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Impacts of forest biomass removal on soil nutrient status under climate change: a catchment-based modelling study for Finland

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Abstract The environmental impact of different forest harvesting scenarios on soil nutrient status and water chemistry under current and future (IPCC A2) climate was evaluated for a random sample of lake catchments ($n = 1066$) covering Finland. Biomass removal scenarios were derived from a management-oriented large-scale forest model based on data from national forest inventories. Forest ecosystem sustainability was assessed by evaluating soil base cation balances as well as temporal changes (2010–2050) in soil base saturation and lake water acid neutralising capacity, using a dynamic hydro-geochemical model. The harvesting scenarios had very different effects on biomass and element removal as well as soil and water quality; only harvesting of above-ground woody biomass (stem-only or stem-and-branches harvesting scenarios) was predicted to be sustainable, i.e. not depleting the soil base cation pools in the long term. The most intensive scenario—whole-tree harvesting (including the removal of stumps and roots)—doubled the removal of biomass, tripled the removal of base cations from the catchment soils, and increased nitrogen removal fourfold. Climate change was predicted to have a positive impact by increasing the future supply of base cations from weathering, thus compensating their removal by biomass harvesting. However, additional inputs of nitrogen and potassium will be required to ensure sustained forest growth under intensive biomass harvesting.

Keywords Forest harvesting · Residue removal · Biofuel · IPCC A2 climate · MAGIC · Base saturation · Lake ANC · Nutrient limitation

Introduction

The use of biofuels is increasingly regarded as an important mitigation strategy against the effects of climate change. For example, the European Union has set a mandatory target of 20% for the share of energy from renewable sources in the overall energy consumption by 2020. (Directive 2009/28/EC). However, many biofuels compete with food production for land resources. Furthermore, some methods of biofuel production are seen as conflicting with environmental protection goals. As a consequence,

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a world-wide debate on the merits and trade-offs of biofuel production is underway (see, e.g., Tilman et al. 2009).

In Finland, with its large forest resources (covering >70% of the country) and highly developed industry, there is an increased interest in the use of forest harvesting residues for biofuel production. Forest biomass (non-merchantable timber) harvested for energy use includes living and dead branches, foliage, stem tops and off-cuts, stumps and roots, which are converted to chips in the forest, at the roadside or at the site of end-use. Finnish government policy aims to increase the use of renewable energy to 38% of energy consumption by 2020 and to 60% by 2050 (Climate and Energy Strategy 2008), which includes an increase in forest chips from 4 to 12 million m^3 between 2008 and 2020 (METLA 2009). The governmental Foresight Study includes even more ambitious targets (Foresight report 2009), with an increase in the use of wood energy from 19.3 TWh in 1997 to a maximum of 55.8 TWh in 2050.

Forest harvesting may influence soil and lake water (chemical) quality by removing base cations (Bc) and nitrogen (N) from catchment soils. Long-term ecosystem sustainability requires that these Bc losses are balanced by atmospheric deposition and weathering (e.g., see Sverdrup and Rosen 1998; Joki-Heiskala et al. 2003; Akselsson et al. 2007). In addition to forestry practices, it is well established that (long-range transport of) nitrogen and sulphur deposition can influence soil and water chemistry in forested catchments (Forsius et al. 2003; Posch et al. 2008). Moreover, all these processes are influenced by changes in the global climate (e.g., Park et al. 2010). Dynamic hydro-geochemical models can be used to analyse the combined impacts of climate change, acidic deposition and forest harvesting practices, and for assessing the impacts of different future scenarios (Forsius et al. 1997; Aherne et al. 2008; Posch et al. 2008; Moldan et al. 2009).

The objective of this paper was to evaluate the impacts of forest biomass removal and climate on the nutrient status of soils and water chemistry in Finnish lake catchments for the period 2010–2050. In particular, the impact of different levels of forest residue removal (for biofuel production) under current and future climate was evaluated. Forest biomass removal scenarios were derived from the management-oriented large-scale forestry (MELA) model and based on sample plot and tree data from the tenth National Forest Inventory. These scenarios were developed to guide future Finnish forest management with respect to biofuel use. Sustainability under each forest harvesting scenario was assessed by evaluating soil Bc balances and lake water acid neutralising capacity (ANC). The influence of forest biomass removal on carbon balances and greenhouse gas budgets have been previously investigated, e.g., in Sievänen et al. (2007) and Repo et al. (2010).

Materials and methods

Study sites

During 1987, a country-wide lake survey was conducted under the Finnish Research Project on Acidification (HAPRO; see Kauppi et al. 1990). A random sample of 987 lakes, from the ~56,000 lakes in Finland >1 ha (with ~15,700 >10 ha), were sampled during fall overturn and analysed for all major ions in the laboratories of the (then) National Board of Waters and the Environment. Statistical procedures for lake selection, sampling protocols, analytical methods and quality control procedures are described in detail by Forsius et al. (1990). An analysis and discussion of the lake water chemistry is given by Kortelainen et al. (1989) and Kämäri et al. (1991).

North of 66.13° (7340 km in the Finnish coordinate system; shown by a line in Fig. 1) only lakes >10 ha were sampled, owing to logistical limitations and the very large number of small lakes in Finnish Lapland (Forsius et al. 1990). Thus, the data set was supplemented by a comprehensive set of lakes sampled around the same time by the water and environment district office of Lapland. This combined set of lakes constitutes the basis for the assessment of critical loads for surface waters in Finland under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP; Posch et al. 1997). In the current study, only lakes north of the 7340-line and <10 ha were selected from the Lappish lake set. Moreover, the number of (small) lakes was limited so that their proportion was the same as that south of the 7340-line, resulting in a total of 1066 lake catchments (Table 1 and Fig. 1). Not all variables were available for the additional Lappish lakes, e.g., total organic carbon (TOC) was estimated from a linear regression with

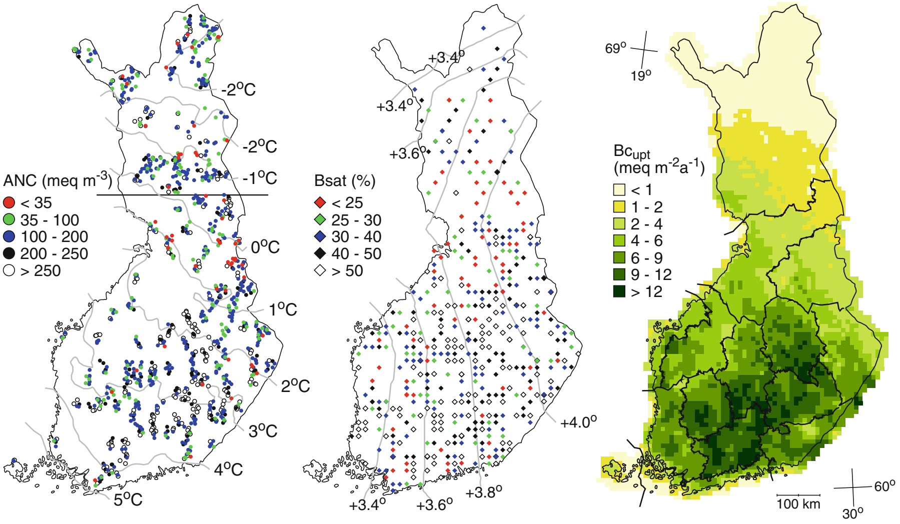

Fig. 1 *Left map* Location of the study lakes ($n = 1066$) stratified according to lake acid neutralising capacity (ANC); isolines of the long-term (1961–1990) annual mean temperature are also shown. North of the *horizontal black line* Lappish lake survey data are included. *Centre map* Location of the soil survey plots ($n = 488$) stratified by percent base saturation (Bsat) ranges; isolines of the annual mean temperature *increase* by 2050 under the A2 climate scenario are also shown. *Right map* Gridded (10 × 10 km) base cation ($Bc = Ca^{2+} + Mg^{2+} + K^{+}$) annual mean net uptake by forests (removal by harvesting) in 2010; borders of the 15 Forest Centre districts are also shown

measured chemical oxygen demand (COD; Kortelainen 1993).

A comprehensive assessment of the impacts of forest harvesting removal on soil and surface water chemistry under future atmospheric deposition and climate (incorporating the influence of a changing climate on biomass growth and weathering) was carried out at each study catchment. Historic and future (1880–2050) soil and surface water chemistry under several harvesting scenarios was simulated using a hydro–geochemical model; the required model inputs were obtained from site-specific observations, data interpolated from spatially explicit data bases and output of the MELA forestry model. In general, data procedures follow Aherne et al. (2008).

Model description

The model of acidification of groundwater in catchments (MAGIC) is a lumped-parameter model of intermediate complexity, developed to predict the long-term effects of acidic deposition on soil and surface water chemistry. The model was first described by Cosby et al. (1985) and developments are reviewed in Cosby et al. (2001). The model predicts annual average concentrations of the major ions for soil solution and surface water chemistry.

MAGIC represents the catchment with spatially and vertically aggregated soil compartments (up to three), and a surface water compartment that can be either a lake or a stream. Time series inputs to the model include: deposition of ions from the atmosphere (wet plus dry deposition); discharge volume and flow routing within the catchment; net removal of nutrients (N and Bc) by harvesting; internal sources and sinks of ions from weathering or precipitation reactions; and meteorological data. Constant parameters in the model include physical and chemical characteristics of the soils and surface waters, and thermodynamic constants. Soil Bc weathering rate and initial Bc exchangeable fractions are calibrated (using an automated optimisation procedure) against observed (or 'target') values of surface water and soil chemistry for a specified period. The minimum required site-measured target variables are: surface water concentrations for the major ions and soil exchangeable fractions for calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}) and potassium (K^{+}).

In the current study, all catchments were represented by one soil compartment receiving deposition

Table 1 Statistical summaries (fifth percentile, median, 95th percentile) of catchment characteristics, catchment-average soil properties, climate normals (1961–1990) and lake chemistry (1987) for the study catchments ($n = 1066$; see Fig. 1)

Variable	Unit	5 percentile	Median	95 percentile
Catchment area	ha	28.0	185.5	5932.5
Forest area	% terrestrial area	34.3	70.8	84.1
Peatland area	% terrestrial area	0.9	19.3	63.7
Soil depth[a]	m	0.38	0.56	0.72
Lumped soil bulk density[a]	kg m^{-3}	701.6	1090.7	1324.2
Soil pH	pH units	4.24	4.74	5.07
Cation exchange capacity[b]	meq kg^{-1}	9.13	17.50	59.03
Soil base saturation[a]	% CEC	26.57	39.51	76.62
Exchangeable calcium[a]	% CEC	17.60	26.72	49.01
Exchangeable magnesium[a]	% CEC	4.81	7.94	24.90
Exchangeable sodium[a]	% CEC	0.76	1.30	1.80
Exchangeable potassium[a]	% CEC	1.69	2.83	4.52
Lake area	ha	1.8	8.3	120.8
Maximum lake depth	m	2.2	4.8	18.4
Retention time	yr	0.03	0.33	2.72
Catchment discharge	m	0.233	0.321	0.414
Precipitation	m	0.446	0.599	0.654
Temperature	°C	−2.7	1.2	3.5
Calcium	μeq L^{-1}	29.94	119.76	349.30
Magnesium	μeq L^{-1}	18.51	65.82	172.77
Sodium	μeq L^{-1}	26.10	60.90	130.49
Potassium	μeq L^{-1}	2.56	12.79	46.04
Ammonium	μeq L^{-1}	0.14	0.71	5.84
Sulphate	μeq L^{-1}	22.91	62.49	207.77
Chloride	μeq L^{-1}	8.46	25.39	117.77
Nitrate	μeq L^{-1}	0.00	0.86	10.00
Acid neutralising capacity[c]	μeq L^{-1}	35.38	164.99	385.31
pH	pH units	4.84	6.30	7.09
Labile aluminium	μeq L^{-1}	0.00	1.45	8.78
Total organic carbon	mg L^{-1}	3.06	10.90	27.38
Gran alkalinity	μeq L^{-1}	−16.0	75.0	278.5
Conductivity	μS cm^{-1}	13.0	29.0	72.0

[a] Catchment soil characteristics are summarised from the closest permanent sampling plots in the eighth National Forest Inventory, [b] Effective cation exchange capacity (CEC), [c] Acid neutralising capacity (ANC) was estimated as the sum of base cations minus the sum of strong acid anions

and releasing discharge to the lake compartment. The soil compartment represented the aggregated (lumped) horizons of the catchment soils, i.e., mineral and organic soils were not segregated within the modelling structure. Simulations were carried out using an annual time-step, with a number of simplifying assumptions applied consistently across all study lakes. Discharge volume was not varied during the calibration (and hindcast) period; it was described using long-term means (see below) with 100% routed to the lake. Detailed process-oriented N dynamics were not modelled—instead a fixed catchment-specific (percent) retention was estimated from input–output observations. The median nitrate concentration of the study lakes is only 0.86 μeq L^{-1} (Table 1) and increasing trends in nitrate for acid sensitive lakes in Finland have not been observed (Vuorenmaa and Forsius 2008). Future changes in N emissions are not expected to have a major impact on the acidity status of Finnish lakes (Forsius et al. 2003) and, therefore, are given limited attention in the present study. Further, it was assumed that N deposition will not lead to N leaching in the future. In contrast, N is generally a growth-limiting factor in boreal ecosystems; accordingly fertilizer amendment (NPK) during forest establishment is a common management practice (depending on site quality). In this context, a simple mass balance approach was employed to explore potential N limitations under each forest harvesting scenario.

Simulations were carried out between 1880 and 2010 (hindcast) and 2010 to 2050 (forecast). Essential data for the current study were time-series of annual atmospheric deposition (wet plus dry), future climate and discharge (IPCC A2 scenario), biological production and removal (forest harvesting), physical and chemical characteristics of the soils and lakes, and observed soil and lake chemistry (calibration targets) for each study lake (see Table 1).

Data sources

Soils

Soil physico-chemical properties were described using a network of permanent plots ($n = 488$) located throughout Finland, established under the eighth national forest inventory (8NFI). The plots were systematically laid out at the intersections of a 16 × 16 km grid in the southern part of the country (<65° 30′ N) and a 24 × 32 km grid in northern Finland (>65° 30′ N; see Fig. 1 for sampling density). Samples of the humus layer ($O_f + O_h$) and 0–5, 5–20 and 20–40 cm mineral soil layers were taken systematically from five pits at each plot and composited into a single sample for each layer. From the 60–70 cm layer a single sample was taken from one of the five soil pits (Tamminen and Starr 1990). Soils were analysed for exchangeable cations, exchangeable acidity, organic matter (OM), carbon (C) and N content according to standard procedures. Base saturation was calculated as the sum of base cations divided by the effective cation exchange capacity (CEC). Further details of sampling and analyses are given in Tamminen and Starr (1990).

Lumped (pit average) soil physico-chemical data for each of the permanent plots were derived by weighting soil properties by depth and density (see Aherne et al. 2008). Bulk density was estimated from soil OM using different empirical relationships (pedotransfer functions) for organic and mineral soil layers. The permanent soil plots do not, in general, lie within the study catchments; mineral soil properties for each study catchment were defined by interpolating (averaging) the three (geographically) nearest soil plots. Peatlands are present in the majority of the catchments; as such, mineral and organic soil data were spatially-weighted to derive the final catchment average (Table 1). Soil physico-chemical data for organic soils are not included in the 8NFI, instead generic organic soil data were derived from literature studies; further the hydro-chemically active depth for peat soils was assumed to be 30 cm (see Aherne et al. (2008) for further details). The lumped catchment soil characteristics were strongly dependent on the physico-chemical properties of the mineral soils. Peat soils and organic forest soil horizons have low bulk densities and accordingly had limited influence on the properties of the final catchment average soils.

Deposition

The spatial distribution of S and N deposition to average land-cover on a 50 × 50 km grid resolution for 1985 and 1990 were taken from the output of the EMEP/MSC-W eulerian dispersion model (Simpson et al. 2003). The deposition of base cations (Ca^{2+}, Mg^{2+}, K^+ and Na^+) and Cl^- were available at a resolution of 0.250 × 0.125° for the period 1986–1990, based on interpolated observations from a nation-wide network of approximately 38 stations measuring monthly bulk deposition (Järvinen and Vänni 1990).

Initially, catchment-specific estimates of base cations, Cl^-, S and N deposition during the calibration year (1987) were estimated by interpolating (inverse distance averaging) the surrounding grid values. These data represent bulk deposition or total deposition to an average land-cover; however, depending on catchment characteristics, such as land-cover, elevation, aspect, etc., site-specific deposition may be quite different due to interception and enhancement processes. Sulphate (${SO_4}^{2-}$) and Cl^- inputs were assumed to be conservative with respect to input–output fluxes, i.e., they were not retained in soils or lakes. As such, the total ${SO_4}^{2-}$ input (in deposition) was set to match the ${SO_4}^{2-}$ output from the lake catchments (Aherne et al. 2008). Any difference in the input–output balance for ${SO_4}^{2-}$ and Cl^- was attributed to (unmeasured) dry deposition. Imbalances in Cl^- were assumed to be of sea-salt origin, and all ions were adjusted in proportion to their sea-salt ratios with Cl^-. Imbalances in ${SO_4}^{2-}$ were attributed to anthropogenic sources and assumed to be balanced by anthropogenic Ca^{2+} deposition.

Sulphur and N deposition sequences on a 50 × 50 km grid resolution for the period 1880–2010 were obtained from Schöpp et al. (2003). The historical pattern was relatively consistent over Finland; therefore, a single relative deposition curve for each pollutant was used at all study sites. The modelled deposition showed reasonable agreement to observed data at 19 bulk deposition monitoring stations (Vuorenmaa 2004; for more details see Aherne et al. 2008). The deposition sequences were scaled to unity for the year 2010 (reference year). Combining these curves with deposition in the year 1987 (calibration year) allows the calculation of S and N deposition at every (grid) location in Finland at any time between 1880 and 2010. Calcium and Mg^{2+} deposition were assumed to follow the same historical sequence as S (see Aherne et al. 2008), the remaining base cations and Cl^- were assumed constant throughout the simulation period. Future (2010–2050) S and N deposition was based on a 'Current Legislation' (CLe) scenario that assumed the implementation of the 1999 Gothenburg Protocol of the LRTAP Convention as well as the EU National Emissions Ceiling (NEC) Directive. The scenario was phased in linearly until 2020 and assumed constant thereafter.

Climate and discharge

Long-term average annual climate (1961–1990) for each study catchment was derived using precipitation and temperature data from the Finnish Meteorological Institute. Further, these data were used as inputs to the Finnish watershed simulation and forecasting system (WSFS) to estimate long-term (1961–1990) catchment discharge or runoff (Vehviläinen 2007). In the current study, long-term climate normals for temperature, precipitation and discharge were obtained from the FINESSI project (www.finessi.info), which provided data for all of Finland on a 10 × 10 km grid resolution. The long-term normals for each study catchment were taken from the values of the grid cell within which the lake was located (see Table 1).

Future climate (temperature and precipitation) was derived from the ECHAM4/OPYC3 (Max Planck Institute, Germany) general circulation model (GCM) under the A2 emission scenario of the Intergovernmental Panel on Climate Change. The A2 scenario represents a more pessimistic future (highest predicted change) with high emissions of greenhouse gases (Nakićenović et al. 2000). However, current carbon dioxide (CO_2) emission trends exceed even the highest IPCC scenario (Raupach et al. 2007), and therefore the A2 scenario may be viewed as a more 'realistic' future compared with an unchanged (1961–1990) climate (albeit potentially conservative). Under FINESSI, temperature, precipitation and discharge were available for three 30-year periods: 2010–2039, 2040–2069 and 2070–2099. In the current study, temperature and discharge were assumed to change linearly between the midpoints of these periods (2025 and 2055). In contrast to the deposition scenario, which used a common future sequence for all sites, individual temperature and discharge scenarios were derived for each study site. Temperatures were predicted to increase 3.3–4.1°C by 2050 (see Fig. 1); precipitation increases were also pronounced, but less uniform. Accordingly, changes in modelled (WSFS) discharge were quite small because higher temperatures 'counteracted' higher precipitation by increased evapotranspiration (Posch et al. 2008).

Forest growth and harvesting scenarios

The harvest (removal) of biomass from Finnish forests was available for three (groups of) species: pine, spruce and broad-leaved (mostly birch). Historic data during the period 1955–2005 (METLA 2009) show a clear increasing trend (almost a doubling of biomass harvesting during 50 years, mainly due to an increase in the forest area owing to the drainage of peatlands and agricultural set-asides), but also a strong inter-annual variability due to climatic and economic factors (Fig. 2). Future projections of forest growth and harvesting until 2050 were obtained from the MELA model (Redsven et al. 2004; Kärkkäinen et al. 2008). The 'business-as-usual' (BAU) scenario assumed no change in policies and climate (i.e., continued cuttings at present-day level). Historically and under the BAU scenario not all of the annual wood increment was harvested, i.e., the growing stock has been increasing, and is projected to increase further under the BAU scenario. Accordingly, a 'sustainable cutting' (SuC) scenario maximising the net present value of timber production without decreasing the flow of wood (and income) during the period 2020–2050 was also produced by MELA. Further details, including

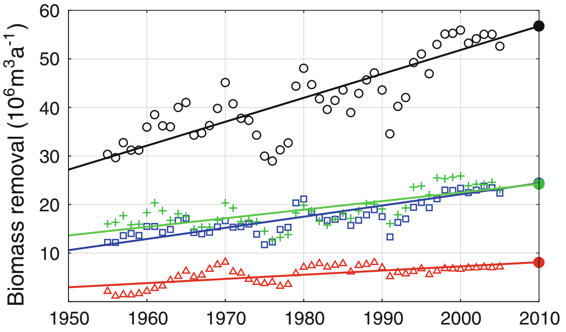

Fig. 2 Historic (*open symbols*) and 2010 (BAU scenario, *filled circles*) annual biomass removal (10^6 m^3a^{-1}) for pine (*blue square*), spruce (*green plus*), broadleaves (*red triangle*) and their sum (*black circle*) in Finland. The regression lines forced through 2010 were used to estimate net biomass removal during the period 1950–2010

economic aspects of this scenario, can be found in Kärkkäinen et al. (2008). The overall amount of biomass removed under the SuC scenario during 2020–2050 was 41.5–45.5% higher than under the 2010 BAU scenario.

Under the BAU and SuC scenarios only stems (i.e. stem-wood and bark) were harvested (stem-only harvesting, SOH). However, data on the mass (Gg a^{-1}) of the remaining tree compartments (left-over stems and bark, foliage, live and dead branches, stumps and roots) were also available from MELA for each Forest Centre (15 Centres, see Fig. 1), based on biomass equations by Marklund (1988). These regional biomass (removal) data were allocated to the 10 × 10 km grid by using detailed forest area information from the 8NFI and by weighing them with the average 1961–1990 effective temperature sum (ETS) above 5°C to ensure that in the north of a region (i.e., Forest Centre) there is less growth/harvest per unit area than in its southern part.

In addition to the BAU scenario, the impacts of four SuC-based scenarios with increasing biomass removal were investigated in the current study: (a) SOH: stem-only harvesting, (b) SBH: stems plus branches harvesting (SOH + remaining stems and bark plus branches), (c) AGH: above-ground harvesting (SBH + foliage), and (d) WTH: whole-tree harvesting (AGH + stumps and roots). Biomass removal was assumed to change linearly between the 10 year intervals obtained from MELA during the period 2020–2050. Furthermore, the SuC-based scenarios starting in 2020 were assumed to increase linearly from 2010 BAU. The temporal removal (harvest) of Bc and N for each scenario was estimated using element concentrations in the different tree compartments (Table 2). These data (ICP Integrated Monitoring 2004) were based on a compilation of Nordic sources (e.g., Bringmark 1977; Mälkönen 1977; Finér 1989; Finér and Brække 1991). The gridded base cation (Ca^{2+} + Mg^{2+} +K^{+}) annual mean net uptake (removal by harvesting) reflects the north–south gradient in forest growth as well as the distribution of forests in Finland (see Fig. 1).

Model calibration and future scenarios

Site-specific parameter files were prepared using catchment and soil physico-chemical characteristics based on, or estimated from, field observations (lake area, lake retention, soil bulk density, soil cation exchange capacity, etc.), and considered 'fixed' in the model (see Table 1). A number of default parameter values were uniformly applied across the region (see Aherne et al. 2008). Furthermore, a number of model parameters were calibrated to observed data: lake DOC concentrations, pCO_2 and aluminium solubility constants were calibrated to observed lake pH, bicarbonate (alkalinity) and Al^{3+} concentrations, respectively. The coefficient for N in-lake retention was set to 5 m a^{-1} for all lakes. Soil and surface water nitrification was set to 100%. Nitrogen (NO_3^- and NH_4^+) retention (or uptake) in the terrestrial and lake compartments was set to match observed lake concentrations. Nitrogen transformations were described as a catchment net retention calculated simply as the difference between input and output flux during the calibration year. This percentage was assumed to be constant throughout the simulation. Base cation weathering rates and initial soil exchangeable fractions were calibrated using numerical techniques to select parameter values that result in a minimum sum of squares error between simulated and observed target variables for each catchment in the calibration year (1987).

Future soil and surface water chemistries for each lake catchment were simulated for the period 2010–2050 under CLe deposition, current and A2 climate, and five forest harvesting scenarios (BAU, SOH, SBH, AGH, WTH; see above). Processes such as nutrient uptake (forest growth) and release (weathering) are influenced by climate change. The influence upon forest growth was accommodated using

Table 2 Element concentrations (g kg^{-1} dry weight) in tree compartments for pine, spruce and birch (source: ICP Integrated Monitoring 2004)

Species	Compartment	Ca g kg^{-1} dry weight	Mg	K	N	Mass (%)
Pine	Stem wood	0.48	0.15	0.24	0.59	58.5
	Stem bark	3.46	0.50	1.19	3.83	4.4
	Needles	3.02	0.90	4.04	13.67	3.7
	Live branches	2.50	0.60	2.33	5.12	9.5
	Dead branches	2.06	0.14	0.26	3.51	1.8
	Stump + roots	0.39	0.12	0.65	0.74	22.0
	Whole tree	1.22	0.27	0.70	2.35	100.0
Spruce	Stem wood	0.73	0.11	0.26	0.72	49.3
	Stem bark	9.60	0.74	1.92	4.09	4.8
	Needles	4.63	1.02	4.78	12.95	8.8
	Live branches	4.39	0.62	2.28	5.80	14.4
	Dead branches	4.71	0.12	0.27	3.28	1.3
	Stump + roots	1.44	0.16	0.88	1.49	21.4
	Whole tree	1.92	0.27	0.97	2.57	100.0
Birch	Stem wood	0.54	0.15	0.27	1.04	53.9
	Stem bark	4.33	0.38	0.93	5.13	9.3
	Leaves	8.02	2.75	6.45	22.90	3.4
	Live branches	3.61	0.57	0.99	4.98	16.2
	Dead branches	3.20	0.33	0.39	4.52	0.8
	Stump + big roots	1.83	0.31	0.58	3.52	16.4
	Whole tree	1.58	0.30	0.62	2.66	100.0

The mass percentages of the different compartments were computed from the country-average MELA harvest residue data for 2010; and these, in turn, were used to compute the mean 'whole tree' element concentrations. These mass percentages also allow estimating total element pools

modifying factors that incorporate temperature and respiration effects (Appendix A). These factors were taken from the C-Fix model (Veroustraete et al. 2002) and have been previously used in a European-scale dynamic modelling assessment of forest plots (Reinds et al. 2009). The temperature-dependence of basic chemical weathering reactions was modelled using Arrhenius-factors (Sverdrup and Warfvinge 1993; see Appendix B), resulting in higher weathering rates with increasing temperatures (e.g., Sverdrup 1990; Brady 1991). The reference or 'base' scenario was composed of the CLe (deposition), current climate and BAU forest harvesting scenarios, which represent the 'present' future, based on current legislation and current forest practices. The future state of soil and surface water chemistry (until 2050) was evaluated using two indicators: ANC of lake water and base saturation of the catchment soils, which are key indicators for ecological impacts used in European-scale assessment work on air pollution (Hettelingh et al. 2007). Lake ANC is a standard indicator used in acidification studies, as it is strongly indicative of biological recovery, and is the most widely used chemical criterion in critical load calculations for surface waters (Henriksen and Posch 2001). Base saturation, on the other hand, is a key indicator of soil nutrient status.

Results and discussion

Impact of biomass harvesting scenarios on nutrient removal from soils

During the period 1950–2010, the harvesting of forest biomass (stems) more than doubled (median increase from 0.305 to 0.625 tons ha^{-1}a^{-1}; upper-right quadrants Fig. 3) resulting in a proportional increase in the removal of base cation (median increase from 2.11 to 4.52 meq m^{-2}a^{-1}; Fig. 3). Moreover, the shift from BAU to SuC resulted in a predicted increase in the removal of biomass (stems) as well as base cations by about 50% in 2050 (lower-right quadrants in Fig. 3). Despite considerable changes in the proportion of species grown and harvested between 2020 and 2050 (Kärkkäinen et al. 2008) the overall amount harvested annually under the SuC

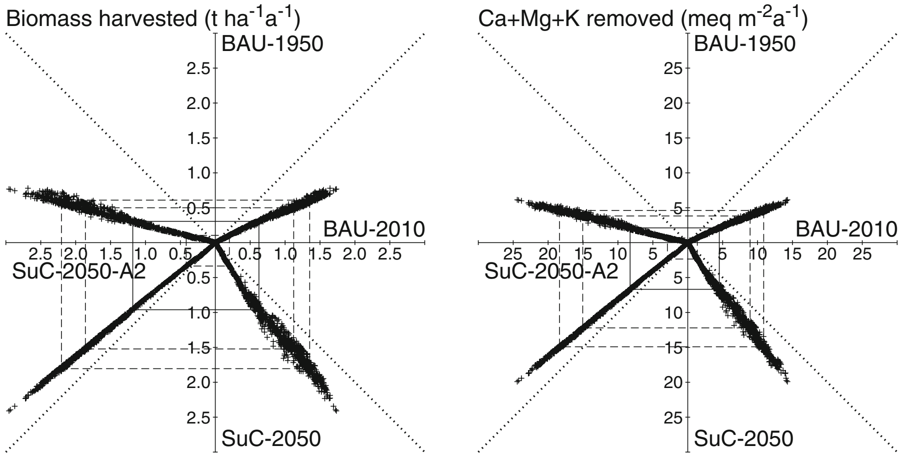

Fig. 3 Quadruple correlations (‘quarrelations’) of harvested biomass (*left*) and nutrient removal ($Ca^{2+} + Mg^{2+} + K^{+}$: *right*) in the study catchments ($n = 1066$) for the BAU scenario in 1950 and 2010, and the SuC scenario in 2050 without and with climate change, all based on stem-only harvesting (SOH). The rectangles connect the 25th, 50th (*solid line*), 75th and 90th percentiles of the four data sets

scenario hardly changed (not shown). The change in species composition, and thus element content, resulted in slightly different correlations for base cations (Fig. 3 right). Under SuC and A2 climate, the increase in stem growth, and thus harvesting, is about 20% in 2050 (lower-left quadrants in Fig. 3), resulting in a four-fold increase in biomass removal since 1950 (upper-left quadrants in Fig. 3).

The four harvesting scenarios (SOH, SBH, AGH and WTH) predicted very different biomass and element removal (Fig. 4). The removal of stems, bark and all branches (scenario SBH) increased biomass harvest by a third (compared to SOH), and doubled the removal of base cations and N. In contrast, the removal of foliage in addition to SBH (scenario AGH) had little influence on biomass, but increased Bc and N removal by 50%, owing to the larger element concentrations of foliage (Table 2). Finally, removing stumps and roots hardly increased Bc and N removal but increased harvested biomass by about 20% (lower-left quadrant in Fig. 4), albeit with the danger of potentially major ecosystem disruptions (Walmsley and Godbold, 2009), such as increased leaching of base cations and N from soils. The shift from SOH to WTH approximately doubled the removal of biomass, tripled the removal of base cations and quadrupled the removal of N from the study catchments (upper-left quadrant in Fig. 4).

It is important to note that the MELA scenarios are primarily focused on stem wood prediction owing to the established forest industry using saw logs and pulpwood in Finland. Predictions for other biomass compartments are less reliable, as they are considered a by-product. Invariably these scenarios will change as bioenergy production becomes a profitable option. However, from this theoretical supply not all biomass removal is (currently) feasible, owing to logistical and ‘good forestry practice’ limitations. In general, harvesting of branches or stumps and roots is limited to more fertile sites. Furthermore, in practice harvesting of branches only captures about 70% of the available biomass.

Weathering rates under current and changed climate

Mineral weathering represents a major long-term supply of Bc to forest ecosystems, as such (Bc) weathering is a key determinant of long-term ecosystem sustainability. In the current study, site-specific weathering rates were calibrated (optimised) against observed values of Bc in surface water and soil chemistry. Calibrated weathering rates for the study sites ranged from 3 to 430 meq m^{-2} a^{-1} (mean: 57 meq m^{-2} a^{-1}; see Fig. 5). Weathering rates varied according to site-factors, with high values found throughout the country (see Aherne et al. 2008). Under climate change, the estimated weathering rate increased between 14.6 and 22.1% (mean 19.8%) during the period 2010 to 2050. This

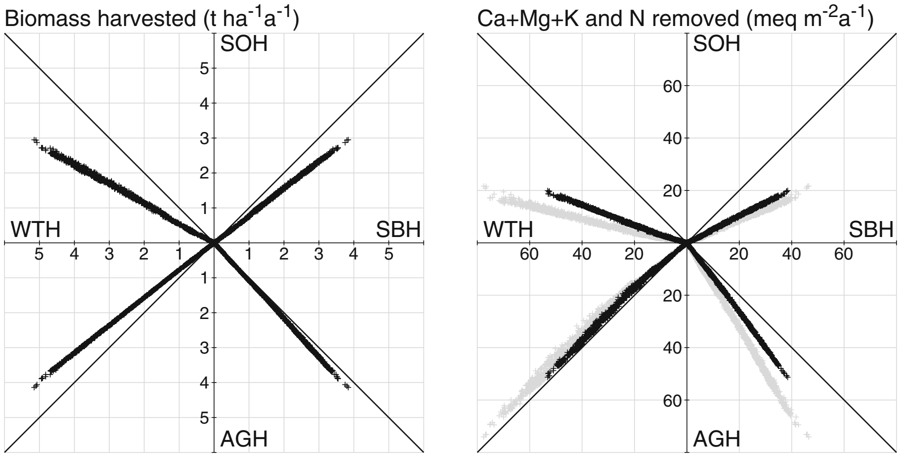

Fig. 4 ‘Quarrelations’ between the four SuC harvest scenarios (SOH, SBH, AGH and WTH) for biomass (*left*) and nutrient removals (*right, black* $Ca^{2+} + Mg^{2+} + K^{+}$, *grey* N) in the study catchments ($n = 1066$) in 2050 (see also Fig. 3)

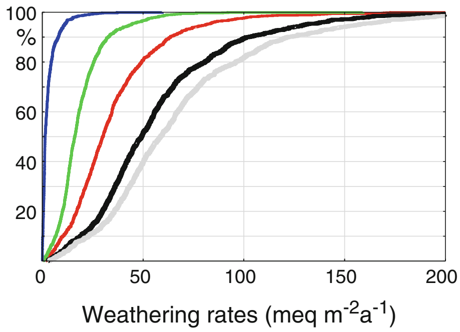

Fig. 5 Cumulative distribution functions of calibrated K^{+} (*blue*), Mg^{2+} (*green*), Ca^{2+} (*red*) and total base cation ($Bc = Ca^{2+} + Mg^{2+} + K^{+}$) weathering under current (*black*) and future (2050 A2) climate (*grey*) in the study catchments ($n = 1066$; *curves* described from *left* to *right*)

corresponded to a 3.6–7.3% (mean 5%) increase per °C increase in temperature, which is well within the range (up to 10% per °C) found in previous studies (Brady 1991; White and Blum 1995). The importance of temperature on mineral weathering rates has been debated; while laboratory studies show a strong dependence on temperature, field studies are often obscured by other environmental factors that co-vary with temperature (Kump et al. 2000). Nonetheless, several studies have reported field-based temperature dependence on natural weathering (Velbel 1993, White and Blum 1995). Mineral weathering ultimately depends on soil mineralogy; chemical kinetics dictate that the dissolution rate of minerals per unit surface area is temperature dependent (Sverdrup 1990). Moreover, the active mineral surface area is influenced by soil moisture and acidity (Sverdrup 2009). While many studies have suggested that soil fungi also play a key role in the mineralogical modification (Jongmans et al. 1997, Bonneville et al. 2009), it remains hotly debated (Sverdrup 2009). Site disturbance following harvesting can have a significant impact on soil Bc exchangeable pools, cation leaching (Huber et al. 2010) and soil moisture on a local scale. However, Finnish ‘good forestry practices’ limit the level of site disturbances (Äijälä et al. 2010). These short-term changes in soil moisture and acidity may influence Bc weathering rate; however, at the catchment scale (and over the forest rotation) these influences are probably minor. In the current study, the model formulation (Appendix B) only accounted for predicted changes in temperature. Other climate-related controls on weathering rates, such as soil moisture (Sverdrup 1990; White and Blum 1995) and freeze–thaw cycles, could potentially offset increases due to temperature; however, previous studies have shown that soil moisture (on an annual basis) does not change much under the A2 climate change scenario in Finland (Posch et al. 2008).

Influence of harvesting scenarios and climate on soil and water chemistry

The type of harvesting scenario determines the impact on future soil and water quality (Fig. 6), with

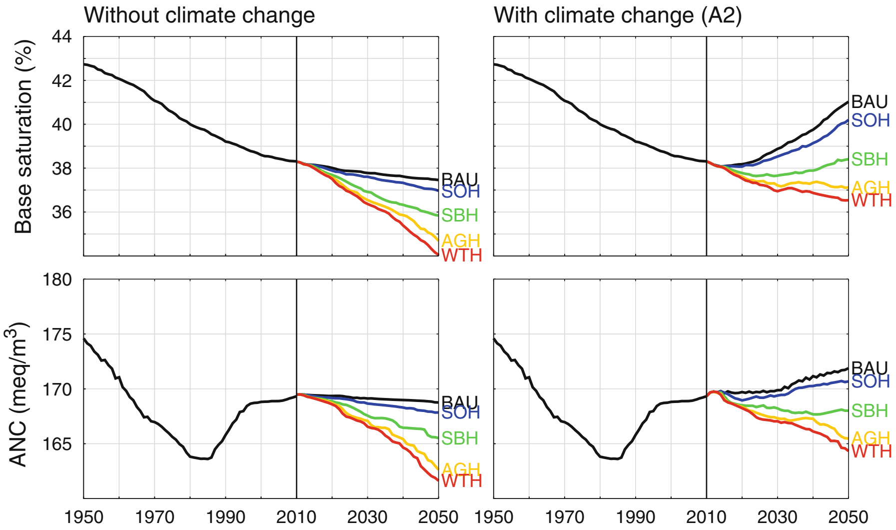

Fig. 6 Temporal development (1950–2050) of the median soil base saturation (*top*) and lake water ANC (*bottom*) for the study catchments ($n = 1066$), under constant climate (*left*) and the A2 climate scenario (*right*) for the five harvesting scenarios (BAU, SOH, SBH, AGH and WTH)

increasing element removal causing a greater decline in the chemical indicators (base saturation for catchment soils and ANC for water quality). The most intensive management scenarios, AGH and WTH, predicted a clear deterioration in soil and water quality. The decrease in atmospheric acid deposition since the 1980s has caused a significant recovery in lake ANC (Posch et al. 2008), which has also been documented by monitoring data (Vuorenmaa and Forsius 2008). Nonetheless, forest harvesting was predicted to cause a decline in both base saturation and ANC from 2010 onwards reversing recovery in ANC under the more intensive scenarios (Fig. 6). Climate change was predicted to significantly influence future chemistry; the key processes affected in the modelling are forest nutrient uptake and the weathering rate of Bc in soils. Climate change was predicted to have a positive effect on future soil and water quality owing to the fact that the estimated increase in weathering rates was higher than the increase in forest Bc uptake. Notably, climate change was predicted to result in recovery of both soil base saturation and lake water ANC under BAU and SOH. In contrast, the AGH and WTH scenarios were still predicted to decrease both soil base saturation and lake water ANC at the sites (Fig. 6, right).

Acid sensitive ecosystems are important indicators for ecosystem changes. The 5% most sensitive lake catchments ($n = 53$, with 1987 lake ANC <35 meq m^{-3}) showed a similar pattern in predicted future soil and water quality (Fig. 7). These lakes are located throughout the country, but more concentrated in the northern parts of Finland (see Fig. 1), and also have lower soil base saturation (Figs. 6, 7). The most significant difference in predicted future responses for these sensitive ecosystems was the response to climate change; limited recovery was predicted compared with the average conditions (median values in Fig. 6).

The SBH, AGH and WTH scenarios incorporated progressive removal of forest residue compartments. Further, it was assumed that 100% of these compartments were removed during harvesting. In practice, however, due to logistical and forestry guideline limitations, not all biomass is recovered, and thus the predicted impacts may be viewed as pessimistic. Moreover, current guidelines for ‘good forestry practice’ limit the harvesting of forest residues to more fertile sites. The sensitive lakes are, in general, associated with less fertile soils and, as such, are unlikely to be exploited for biomass removal. Moreover, newly issued forestry guidelines (Äijälä et al. 2010) recommend leaving 30% of the foliage on site ‘as evenly distributed as possible’ and (or) allowing harvested biomass to dry in situ before removing residues.

Sustainable forest management practices should ensure that the long-term uptake of Bc does not exceed weathering plus deposition. In the short term,

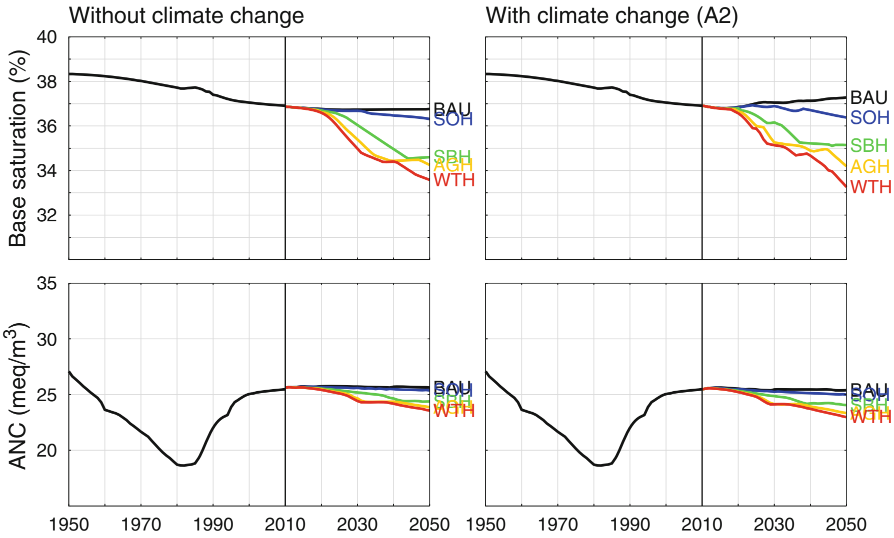

Fig. 7 Temporal development (1950–2050) of the median soil base saturation (*top*) and lake-water ANC (*bottom*) for the 53 most sensitive catchments (5% of all catchments), under constant climate (*left*) and the A2 climate scenario (*right*) for the five harvesting scenarios (BAU, SOH, SBH, AGH and WTH)

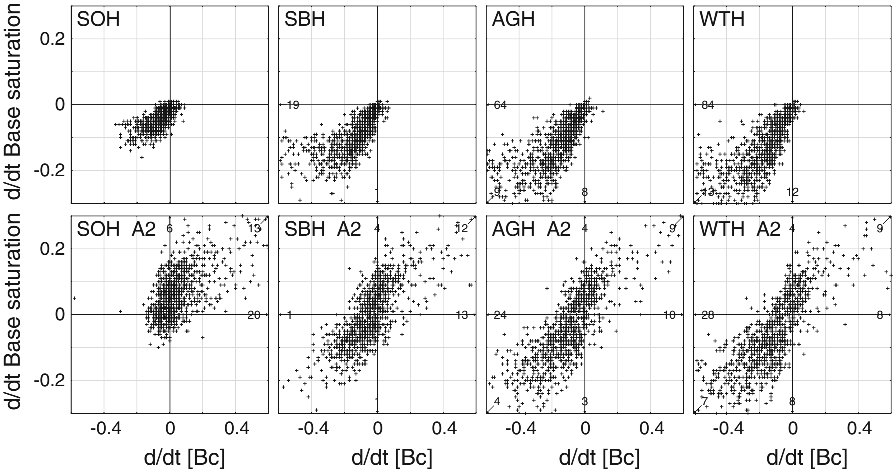

Fig. 8 Change from 2049 to 2050 in base cation ($Bc = Ca^{2+} + Mg^{2+} + K^{+}$) lake water concentration ($meq\ m^{-3}\ a^{-1}$) versus change in catchment soil base saturation ($\%\ a^{-1}$) for the SOH, SBH, AGH and WTH harvesting scenarios under constant climate (*top*) and the A2 climate change scenario (*bottom*) for each study catchment ($n = 1066$)

imbalances are accommodated by the exchangeable cation pool; however, in the long term they will lead to a decrease in soil base saturation. Sustainability in terms of soil conditions can thus be defined as a situation where no long-term reduction in the Bc pool (base saturation) is allowed. Under present climate, the majority of the study catchments showed a decrease in Bc concentration ($meq\ m^{-3}a^{-1}$) and in base saturation (% per year) at the end of the simulation period (2049–2050; Fig. 8). Moreover, the harvesting scenarios predicted an accelerated decline in Bc concentration and base saturation in 2050 (see also median values in Figs. 6 and 7). In contrast, under the A2 climate, soil base saturation increased in many of the study catchments thus increasing the proportion of catchments where a more intensive management is sustainable (almost a third of the sites for the WTH scenario). Furthermore, under the A2 climate both soil base saturation and Bc leaching increased simultaneously (see points in the first quadrant in Fig. 8) due to the increased supply of Bc by weathering owing to higher temperatures.

Processes that determine the quality of soil and water chemistry in forested ecosystems are impacted by changes in climate, atmospheric deposition and land-use practices. Changes in any of these major drivers can affect key processes, such as the accumulation of soil carbon (Powers et al. 2005), base cation fluxes (Joki-Heiskala et al. 2003) and leaching of N and acidity (Wright et al. 2006). Biomass growth and harvesting has a significant effect on the acid–base status of forest soils owing to the excess accumulation of Bc compared with anions during tree growth (e.g., see Olsson et al. 1993). Moreover, element concentrations in different tree compartments vary significantly, with much higher concentrations in needles, leaves and branches than in stem wood (Table 2). Accordingly, alternative forest harvesting strategies have different impacts on soil and water quality. Despite an increased growth under climate change, increased weathering rates compensated the increased removal of base cations in this study (Figs. 6, 7). Increased weathering under climate change is a key assumption in the present study, with a modelled 5% average increase per degree temperature.

Nutrient limitations under biomass harvesting

Nitrogen is generally a growth-limiting factor in boreal ecosystems (e.g., Tamm et al. 1999; Hyvönen et al. 2008). Therefore, long-term sources of N such as deposition need to be quantified for sustainable forest management. However, isotope tracer studies in temperate forests suggest that soils are the dominant sinks for N deposition, with only low uptake in tree woody biomass (Nadelhoffer et al. 2004). The uptake of organic N is also an important source (see Näsholm et al. 1998), which is (partially) the result of N fixation. DeLuca et al. 2002 have suggested that fixation can reach up to 3 kg N $ha^{-1}a^{-1}$ in boreal ecosystems. In contrast, harvesting of biomass permanently removes N from forest ecosystems; and there is also increasing evidence that other nutrients, such as K and phosphorus, become limiting under forest harvesting (Akselsson et al. 2008). In the current study, the intensive forest management scenarios resulted in both N and K deficiencies at 68 (AGH) and 104 (WTH) catchments (Fig. 9). In addition, deficiencies in individual elements were estimated at many more catchments (assuming N fixation at 3 kg N $ha^{-1}a^{-1}$). Fertilizer amendment (with NPK) during forest establishment is already a common management practice depending on site quality (and to some extent is incorporated into the MELA growth scenarios). Moreover, current guidelines for 'good forestry practices' recognise the potential for additional nutrient deficiencies and recommend woodash amendment depending on soil quality and harvesting intensity (Äijälä et al. 2010).

Under the A2 climate and WTH scenario, the increased growth and removal of N clearly exceeded

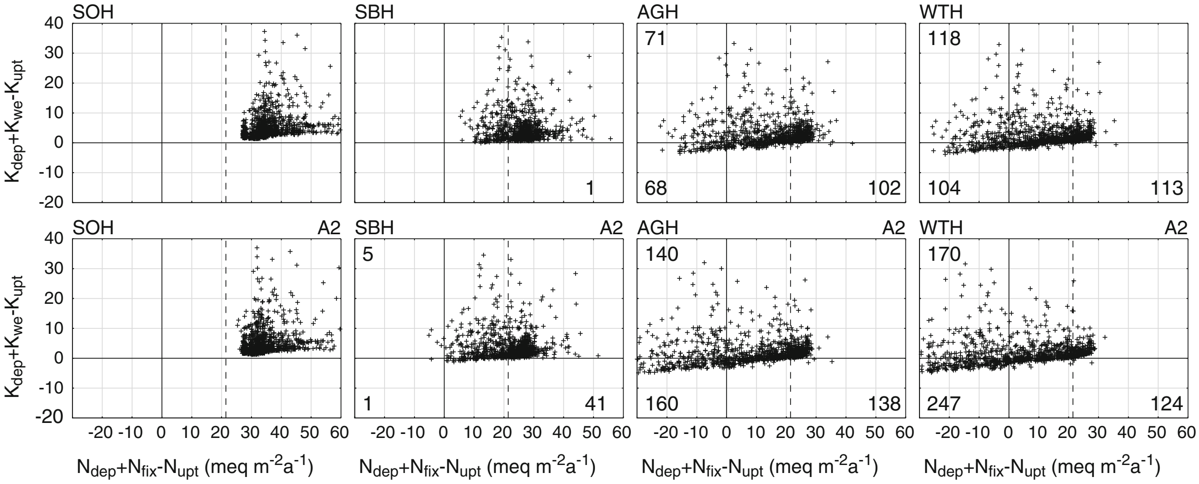

Fig. 9 Steady-state input–output budget of nitrogen (N_{dep} + N_{fix}–N_{upt}) against potassium (K_{dep} + K_{we}–K_{upt}) in the study catchments ($n = 1066$) for the four harvesting scenarios (SOH, SBH, AGH, WTH) under constant climate (*top*) and the A2 climate scenario (*bottom*) in 2050. The number of catchments with N limitations only, K limitations only and both N and K limitations is displayed in the respective quadrant. The *vertical dashed* line illustrates how many more catchments would have N limitations if no N fixation was assumed (N_{fix} = 3 kg N ha^{-1} a^{-1} [= 21.43 meq $m^{-2}a^{-1}$])

long-term N sources (deposition plus fixation) in more than 400 catchments (Fig. 9). In concert, more than a third of the catchments ($n = 371$) were K limited, of which two-thirds ($n = 247$) were also N limited under the WTH scenario. Nitrogen and K were deficit in a large number of the study catchments under the AGH and WTH scenarios, as they include removal of foliage, which is high in N and K (Table 2). Although additional K is supplied from the soil exchange complex, this would lead to depletion in soils in the long-term. Further, although N mineralization is likely to increase under a warmer climate (Wright 1998), the long-term supply is uncertain. To get a better insight into short-term nutrient dynamics more detailed process descriptions for nutrient cycling are required to evaluate their limitations under biomass harvesting and climate change. Nonetheless from an economic viewpoint, it would seem prudent to restrict the utilization of needles and leaves for biofuel and energy production, rather than to remove nutrients in biomass and later to return them as fertilizers.

Uncertainties

There are numerous uncertainties involved in any regional-scale model application. These can be roughly grouped into two categories: (a) uncertainties owing to model(s) structure (conceptualisation), and (b) uncertainties in the model parameters and input data. A general discussion on these topics in the context of a previous modelling study for Finland is presented in Aherne et al. (2008). An additional uncertainty in the present study concerns the modelling of forest growth and element uptake. Forest growth was not integrated into the dynamic modelling framework and net-removal alone determined element uptake. There was thus no feedback between availability of nutrients and the growth of forests, despite the fact that the more intensive management scenarios suggested an increased removal of these nutrients from the soil. Further, the interaction between climate change and N mineralization was neglected in this study; increasing temperature may stimulate plant growth by increasing N availability owing to enhanced N mineralization (e.g., Bonan and Van Cleve 1992). The lack of feedback between soil (N) status and forest growth is a limitation of this study; there are indications that re-growth following whole-tree harvest are depleted in nutrients compared with re-growth following stem only harvest (Thiffault et al. 2006, but see also Luiro et al. 2010), especially at sites with low fertility (Jacobson et al. 1996; 2000). Further, the current study used net rather than gross uptake and return. However, to some extent nutrient dynamics is incorporated into the forest harvesting scenarios from MELA, which is a management model based on observed yield from national forest inventories. Moreover, these scenarios define production goals, and management action would likely be triggered if these goals were jeopardised by, e.g., predicted nutrient deficiencies. Although the MELA model does not explicitly accommodate the influence of nutrient feedback on forest growth, the yield-based approach provides realistic harvesting scenarios, used by the Finnish forest authorities. Nonetheless, the harvesting scenarios were ‘idealized’ in the sense that it was assumed that 100% of the biomass assigned for removal was indeed removed. This is not the case in practice: guidelines for ‘good forestry practices’ in Finland limit the removal of stumps, roots and branches to more fertile sites (which may potentially limit biofuel production). Also technical constraints limit the removal of branch (and leaf) biomass to about 70% of the total. This must be borne in mind when interpreting the results of this study.

In the current study, the direct effect of CO_2 on forest growth under climate change was not considered as it is disputed whether increased atmospheric CO_2 concentration will enhance growth. Some studies indicate substantial effects (e.g., Hyvönen et al. 2007), whereas others doubt whether there is any effect at all, especially on mature trees (e.g., Körner et al. 2005). Moreover, potential drought effects on growth were neglected, since the amount of precipitation in Finland is generally not limiting, and is predicted to increase under the A2 climate scenario.

Conclusions

The potential impacts of climate change currently drive the search for alternatives to fossil fuel, and policy is set to substantially increase the use of forest biomass for biofuel and energy production. Long-term sustainable forest ecosystems require that the environmental impacts of changes in management are quantified, while simultaneously evaluating the impacts of other key drivers. In the current study the long-term impacts of forest biomass removal, climate and deposition on the soil and water chemistry was studied, using extensive datasets covering the whole of Finland. The use of a large random sample of catchments covering the whole of Finland allowed for a regionally representative assessment, in contrast to similar studies on individual sites. The biogeochemical modelling framework presented in this study can be used to develop both climate-optimal and environmentally sustainable scenarios for increased use of renewable energy sources from forested ecosystems.

The harvesting scenarios resulted in very different levels of biomass and element removal. The SBH scenario increased biomass harvest by a third (compared with SOH), but doubled the removal of Bc and N. In contrast, the AGH scenario increased Bc and N removal by another 50%, with little gain in biomass. Further, the WTH scenario doubled biomass removal, while the removal of base cations from the catchment soils tripled and removed N increased fourfold.

Hydro-geochemical modelling predicted that the harvesting scenarios clearly have different impacts on future conditions in soils and lakes, with increasing element removal causing a consistent decline in these environmental impact indicators. The most intensive management scenarios, AGH and WTH, were predicted to lead to a significant deterioration in soil and water quality, if not compensated by management measures. Surprisingly, climate change had a positive impact on future conditions due to the increased supply of Bc by weathering, in excess of increased uptake. Nonetheless, even when considering this likely future ‘benefit’ of climate change, the AGH and WTH scenarios resulted in a decrease in soil and water quality. Thus it seems prudent not to remove foliage (needles and leaves) owing to their high element concentrations and low mass (and thus low economic gain). In addition to Bc, increased removal of biomass will most likely require additional inputs of N and K (and possibly other nutrients) to ensure sustained forest growth. Thus, without additional nutrient inputs, only harvesting of above-ground woody biomass (SOH and SBH scenarios) would ensure long-term sustainable forests.

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Appendix A: Forest growth under climate change

Forest growth under climate change is obtained from the reference growth G_0 as:

$$G = G_0 \cdot f_{temp} \cdot f_{resp} \tag{1}$$

i.e. the modified growth G is obtained by multiplying the (measured) reference growth with a temperature factor f_{temp} and a respiration factor f_{resp}. Both the temperature and the respiration factor are taken from the C-Fix model (Veroustraete et al. 2002). The temperature effect on growth is modelled according to:

$$f_{temp} = \frac{g(T)}{g(T_0)} \quad \text{with} \quad g(T) = \frac{\exp\left(C_1 - \frac{\Delta H_a}{RT}\right)}{1 + \exp\left(\frac{\Delta S T - \Delta H_d}{RT}\right)} \tag{2}$$

where ΔH_a is the activation energy (52 750 J mol^{-1}), ΔH_d the deactivation energy (211 000 J mol^{-1}), ΔS the entropy of the denaturation equilibrium of CO_2 (704.98 J K^{-1}mol^{-1}), $C_1 = 21.77$, and $R = 8.314$ J K^{-1}mol^{-1} the universal gas constant. T is the temperature and T_0 the reference temperature at which G_0 is measured (both in K). The respiration factor is modelled as the ratio of autotrophic respiration at different temperatures:

$$f_{resp} = \frac{1 - A_d(\theta)}{1 - A_d(\theta_0)} \quad \text{with} \quad A_d(\theta) = (7.825 - 1.145 \cdot \theta)/100 \tag{3}$$

where θ and θ_0 are the temperatures in degrees Celsius ($\theta = T$–273.15).

Appendix B: Base cation weathering under climate change

The temperature-dependence of base cation ($Ca^{2+} + Mg^{2+} + K^{+}$) weathering rates is modelled as:

$$Bc_w(T) = Bc_w(T_0) \cdot A(T_0, T) \tag{4}$$

Since the function A describes the temperature-dependence of the underlying chemical reactions, it is assumed to have the form of an Arrhenius-factor (T and T_0 in Kelvin):

$$A(T_0, T) = \exp\left(\frac{\Delta H}{R}\left(\frac{1}{T_0} - \frac{1}{T}\right)\right) \tag{5}$$

For the activation energy divided by R we used $\Delta H/R = 3600$ K. This is an average of values for different mineral groups given in Sverdrup and Warfvinge (1993), which is also used as a default value in European critical load calculations (UBA 2004). For $T_0 = 274.2$ K (the median temperature at the 1066 catchments; see Table 1) and $T = T_0 + 3.8$ (typical increase in Finland by 2050 under A2 climate change scenario; see Fig. 1) one obtains $A = 1.197$, i.e. an increase in weathering by about 20%. The sensitivity to the value of $\Delta H/R$ is small: for $\Delta H/R = 3200$ and $\Delta H/R = 4000$ (a more than 10% variation around 3600) one obtains $A = 1.173$ and 1.221, respectively, i.e. less than 3% change.

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