyvonen et al. (2000) who give decay curves for various litter fractions. We used data for the site Slogberget as it is climatically closest to Birkenes and also has spruce forest. We used the curves for needles and small branches (10 mm) for both C and N. We assumed that fine roots decayed as needles and coarse roots decayed as branches. Further, we assumed that the base cations were released proportional to C. Hyvonen et al. (2000) showed that C is released faster than N in the decay of logging residues. The litter and decay of thinning residues was assumed to be the same as for the clearcut residues, *albeit* with smaller initial amounts of residues as only 30% of the trees were assumed felled.

For the fertiliser scenarios, the amount of N in the fertiliser was added to the carbon–nitrogen file whereas the base cations (BC) were added to the source file for the year 2018.

Uptake from the soil

Uptake by vegetation removes BC and N from the soil. BC and N were treated differently. For BC only the net uptake (the annual increment added to the standing biomass) was specified. For N the gross uptake and gross litterfall were specified. We used again the forest re-growth model from Plesne Lake (Oulehle et al. 2019) scaled by the factor 0.6.

Table 11. Scenarios of future forest fertilisation, thinning and harvesting.

Scenario	Fertiliser in 2018	Harvest in 2029
SOH	no	Stem-only harvest
SOH-fert	yes	Stem-only harvest
WTH	no	Whole-tree harvest
WTH-fert	yes	Whole-tree harvest

Scenario description

We evaluated four scenarios (Table 11). The scenarios are all based on the same hindcast simulation for the period 1860–2017. The hindcast assumes a steady-state forest with the selective cutting of 1% trees per year and age in 1990 of 95 years.

In all the scenarios, we assumed that replanting occurred in year 2030, the new forest was thinned in year 2053, and that canopy closure occurred after 30 years, i.e. year 2060. The fertiliser and amount were assumed to be the type widely used in Norway, YaraBela® OPTI-KAS™ SKOG coarse granulated (8 mm). The amount was assumed to be 150 kgN/ha applied in a single dose. This amount is equivalent to 535 mmol/m^2 NO_3-N, 535 mmol/ m^2 NH_4-N. The fertiliser also contains 107 meq/m^2 Mg and 71 meq/m^2 K. We assume that all the fertiliser is dissolved and enters the soil in the year of application.

Results

Calibration hindcast period 1860–2017

The re-calibration of the hindcast using the additional years of observations resulted in only minor changes relative to the calibration of Larssen (2005). For the entire record 1974–2017, the average simulated concentrations of major ions in stream water differed from the observed by < 1 µeq/l (Table 12). The long-term stream water data indicate that the assumptions for Cl and SO_4 deposition are valid. The long-term trends in SO_4 concentrations in the stream water were well-explained by the model. The concentrations of NO_3 in streamwater were very low in the hindcast both observed and simulated (Figure 3(A)). The long-term trends in concentrations of BC in streamwater were also simulated satisfactorily. Consequently, the simulated ANC concentrations also agreed with the observed (Figure 3(B)). The re-

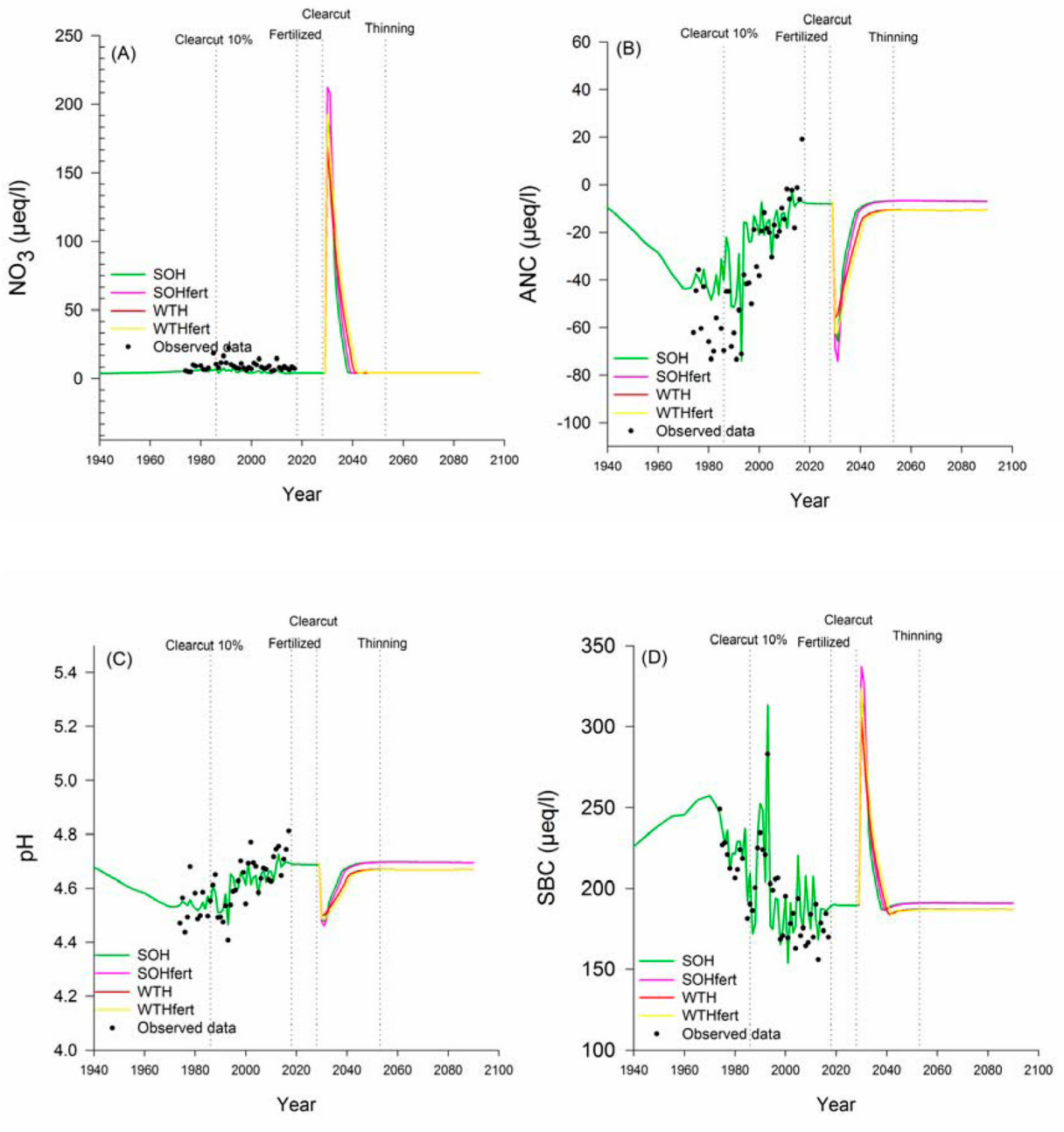

Figure 3. Surface water parameters simulated and observed for the whole-time period (1860–2100) given four scenarios of forest fertilisation and harvest. Vertical dashed lines mark four major anthropogenic disturbances; 1986 10% clear cut; 2018 fertilisation; 2029 clear cut; 2053 thinning. SBC is the equivalent sum of Ca, Na, K, and Mg.

calibration procedure forced the simulated exchangeable % base cation amounts C and N pools in the soil to agree with the observed (Figure 4(C,D)).

The hindcast scenario from 1860 to 2017 entailed only selective cutting of a few mature trees each year and no removal of biomass since the monitoring started in 1973. The exception is a small clearcut of 10% of the catchment in 1986. The observed changes in surface water chemistry since 1973 are due to atmospheric deposition of S and N without any significant intervention from forestry practice.

Table 12. Observed and simulated mean concentrations of major ions in streamwater at Birkenes

	pH	Ca	Mg	Na	K	SO_4	Cl	NO_3	ANC
Observed 1974–2017	4.61	41.8	26.4	124.8	3.7	94.0	130.0	9.1	−36.2
Simulated-observed 1974–2017	0.00	0.5	1.2	1.2	−1.1	−0.8	0.5	−0.0	0.5
Observed 1996–2000	4.62	37.9	24.0	124.9	2.4	84.4	133.5	7.9	−36.6
Simulated-observed 1996–2000	0.00	0.2	0.4	−4.4	0.1	−6.4	−11.2	0.5	17.0

Units: µeq/l.

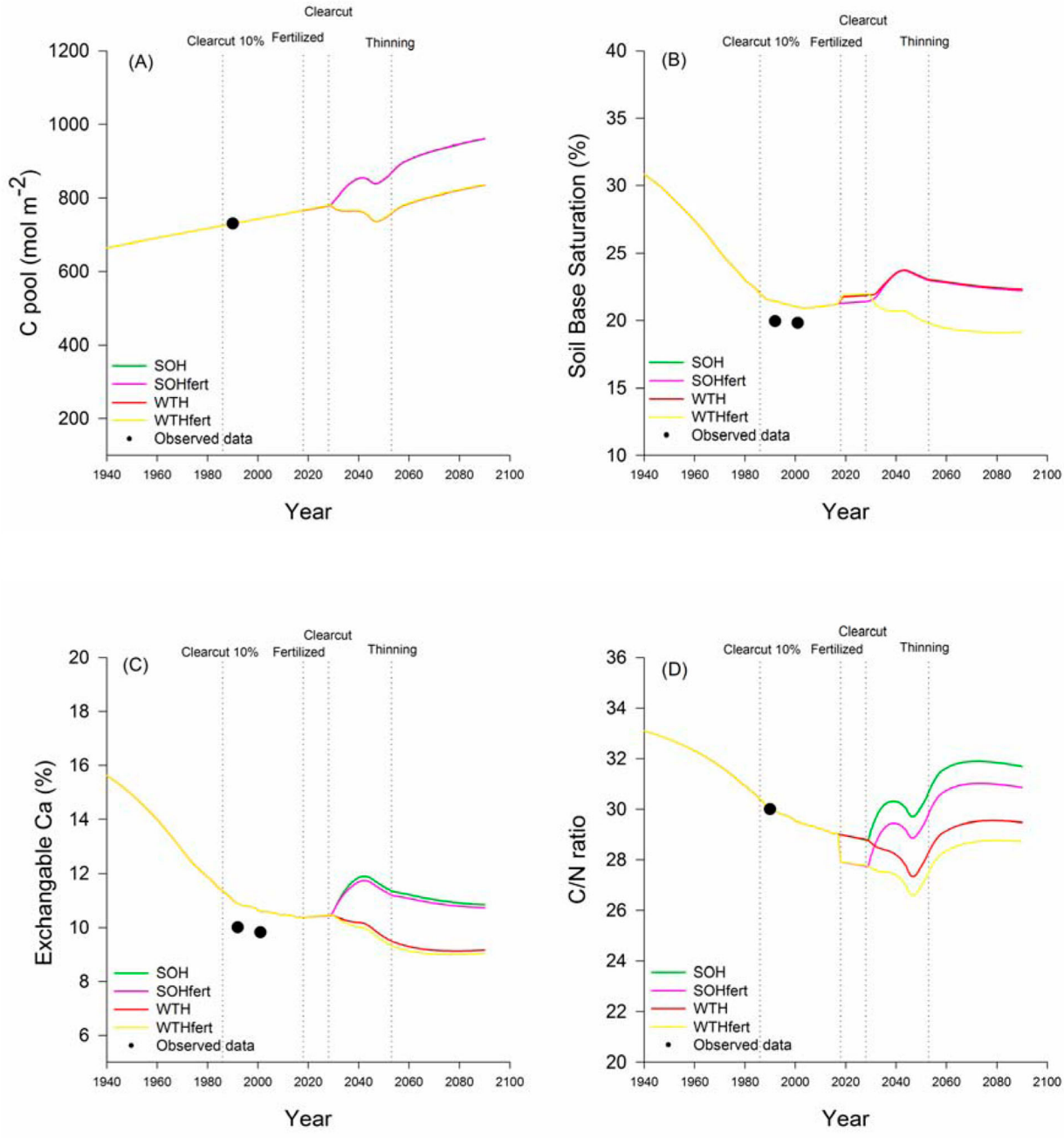

Figure 4. Soil parameters simulated and observed for the whole-time period (1860–2100) given four scenarios of forest fertilisation and harvest. Vertical dashed lines mark the three major anthropogenic disturbances; 2018 forest fertilisation; 2029 forest clear cut; 2053 forest thinning.

The hindcast modelling presents clear evidence of anthropogenic acidification of both surface waters and soils. Birkenes is an acid-sensitive catchment, and the simulations indicate a decrease ANC from 20 µeq/l in the 1800s to −70 µeq/l in 1990 and a drop in pH from about 5 to 4.5 (Figure 3(B,C)). In response to the reductions of S and N emissions mainly driven by the CLRTAP agreements, a substantial recovery has occurred in the streamwater. ANC levels are now around 0 µeq/l and pH levels are 4.8. Although there are clear signs of recovery from acidification, the area is still very sensitive to re-acidification as the base cations in the soils are depleted and the natural protection against acidity is diminished (Figure 4(B,C)).

For the hindcast period 1860–2018, the simulated soil parameters follow the same trends as in surface waters with changes in acid sensitive parameters (Figure 4(A–C)). The soil simulation is within acceptable deviation from measured parameters. During the historical period from 1860 to 1970, there was only selective cutting of mature trees in the catchment, and we assume that the forest soil slowly accumulated C during this time (Figure 4(A)), similar to that in unmanaged forest. Soil base saturation (%) and exchangeable Ca (%) decreased as an effect of atmospheric acid deposition (Figure 4(B,C)). The simulated C/N ratio first increased due to the increasing C pool in the soil, but then began to decrease due to the

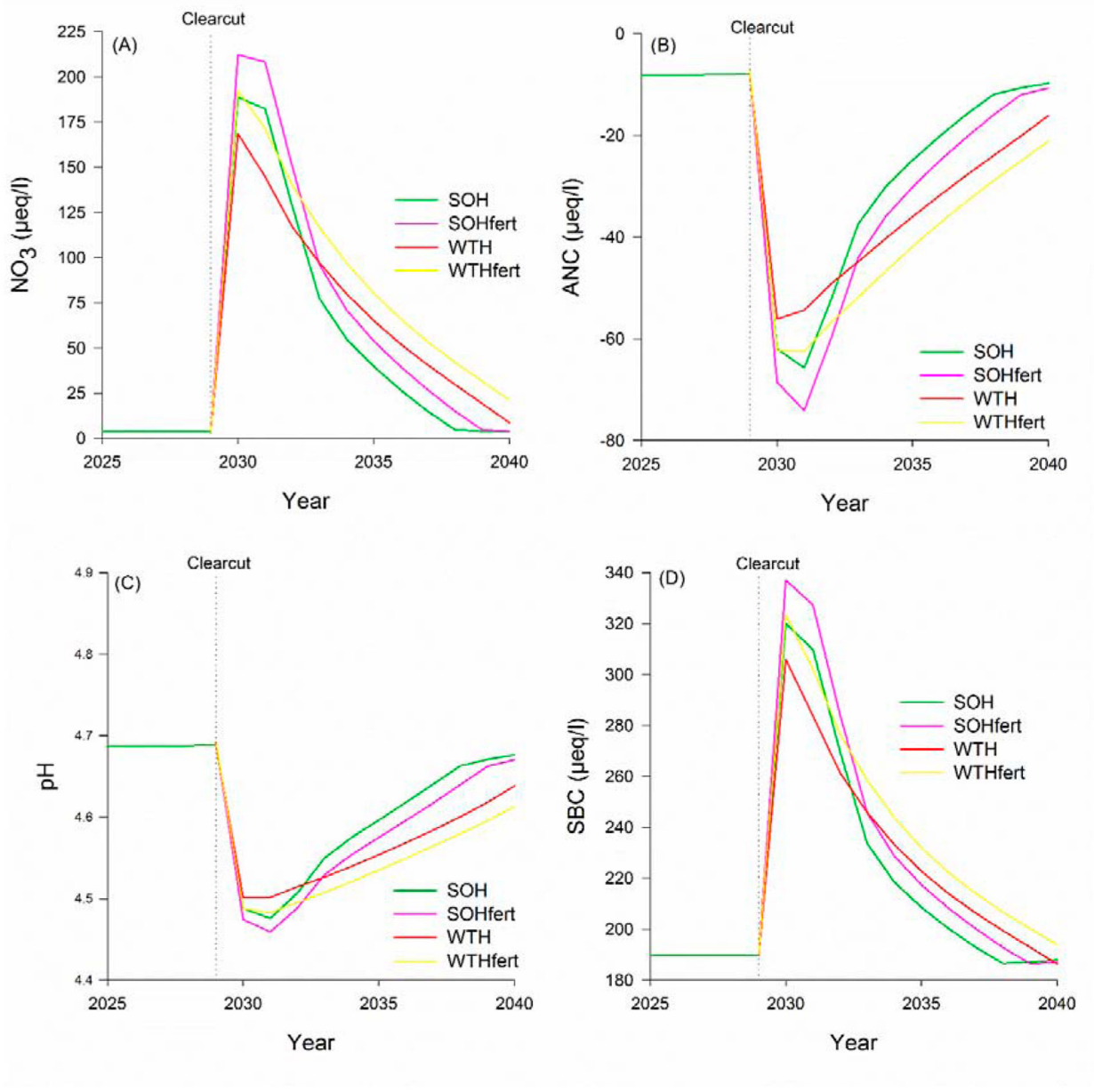

Figure 5. Surface water parameters simulated for the period immediately before and after the clearcut (2025–2040) given four scenarios of forest fertilisation and harvest. Vertical dashed line mark the clear cut in 2029. SBC is the equivalent sum of Ca, Na, K, and Mg.

immobilisation of incoming atmospheric N deposition (Figure 4(D)).

Forecast (scenarios)

Four scenarios were run until 2100 to identify the potential long-term effects of different forest management scenarios on surface water and soil quality. There are evident differences between the scenarios dependent on whether fertiliser was added and the type of clearcut. The modelled leaching of NO_3 to the stream after clearcut was substantial for all four scenarios; the largest leaching occurred with the SOHfert scenario, with a peak of 224 µeq/l in the year 2030. The other scenarios had a slightly smaller peaks of NO_3; the SOH scenario had a peak of 188 µeq/l, WTH of 168 µeq/l and WTHfert of 192 µeq/l respectively. The WTHfert and SOH scenario had very similar peaks of NO_3 leaching to the stream (Figure 5(A)). The large peak of NO_3 was mainly responsible for the re-acidification with sharp drops in ANC and pH in streamwater. The effects in ANC and pH followed the same trend as for NO_3 with the largest effect in the SOHfert scenario and smallest in the WTH scenario (Figure 5 (B,C)). Concentrations of BC followed NO_3 (Figure 5(D)). The large NO_3 peak results in a re-acidification of the streamwater back to levels of maximum acidification in the late 1980s and reverses the substantial recovery that had occurred in the past 30 years due to the reduction of acid deposition. New recovery of streamwater to levels came about 15 years after the clearcut in the SOH scenarios, while much longer recovery times were required in the WTH scenarios.

The simulated forecast soil parameters showed large differences between the SOH and WTH scenarios, but only

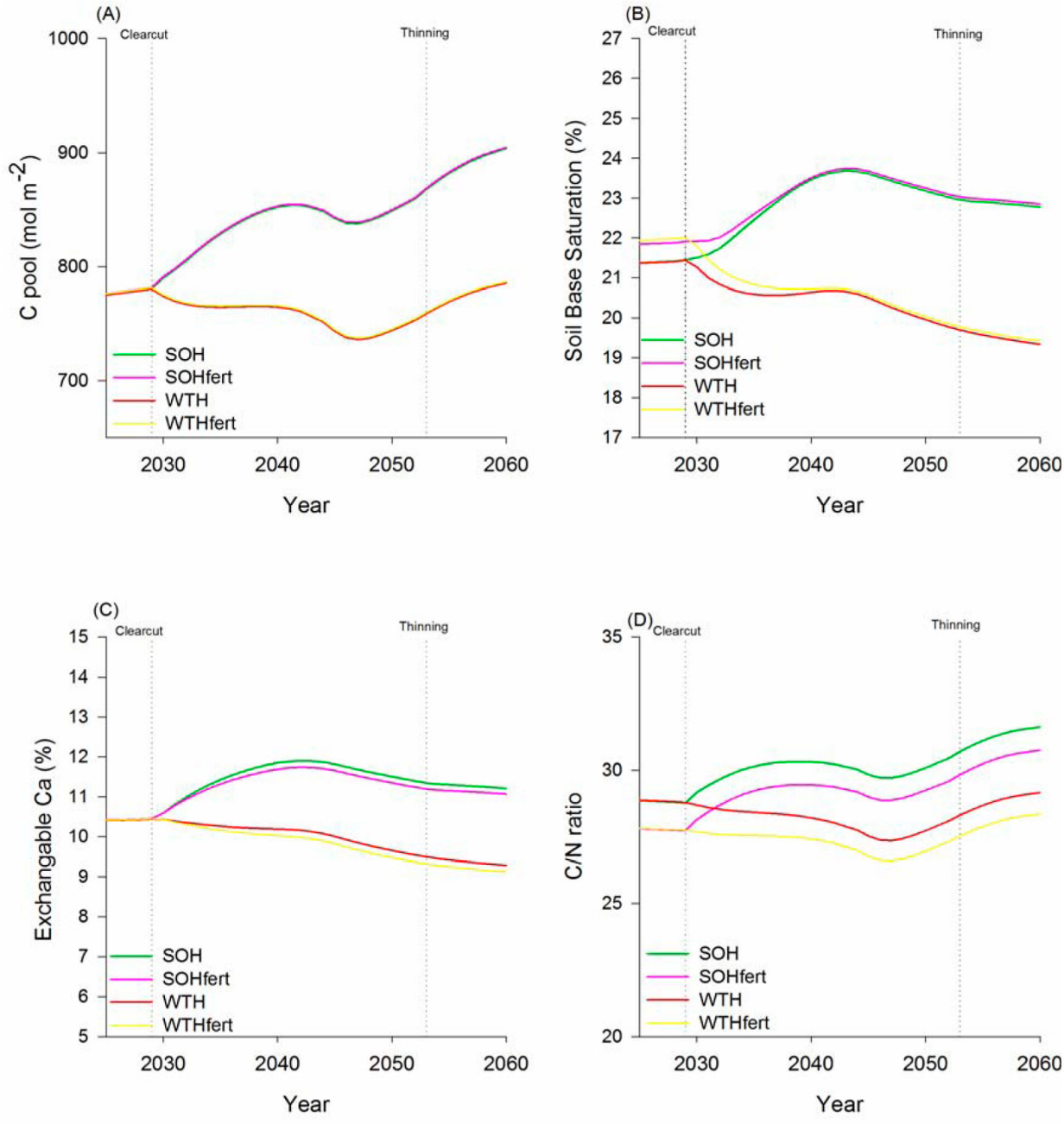

Figure 6. Soil parameters simulated for the period immediately before and after the clearcut (2025–2040) given four scenarios of forest fertilisation and harvest. Vertical dashed line marks the 2029 clear cut.

minor differences between fertilisation or no fertilisation scenarios (Figure 6). The SOH scenarios indicated an increase in the C pool after clear cut, while the WTH scenarios had a decrease in the C pool (Figure 6(A)). The difference between SOH and WTH is due to the removal of needles and branches in WTH, which removes the decomposable material from the system and hence depletes the soil C pool in the long term. The simulations suggest that the C pool does not fully recover during the next 80 years. The removal of needles and branches also deprives the soil of the BC that would otherwise be returned to the soil from the needles and branches. This implies a decrease of base saturation and exchangeable Ca in the WTH scenarios, whereas the soil base saturation and exchangeable Ca increase in the SOH scenarios (Figure 6(B,C)). The simulations indicate a long lag time of the effects to the soil system. The C/N ratio decreases with the WTH scenarios, because the needles and branches that are rich in N are removed from the system. The addition of N fertiliser lowers the C/N ratio prior to the clear cut, but in the SOH scenarios the simulated C/N ratio recovers quickly.

Discussion

Many countries have identified intensified forestry as an effective way to store more carbon and thereby reduce

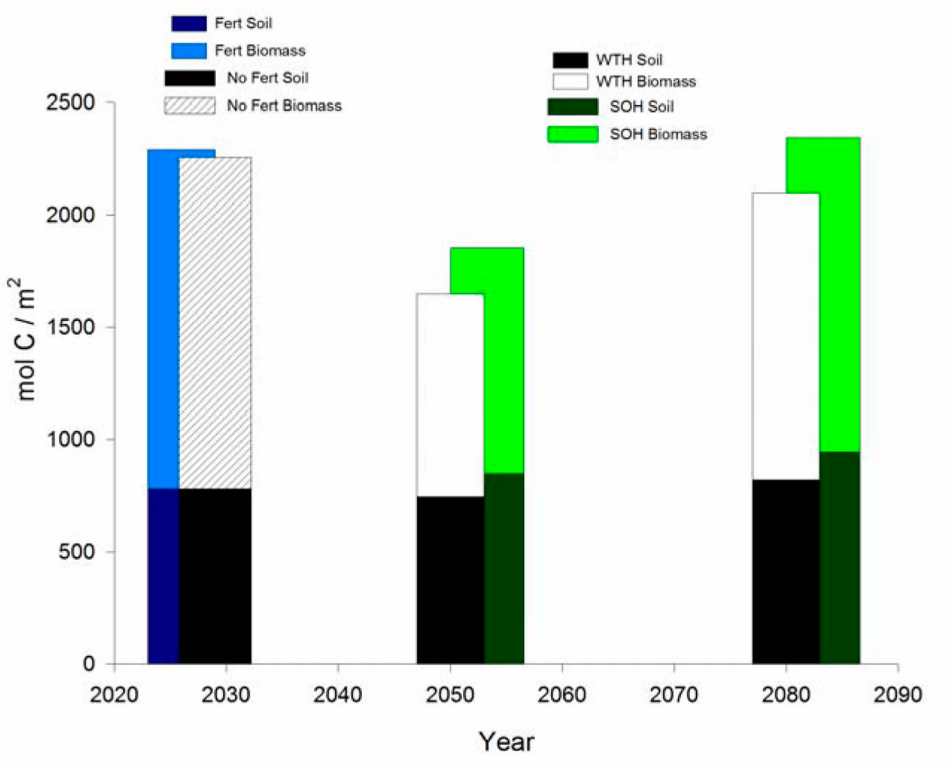

Figure 7. Estimated C pools in biomass and soil at Birkenes over one rotation period of the forest. Shown are estimates for scenarios the difference in C-pool in biomass and soil between stem-only harvest (SOH) and whole-tree harvest (WTH) in 2030, 2050 and 2080. Addition of fertiliser in year 2019 gives only a minor increase in biomass C pool in year 2029 (not shown). The biomass in year 2029 was 374 molC/m^2 stems and 165 molC/m^2 branches and needles.

their net CO_2 emissions. A more intensive forestry might include afforestation of new areas, increasing tree density in existing forests, fertilisation prior to harvest, and whole-tree-harvest to produce biofuels that can replace fossil fuels. However, a strong investment in intensified forestry as a climate mitigation measure can also have negative environmental effects, e.g. on surface water quality in sensitive areas.

The link between clearcutting and N leaching is well established (Likens et al. 1970; Kreutzweiser et al. 2008; Schelker et al. 2016). Clearcutting reduces the vegetation demand for N and can promote increased mineralisation of organic N stored in forest soils and decomposing residues. This decrease in demand and increase in supply is typically manifested in increased soil solution and surface water NO_3 concentrations which can be observed for as much as ten years following clearcutting (Futter et al. 2014). The magnitude and duration of N leaching are generally higher in more productive stands. Thus, it can be expected that fertilisation will lead to higher rates of N leaching following clearcut.

There are relatively few long-term studies on the effects of forestry on surface waters, and challenges with finding good replicates in the field makes it difficult to carry out comparative studies on how different forest management strategies (whole-tree harvest vs. stem-only harvest, fertilised vs. unfertilised forests) affect surface water quality on a catchment scale and over a longer time span. To overcome some of these difficulties we used the biogeochemical catchment model MAGIC to simulate possible long-term impacts of different forest management scenarios at Birkenes, an intensively studied and unmanaged forested catchment in southern Norway.

Forest fertilisation

Forest fertilisation has several direct and indirect effects on forest growth, water pollution and greenhouse gas dynamics. In the Norwegian strategy for intensified forestry as a climate mitigation measure, the recommended forest fertilisation application is 150 kgN/ha about 10 years prior to clear cut (Ellen et al. 2014). The main goal of fertilisation is to increase tree growth. There is some evidence of lower N losses when multiple, smaller fertiliser applications are made over multiple years (e.g. 50 kg N/ha every second year for six years) compared to a single application of 150 kg N/ha (Ring et al. 2011). In northern coniferous forests, forest growth is mainly limited by N availability, and studies have shown that fertilisation in mature forest provides an economic benefit as the growth of the stems (boles) increases substantially, but this may also entail the enrichment of needles and branches and cause a greater leaching of N after clear cut (Saarsalmi and Mälkönen 2001; Hedwall et al. 2014). The MAGIC simulations here assumed that forest fertilisation leads to increased uptake and stem growth during the 10-year period after fertilisation. The main effect of the forest fertilisation on streamwater comes after clearcutting the forest (Figure 5). The fertilisation scenarios had higher leaching of N to the stream after clearcut, 12% higher in the SOH

scenarios and 16% higher in the WTH scenarios. The higher N leaching is due to the enrichment of N in the soil by the fertiliser. The increased leaching of N caused a larger decrease in ANC and longer recovery after the clearcut.

The simulations suggest that the actual fertilisation event did not cause undue leaching of NO_3 on an annual basis. Although transient losses of NO_3 and NH_4 may occur immediately after fertilisation, the effects were too short or too small to be apparent in the annual mean concentrations. The model simulations are in line with experimental studies from Finland, which show that a fertilisation event such as the one simulated at Birkenes does not result in a large pulse of NO_3 to streamwater as long as the forest is N limited (Saarsalmi and Mälkönen 2001). Soil solution and streamwater NO_3 levels are typically very low in Nordic forests (Sponseller et al. 2016). Inorganic N (primarily NO_3) concentrations are almost always below 70 µeq/l (1 mg NO_3-N/l), well below the threshold of 11.7 mg N/l as NO_3 stipulated by the EU Nitrates Directive. Given the high demand for N in most Nordic forests, water quality effects are hard to detect even a few hundred meters downstream of fertilised sites (Schelker et al. 2016).

Stem-only vs. whole-tree harvest

Clearcutting reduces the stand level demand for N and can promote increased N leaching which can be observed for as much as ten years following clearcutting (Likens et al. 1970; Kreutzweiser et al. 2008; Futter et al. 2014; Schelker et al. 2016). Compared to conventional harvest (SOH) removal of felling residues (WTH) reduces the leaching of N after the clearcut and can thereby counteract N accumulation resulting from atmospheric deposition (Lundborg 1997; Clarke et al. 2018). The magnitude and duration of N leaching is generally higher in more productive stands, and higher NO_3 leaching results in larger depletion of Ca in the soil pool. In a field study of the effect of SOH vs. WTH in Norway, Clarke et al. (2018) found that removal of needles and branches substantially reduced the leaching of N to soil water, presumably because there was less dead organic matter available for decomposition.

The management methods have very long-term effects. The SOH scenarios gave a larger depression of ANC and pH initially, while they recovered within 10–15 years after the clearcut to levels similar to those prior to the clearcut. The WTH scenarios had a long-term depression and did not return to levels prior to the clearcut by 2100, although the depression of ANC just after the clearcut was smaller (Figure 5).

Modelling studies suggest that WTH removes base cations from forest soils faster than they can be replaced by mineral weathering. Aherne et al. (2011) applied the MAGIC model predict the response of 163 Finnish lake catchments to historic and future atmospheric deposition (1880–2100) and future tree harvesting practices. They concluded that the current practice of SOH was close to the maximum sustainable harvesting under current legislated atmospheric deposition in Finland, and that WTH would imply significant long-term re-acidification of soil and surface waters. Zetterberg et al. (2014) used data from three Swedish forested catchments and the MAGIC model to simulate changes in forest soil exchangeable calcium pools and streamwater ANC. Large depletions in soil calcium supply and a reversal of the positive trend in stream ANC were predicted for all three sites after WTH. The greatest effect on soil and soil water, however, was observed at the well-buffered site, and not at the acidic sites where negative consequences for soil and surface water acidification would have been more pronounced. Another modelling exercise conducted by Moldan et al. (2017) on the potential effects of intensified forest harvesting on re-acidification of a set of 3239 Swedish lakes based on scenarios with varying intensities of forest biomass harvest and acid deposition. The MAGIC model results indicated that all plausible harvest scenarios would delay recovery due to increased rates of base cation removal, which led the authors to conclude that forest harvest intensity and regional environmental change must be carefully considered in future calculations of critical loads for acid deposition.

Greater biomass removal at harvest may have negative consequences for soil and surface water acidification as more intensive harvest removes more base cations than conventional harvest. This can have especially large consequences in poorly-buffered, slow-weathering forest soils, such as those at Birkenes. In a regional study of pine and spruce forests in Sweden, Akselsson et al. (2007) found net losses of Ca and Mg in almost the whole country for both SOH and WTH, whereas WTH in spruce forests led to substantially higher net losses of K and Ca than SOH. Long-term treatment differences in soil exchangeable calcium pools were examined by Zetterberg et al. (2013) at three coniferous sites in Sweden that were cut by conventional harvest and with whole-tree harvesting (WTH) in the mid-1970s. They found that soil water Ca concentrations were 40% lower in the WTH plots compared with the SOH plots. Long-term monitoring studies in Sweden found that the effects in soil acidification between SOH and WTH had decreased substantially after 15–25 years although not fully diminished (Brandtberg and Olsson 2012). Iwald et al. (2013) estimated the acidifying effect that various levels of harvesting of tree stumps and logging residues in Sweden and compared this with the acidification caused by acid deposition. They found that the acidifying effect of harvesting was 114–263% of that of acid deposition. The highest numbers were associated with WTH, and the higher acidifying effect of logging residues was explained by higher content of base cations in needles and branches compared to stem wood. The results from our model simulations at Birkenes are consistent with these field and modelling studies.

There were clear differences between the harvest scenarios and between the fertilisation scenarios. There appears to be a trade-off between SOH and WTH in an acid-sensitive catchment such as Birkenes; SOH would create a larger pulse of acidification but shorter-lasting, whereas WTH gives a smaller pulse but with longer duration. From a management perspective, this must be taken into consideration when addressing the potential effects of forest intensification as a climate mitigation measure in

Norway as there is a direct conflict with other environmental goals and objectives.

Long-term effects on carbon sequestration

Forests can influence the overall C balance of a region in several ways. Growing forests sequester atmospheric C in plant biomass. Forest management influences the rate of accumulation and loss of C in soil. For example, afforestation or fertilisation may increase rates of soil C sequestration while disturbance associated with clearcutting often leads to losses of soil C, especially in the organic layer (Covington 1981; Goodale et al. 2002; Yanai et al. 2003; Clarke et al. 2015).

The simulations here for Birkenes can be used to estimate the net changes in C sequestration in the ecosystem over the rotation period of the forest. The pools of C in biomass (565 molC/m^2) and soil (730 molC/m^2) were measured in 1990. Given the assumptions of continued incremental growth of the stems and build-up of new organic matter in soil due to N deposition, the amount of C stored in the ecosystem increased from about 1300 molC/ m^2 to 1430 molC/ m^2 over the period 1990–2018, an ecosystem C sequestration of about 4.5 molC/m^2/yr. The total stored was forecast to increase further by the year 2029 prior to clearcut to 1480 and 1510 molC/m^2 in the SOH and SOHfert scenarios, respectively. After clearcut (either by SOH or WTH) the C pool in tree biomass falls to zero and then begins to re-accumulate C with re-growth of the forest. In the case of SOH by the year 2080 (50 years following the clearcut), the ecosystem C pool again reaches the pre-clearcut levels, whereas in the case of WTH the ecosystem C pool is still lower than the pre-clearcut levels (Figure 7).

A complete assessment of the C budget for forest fertilisation requires consideration of the fate of the biomass (and thus C) removed from the forest after clearcut. If the removed stems and harvest residues (branches, needles) are burned or turned into products that have a short lifetime, then most of the C will return to the atmosphere. Clearcutting in this case will *increase* the net CO_2 emissions to the atmosphere, at least for the next few decades until the new forest regrows. But if the removed stems and harvest residues are used for construction materials or other long-term uses, then clearcutting will result in a net sequestration of C of about 7–8 molC/ m^2/yr and thus *decrease* the net CO_2 emissions.

de Wit et al. (2015b) have estimated that in Norway C sequestration in forests offsets anthropogenic C emissions by about 40%. This is partly because removal of C in biomass from forests does not keep up with accumulation of C in biomass. Liski et al. (2006) estimated that the amount of C stored in forests between 1922 and 2004 increased by 29% because of active forest management. Currently, forests are net sequesters of C also at the European scale (Valade et al. 2017). This assumes that the soil C lost following clearcutting is rebuilt during the growth of the forest in the next rotation period. In a review of 432 studies of changes in soil C pools following harvesting in temperate forests Nave et al. (2010) found that harvesting reduced forest floor C pools by about 20% in coniferous/mixed stands. Most of this was replenished during the next rotation period of the forest. For boreal forests such as that at Birkenes, there is a paucity of experimental data on the long-term impact of forest harvest intensity on soil C pools (Clarke et al. 2015). Our study indicates that the WTH method can result in net depletion of the soil C stocks that are not fully replenished over the rotation period.

Uncertainties

As is the case for most model simulations, there are many possible sources of uncertainties. The measured and estimated data used as model inputs may not be correct or may not be representative for the entire catchment. In this application to Birkenes, we have estimated dry deposition of both seasalt components and acidifying compounds. The estimates for dry deposition of N compounds is uncertain, in that there are no direct measurements and due to uptake of N compounds in the forest canopy, throughfall measurements do not help. Many of the necessary inputs values for forest growth have not been measured at Birkenes and thus by necessity were taken from the literature. Here we used many of the estimates of (Oulehle et al. 2019) for the forested catchment of Plesne Lake, Czech Republic. We assumed that the forest growth (and regrowth after clearcut) was similar to Plesne, albeit scaled to the longer rotation time at Birkenes. The turnover of below ground biomass is especially difficult to estimate. The evolution of the soil C and N pools over time are difficult to assess and present a source of uncertainty in the specification of effects of the forest harvesting. As for the fertilisation, it is not known how the added NH_4NO_3 is incorporated into the system. We assume that growth is promoted by about 10% and that the trees thus take up some of the added N, but the rest must be stored in the soil, without passing through an aqueous phase, as very little leaches immediately out in the stream.

The model itself is, of course, a simplification of reality and may be in part incorrect. The acid–base biogeochemistry aspects of MAGIC have proven to be quite robust and have been evaluated many times by whole-catchment experiments (Wright et al., 1990) and comparison with long-term data series (such as the series here from Birkenes) (Helliwell et al. 2014). The forest growth and soil C and N pool aspects are less robust, in part due to insufficient knowledge to properly quantify the processes affecting the pools and fluxes of C and N in forest ecosystems.

There are several assumptions in the MAGIC applications here that add uncertainty to the results. The weathering rate of base cations, the soil cation exchange capacity (CEC), and the cation selectivity coefficients are all assumed to be constant throughout the 200-year simulations. These interact and affect the stream water chemistry as well as the availability of base cations for uptake by the vegetation. Unfortunately, there are no direct measurements over time for these at Birkenes.

As suggested by Zetterberg et al. (2014) weathering rates may increase following forest harvesting, perhaps by

increased root activity on new mineral surfaces. This would act to enhance the base cation pools in the soil and thus counteract acidification of surface water. There are no data for any such changes in weathering rates. The CEC is linked to soil C as many of the exchangeable cation sites are associated with soil organic matter. Loss of soil organic matter following forest harvesting thus entails a decrease in CEC, followed by a long-term build-up in the next generation of forest. Changes in the CEC, however, do not directly affect the pools of soil base cations. A decrease in CEC following forest harvest will be manifest by an increase in the %BS in the soil, but the pools of base cations in the soil are unchanged. As the %BS affects the relative amounts of base to acid cations leached from the soil, the temporary decrease in CEC following forest harvest may lead to higher concentrations of base cations and higher pH in runoff. The MAGIC simulations presented here do not take into account such changes in CEC.

Both the height and duration of the NO_3 peak in streamwater are dependent on the assumptions and uncertainties inherent in this modelling exercise. The NH_4 resulting from increased mineralisation of organic N in decomposing logging residues will to a great extent be assimilated by soil microbes and ground vegetation, adsorbed to soil particles, or nitrified to NO_3. The model assumption that all NH_4 in soil solution (not retained by other sinks) was nitrified to NO_3 is based on observations that NH_4 concentrations in upland streams usually are very low compared to NO_3, even after forest clearcut (Likens et al. 1970). The relative shapes and heights of the NO_3 peak for the various scenarios are more certain than the absolute concentrations. For example, stem-only harvest gave a higher NO_3 peak compared to whole-tree harvest. Thus, the simulations presented here cannot be taken as "absolute".

Moderating factors

There are several moderating factors that might reduce the adverse effects of intensified forestry on surface water quality. The simulations for the Birkenes catchment indicate severe acidification of the streamwater for several years following clearcut of the entire catchment. In practice, however, conventional forest harvesting in Norway (and elsewhere) proscribes buffer strips of untreated forest along streams and lakes. These will damp the NO_3 and acid peaks in the streamwater (Laudon et al. 2016). Secondly, there is a scale effect: clearcut areas often cover only relatively small fractions of the catchments to larger streams, rivers and lakes, the adverse effects on surface waters will be much reduced downstream (Schelker et al. 2016). Indeed, clearcutting has been going on for decades in Norway and elsewhere in the boreal forest, without any large-scale measurable impact on surface water chemistry of larger streams and lakes. For a large river basin covered by forest with a rotation time of 50 years, on the average about 2% of the catchment will be clearcut in any given year. If the simulated peak of 100 µeq/l in NO_3 concentrations after clearcut at Birkenes is typical, the clearcutting 2% of a catchment would result in a NO_3 increase of only 2 µeq/l, much too small to significantly affect acidification and well within the observed year-to-year variations in monitoring data from Norwegian surface waters.

Nevertheless, the adverse effects might be significant in small streams if large parts of the catchment are clearcut. The released NO_3 will, of course, have the largest impact in acid-sensitive ecosystems, especially those that have been acidified by long-term chronic acid deposition. The forested catchment at Birkenes is such an ecosystem. Clearcutting of major fraction of the forest here would cause re-acidification of the outflowing stream with adverse biological effects.

Conclusions and implications for management

The ambition to use forestry as a climate mitigation measure should take into consideration the potential effects of forestry management methods on soil and surface water quality, in particular in acid-sensitive areas such as southern Norway. This modelling study at Birkenes demonstrates that intensified forestry may cause substantial effects on surface water quality in acid-sensitive areas. The SOH scenario had a more pronounced effect on N leaching than the WTH scenario, but the NO_3 peak diminished a few years after the clearcut. Due to poorly buffered soils at the study site, the NO_3 leaching resulted in a transient, but significant depression of ANC and pH. The greater biomass removal with the WTH scenario resulted in a long-term depletion of in soil base cations and a setback in the positive trend in stream ANC by several decades. The simulated N fertilisation ten years before harvest was not followed by an immediate N pulse in surface water, but gave elevated N leaching after clearcut, both with the SOH and WTH scenarios.

It must be noted that the Birkenes study represents an extreme case with 100% clear cut of a catchment. In reality, only a small fraction of a catchment will be cut each year, and it is usual to establish buffer zones along streams with permanent flow. Even though the effects following harvest can be substantial in first-order streams, the signals are relatively short-lived and are rapidly weakend further downstream in stream networks. Nevertheless, it is important that forest and environmental managers carefully consider surface water sensitivity to acidification when selecting sites for fertilisation and deciding on harvesting methods.

We recommend that if forest fertilisation or other management methods such as WTH are to be used in future climate mitigation efforts, the risks to soil and surface water quality should be evaluated, in particular, in acid-sensitive areas to ensure that surface waters are safegaurded from the potential negative effects of forest management.

Acknowledgements

We thank Wenche Aas (Norwegian Institute for Air Research) for deposition data, Nicholas Clarke (Norwegian Institute of Bioeconomy Research) for forest data, B. Jack Cosby (Centre for Ecology and Hydrology, UK) for help with the MAGIC modelling, and Filip Oulehle (Czech Geological Survey, CZ) for data from the Plesne Lake study.

Disclosure statement

No potential conflict of interest was reported by the author.

Funding

This work was supported by the Research Council of Norway through the SURFER project [NFR project number 267481], NordForsk through the BIOWATER project [NF project number 82263] and the Norwegian Institute for Water Research NIVA.

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