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Modelling future NO_3 leaching from an upland headwater catchment in SW Norway using the MAGIC model: II. Simulation of future nitrate leaching given scenarios of climate change and nitrogen deposition

Anne Merete S. Sjøeng, Øyvind Kaste and Richard F. Wright

ABSTRACT

Future nitrate (NO_3) leaching to surface water at an upland heathland catchment, Øygardsbekken, southwestern Norway, was simulated using the MAGIC model with monthly time steps. Øygardsbekken has high nitrogen (N) load and exhibits seasonally elevated NO_3 leaching. Future estimates for temperature, precipitation and N deposition were implemented. The climate scenarios were based on dynamically downscaled data from the Rossby Centre Regional Climate model (RCAO) driven by two scenarios of greenhouse gas emissions, A2 and B2, and run with two global climate models, HadAM3 and ECHAM4/OPYC3 from the Hadley Center and Max Planck Institute, respectively. Estimates of future rates of nitrogen (N) and carbon (C) processes in the catchment were based on the downscaled temperature scenarios and two different storylines, one assuming changes only in soil processes (mineralisation N, decomposition C, plant N uptake, N immobilisation, litterfall N) due to future warming and N deposition (SLsoil), and the other assuming changes in both vegetation (plant N uptake and litterfall C/N ratio) and soil processes (SLsoil + veg). Compared to the present, MAGIC simulated higher future NO_3 leaching for both storylines with much higher rates for SLsoil. The results suggest that differences between the two storylines were larger than differences between the different scenarios within each storyline. For the scenarios with the highest future leaching rates the pronounced seasonal NO_3 pattern levelled out, while for the scenarios with moderate projected NO_3 leaching the seasonal pattern prevailed but was skewed towards highest leaching during spring rather than in winter as at present.

Key words | climate change, heathland, modelling, nitrogen deposition, Norway, water

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INTRODUCTION

Global air temperatures increased by 0.7°C during the 20th century and are projected to increase by between 1.1–6.4°C during the 21st century (IPCC 2007). These increases may be accompanied by changes in the seasonal precipitation pattern and evapotranspiration. Large impacts are expected on high-altitude upland ecosystems (Beniston *et al.* 1997) because these are considered to be particularly sensitive due to the rapid change in climate with altitude and species' close adaptations to the cold environment (Körner 1999). For Norway regional climate models project 2.5–3.5°C increase in annual mean temperature from 1961–1990 to 2071–2100 under several scenarios of greenhouse gas emissions (REGCLIM 2005). The direct and indirect effects of the potential increase in temperature, precipitation and evapotranspiration on ecosystems are likely to be complex and highly variable in time and space. One consequence of

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climate change in upland ecosystems in Norway may be increased leaching of nitrogen (N) from soil to runoff due to increased mineralization of soil organic matter, as shown by the CLIMEX project (van Breemen *et al.* 1998; Wright 1998). The N released can exacerbate acidification of surface waters and supply nutrients to coastal marine ecosystems. Simulation models offer a valuable complement to experimental studies by allowing effects of different N deposition and climate scenarios to be assessed over long time scales and by testing hypotheses about the mechanisms of interaction of possible causal factors.

Temperature, together with moisture, are key factors regulating biogeochemical processes such as litter decomposition (Kirschbaum 1995), respiration (Howard & Howard 1993; Leiros *et al.* 1999), nutrient mineralization and immobilization (Schmidt *et al.* 1999; Jonasson *et al.* 2004). Warming may lead to an extension of the growing season in upland ecosystems in Norway. In general this can have positive effects on plant growth (Myneni *et al.* 1997; Menzel & Fabian 1999) which in turn can cause an increase in litter production.

The main purpose of the present study is to explore the effects of changed temperature, precipitation and N deposition on future long-term and seasonal nitrate (NO_3) concentrations and flux in streamwater given two scenarios of greenhouse gas emissions (A2 and B2 of the UN Intergovernmental Panel on Climate Change (IPCC)) run with two general circulation models (GCMs) (Hadley Center and Max Plank Institute). The MAGIC 7.77 model (Model for Acidification of Groundwater In Catchments) (Cosby *et al.* 1985*a*,*b*, 2001) that has been calibrated to the 12-year data series at Øygardsbekken, southwestern Norway (Sjøeng *et al.* 2009) was used. Øygardsbekken is a small headwater catchment draining upland heathlands and receives high amounts of precipitation and N deposition. NO_3 concentrations and fluxes for the calibration period (1993–2004) were compared with those obtained for the corresponding 12 years during the scenario period (2085–2097) under two different assumptions (storylines) for future soil and vegetation processes. The relative impacts of the different climate scenarios on the long-term change in the soil carbon (C) pool is also considered.

MATERIALS AND METHODS

Scenario estimates

Climate scenarios

Climate scenarios for the Øygard catchment were based on dynamically downscaled temperature, precipitation and evaporation data from the Rossby Centre Regional Climate Model (RCAO) provided by the EU European project PRUDENCE (Christensen & Christensen 2007, http://prudence.dmi.dk). The RCAO simulations were based on two global climate models (GCM), HadAM3 (Hadley, HC) and ECHAM4/OPYC3 (Max Planck Institute, MPI), run with two scenarios of greenhouse gas (GHG) emissions, A2 and B2 (IPCC 2001). Both models use 1961–1990 as the control period and 2071–2100 as the scenario period. The following abbreviations will be used in the following: HCA2 and HCB2 (Hadley scenarios), and MPIA2 and MPIB2 (Max Planck scenarios). The downscaled data from RCAO were allocated to a $50 \times 50\,km^2$ grid chosen as the most representative, based on average elevation of the grid cell and projections of annual precipitation, evaporation and average temperatures for the control period.

The preparation of climate scenarios for the MAGIC model was based on a monthly "delta" approach, i.e. the difference between monthly mean air temperature, precipitation and evaporation for the scenario period (2071–2100) and the corresponding monthly means for the control period (1961–1990). The delta values expressed as °C (air temperature) and % $month^{-1}$ (precipitation, evaporation) were then projected on monthly air temperature, precipitation and runoff time series. Scenario estimates for runoff were based on the % change in precipitation less the absolute delta values for evaporation. The absolute delta values for evaporation were used because no field measurements were available. Moreover the associated error is likely to be small since evaporation most likely is small compared to precipitation and runoff from this wet, upland site.

The mean delta value for each month from the control (1961–1990) to the scenario period (2071–2100) was assumed to represent the change from mid-control period, year 1975, to mid-scenario period, year 2085. According to these estimates, a gradual annual linear increase in the monthly rates starting at the present (i.e. 2011) and ending in 2084 was projected. From 2085 onwards maximum delta values were projected on present data, either on the measured time series (runoff, temperature) or estimated monthly mean rates (C and N processes, N deposition). Thus maximum monthly delta values from the different scenatios were used for each year during the scenario period 2085–2100.

N deposition, runoff and temperature

For future dissolved inorganic N (DIN) deposition there was assumed no change in NO_3 and NH_4 concentrations in precipitation with time after the year 2011. This assumption is based on a study by Hole *et al.* (2008) in which seven sites in southern Norway were investigated for relations between N wet deposition and various climate indices. They found that variations in N deposition were closely related to variations in precipitation. Moreover, results from a modelling study by Hole & Engardt (2008) point towards a substantial increase in historic N deposition over western Norway during the 1980s as a consequence of increasing precipitation. The deposition input file from calibration (1990–2004) was thus extended with mean time series data during 2005–2010, with % change for precipitation linearly distributed on the mean time series from 2011 onwards until 2084, and mean time series with maximum % change for 2085–2100. Accounting for the difference in N deposition levels between the control period (1961–1990) (i.e. year 1975 = mid-period) and year 2011 (Posch *et al.* 2003; Schöpp *et al.* 2003) the annual precipitation increase for 75 instead of 110 years (i.e. 75/110 of annual change in precipitation) was used.

In preparing the runoff file the runoff time series during 1990–2004 (the same as during calibration) and monthly means (1993–2004) for each of the years during 2005–2010 were used. For the years 2085–2096 monthly runoff time series were changed according to % change (scenario relative to the control period) in precipitation less the absolute change in evapotranspiration. The monthly time series for runoff was used to produce year-to-year variations in the runoff input file during the scenario period according to model calibration (Sjøeng *et al.* 2009). For the period 2011–2085, monthly mean (1993–2004) runoff was used with a gradual linear increase according to the % change in precipitation less the evapotranspiration. For the remainder of the scenario period (2097–2100) monthly mean data from the time series with the maximum % change in precipitation less evapotranspiration was used (Figure 1).

The mean monthly delta values for temperature from each of the scenarios were added to daily air temperature data and aggregated to mean monthly air temperature. These new air temperature time series served as the basis for calculation of future estimates of soil temperature using the sine-based equation from model calibration (Table 1). Then soilGDD was calculated for each soil temperature time series for the different scenarios.

C and N processes

During calibration rates for C and N processes were set as functions of temperature (Table 1), thus enabling future process rates to be easily computed using the projected future scenarios for temperature in the equations in Table 1. The calibrated monthly mean rates for C and N processes were used each year to calculate future rates (according to the set-up during model calibration). Present monthly mean temperature data were replaced by the future monthly temperature estimates from the scenario projections in the calculations for present C and N processes (Table 1).

Physical estimates for the lake compartment

A future warmer climate will probably change the thermal regime in the small lake Hedlevatn, situated in the lower part of the catchment. This will likely affect the damping of the seasonal pattern of NO_3 concentrations in runoff from the terrestrial parts of the catchment.

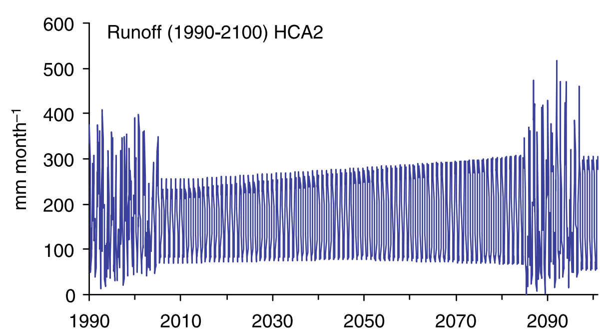

Figure 1 | Estimated runoff (mm month^{-1}) during 1990–2100, examplified for the HCA2 scenario. Measured short-term variations in runoff (1990–2004) were superimposed on a long-term linear trend, according to scenario estimates for precipitation less evaporation. See text for details.

Table 1 | Equations used to estimate different parameters during model calibration (Sjøeng *et al.* 2009)

Parameter	Equation	References and comments
Daily soil temperature (T_{soil})	$T_{soil} = T_{air} - 3\sin\left(\frac{3}{2}\pi\frac{\text{day no.}}{365}\right)$	Whitehead *et al.* (1998); T_{air} = air temperature
Growing degree days (GDD)	$\text{GDD} = \sum(\text{daily mean temperature} - \text{threshold temperature})$	Threshold temperature = 5°C; Gudem & Hovland (1999)
Monthly vegetation N uptake (N_{upt})	$N_{upt} = 0.4471 \times \text{monthly soilGDD}$	soilGDD; soil temperature was used monthly soilGDD = ΣsoilGDD (for a month); Sjøeng *et al.* (2009)
Decomposition C; equation for soil respiration (R)	$R = R_{10}\exp\left(E_0\left(\frac{1}{283.85-T_0} - \frac{1}{T-T_0}\right)\right) = 2459.6\exp\left(308.56\left(\frac{1}{56.02} - \frac{1}{T-227.13}\right)\right)$	Equation and constants E_0 and T_0 from Lloyd & Taylor (1994). R_{10} was calibrated (Sjøeng *et al.* 2009)
Mineralization N (N_{min})	$N_{min} = 96.5\exp\left(308.56\left(\frac{1}{56.02} - \frac{1}{T-227.13}\right)\right)$	New $R_{10} = R_{10}(R)$/measured soil C/N ratio = 2459.6/25.5 = 96.5; Sjøeng *et al.* (2009)
% potential retention (PR) (estimate for gross immobilization)	$\text{PR} = 88.2\exp\left(100\left(\frac{1}{56.02} - \frac{1}{T-227.13}\right)\right)$	New constants for E_0 and T_0 were calibrated Sjøeng *et al.* (2009)

Future changes in lake thermal regime have been estimated for Lake Kolbotnvatn, a small lake in south-eastern Norway (Tjomsland & Rohrlack 2008). The estimates were based on the HCA2 climate scenario (temperature, wind, radiation, relative humidity) from the Rossby Centre using the CE-Qual-W2 model (http://www.ce.pdx.edu/w2) (Tjomsland & Rohrlack 2008). These estimates were used for Lake Hedlevatn. The present-day mean annual temperature at Lake Kolbotnvatn is approximately 0.3°C higher (slightly higher mean air summer and lower winter temperatures) compared to Øygardsbekken.

Simulations from the model CE-Qual-W2 indicate there will be no ice formation on the lake surface during winters 2071–2100; thus the lake will have full mixing throughout the winter (Tjomsland & Rohrlack 2008). Higher air temperatures in spring will cause the summer stagnation to start somewhat earlier (approximately 14 d) and prevail longer (approximately one month) during autumn. These changes were incorporated into the simulations at Øygardsbekken by changing the future monthly % epilimnion volume (Figure 2). This seasonal pattern was assumed for all years and for each of the different scenarios from year 2005 until 2100, while the present-day seasonal stratification pattern was used during the calibration period (1993–2004) (Figure 2). The MAGIC model proved to be quite sensitive to changes in the seasonal stratification pattern during calibration; when epilimnion volume was changed to 100% (i.e. complete mixing of the lake) during each month and for

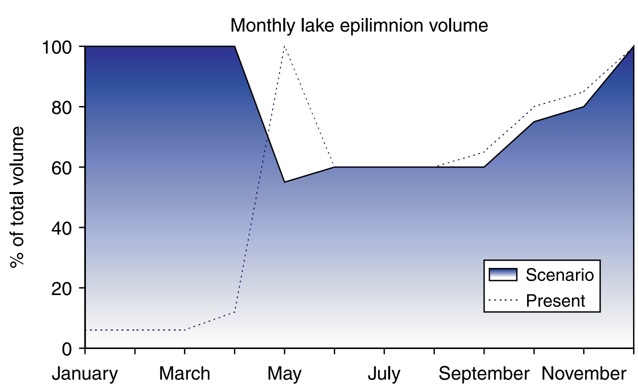

Figure 2 | Calibrated (dotted line) and estimated (filled area) future monthly lake epilimnion volume. Future estimates were based on simulation with the CE-Qual-W2 model run with the HCA2 scenarios for temperature, wind, radiation and relative humidity for Lake Kolbotnvatn (Tjomsland & Rohrlack 2008). 100% epilimnion volume denotes complete mixing of the lake.

every year, the year-to-year variation in simulated surface water NO_3 concentrations was lost (Sjøeng *et al.* 2009).

Two different storylines for the future simulations and assumptions

Two different estimates (henceforth called storylines) for future rates of C and N processes were used and run with each of the four different climate scenarios (HCA2, HCB2, MPIA2, MPIB2), providing a total of eight different N parameter files and MAGIC trials.

Common to all eight trials were: (1) N deposition estimated from the increase in precipitation, (2) temperature-based estimates (i.e. soil temperature) for decomposition C and thus N mineralization and (3) % potential retention unchanged (estimate of short-term immobilisation N in MAGIC). Decomposition rates were assumed to accelerate according to the predicted temperature increase, but for simplicity a possible increase or decrease in the decomposition rate due to changes in litter quality or quantity was not accounted for. Firstly, this is due to a lack of field measurements of decomposition rate. Secondly, neither storyline assumes increased biomass production in response to increased temperature and N deposition. This is because the Øygardsbekken catchment is an upland ecosystem that does not have a large forest stand but vegetation consists mainly of *Calluna* heather, and thus the net change in plant storage of either C or N for the next 90 years is likely to be small. Thirdly, there is still no clear understanding of how different decomposition substrates will respond to increasing temperatures (Davidson & Janssens 2006), in part because of methodological difficulties associated with quantifying substrate availability.

Leaching of organic C and N were assumed constant at 200 mmol C m^{-2} yr^{-1} and 13 mmol N m^{-2} yr^{-1}, respectively (calculated annual mean 1993–2004). Furthermore several other assumptions were made as justified during model calibration (Sjøeng *et al.* 2009): (1) 100% nitrification of NH_4 to NO_3, (2) no denitrification and (3) annual plant N uptake equal to annual litterfall N.

The two storylines explored in this study used the same description and quantification of soil N and C processes; they differed in the quantification of plant processes (Table 2). With the first storyline (SLsoil) no change in vegetation was assumed (i.e. no change in annual biomass

Table 2 | Descriptions of the two storylines (SLsoil and SLsoil + veg) run with each of the four climate scenarios using MAGIC

Scenario estimated rates (for input files)	Reference	SLsoil	SLsoil + veg
N deposition based on delta in precipitation	Estimated this study	×	×
Temperature-dependent decomposition/mineralization based on future estimated soilGDD	Equation from Lloyd & Taylor (1994)	×	×
Plant N uptake based on present estimated soilGDD	Estimation from Sjøeng *et al.* (2009)	×	
Plant N uptake based on future estimated soilGDD	Based on relationship between present soilGDD and plant N uptake (Equation (1))		×
Litter fall N Annual rate equals present plant N uptake	Monthly distribution from Cormack & Gimingham (1964)	×	
Litter fall N Annual rate equals future plant N uptake	Monthly distribution from Cormack & Gimingham (1964)		×
C/N litter assumed constant = 45	Estimation from Sjøeng *et al.* (2009)	×	
C/N litter assumed to decrease from 45 to 35	Estimated this study		×

production or vegetation N content) due to the projected increase in temperature and precipitation; that is, present mean monthly estimates for plant N uptake, litterfall N and litter C/N ratio (i.e. the same N file as during calibration) was used each year. For the second storyline (SLsoil + veg), temperature-induced changes were assumed for plant N uptake rates according to scenario-estimated soilGDD and with equal annual rates for litterfall N but monthly distribution according to Cormack & Gimingham (1964). For this storyline a gradually declining C/N ratio of litter was assumed (used in calculating litterfall C) of about 20% from 45 in 2011 to 35 in 2085 and onwards. The rationale here was that the system becomes more N-rich due to greater N sequestration (Pitcairn *et al.* 2001; Pilkington *et al.* 2005), but with no change in annual biomass production. The values for litter C/N ratios fall within the range of present litter C/N ratios of *Calluna Vulgaris* (L.) as measured in the UK (i.e. Pilkington *et al.* 2005).

RESULTS

Scenarios

Climate scenarios, N deposition and discharge

The four climate scenarios projected a mean annual increase in temperature of 2.1–3.7°C in the scenario period 2071–2100 relative to the control period 1961–1990 (Table 3). The MPI model with the A2 scenario gave the largest increase, while the HC model with the B2 scenario gave the lowest. All scenarios projected an increase during all seasons, although the MPI scenarios showed most pronounced heating during winter and the HC scenarios during late summer and early autumn. The two models gave nearly the same temperature for the control period 1961–1990.

The models projected an annual increase in precipitation of 8–33%, with slightly larger increases for the A2 relative to the B2 scenario (Table 3) and with a threefold increase for MPI relative to HC. The MPI scenarios projected higher precipitation amounts during winter, spring and autumn, and unchanged or slightly reduced amounts during summer. According to the HC scenarios, the summers and early autumns will be dryer and winters wetter than at present.

The increase in discharge (Q) is expected to be slightly less than the increase in actual precipitation, as the projected heating of the systems will increase the evapotranspiration by 12–24% annually. On an annual basis the projected changes in Q were comparable with the increase in precipitation (Table 3). The projections suggested that future discharge will be higher in autumn and winter, but lower in the spring and summer, with a summer minimum

Table 3 | Mean delta for each month and annually from the control (1961–1990) to scenario period (2071–2100) for air temperature (°C), evapotranspiration (mm), precipitation and discharge (%). Delta for discharge was calculated as difference between %precipitation and evapotranspiration

	Temperature (°C)				Evapotranspiration (mm)				Precipitation (%)				Discharge (%)			
Mo.	HCA2	HCB2	MPIA2	MPIB2	HCA2	HCB2	MPIA2	MPIB2	HCA2	HCB2	MPIA2	MPIB2	HCA2	HCB2	MPIA2	MPIB2
1	3.0	2.1	4.2	3.2	7	6	2	2	33	14	55	38	29	11	54	37
2	2.4	1.5	4.7	3.6	7	3	7	6	24	18	53	58	20	16	50	55
3	2.6	1.8	4.8	3.8	8	6	5	4	16	−4	69	34	12	−7	67	31
4	3.1	2.2	3.7	2.9	7	5	5	5	2	5	63	34	−3	2	59	30
5	2.9	2.1	2.7	2.2	8	4	4	4	23	9	33	27	12	3	28	22
6	2.9	1.5	2.7	2.4	13	5	2	3	−24	−5	7	3	−39	−11	5	−1
7	2.8	1.7	3.1	2.7	13	8	3	5	2	−10	−9	−3	−14	−19	−13	−10
8	3.2	2.4	3.7	2.9	12	9	4	3	−37	−32	−12	−4	−48	−41	−15	−6
9	3.9	2.5	3.7	2.8	8	7	0	1	−33	4	−2	1	−39	−2	−2	0
10	3.3	2.5	4.2	3.1	7	7	2	3	20	17	40	38	17	14	39	36
11	3.6	2.6	3.8	3.2	4	4	6	6	31	38	31	26	30	36	28	24
12	3.2	2.3	3.7	3.0	4	5	7	6	33	27	47	38	31	25	44	36
Year	3.1	2.1	3.7	3.0	98	68	45	49	12	10	34	28	9	8	35	27

in the range of 54–75 mm. These future projections suggested that moisture will probably not limit soil or vegetation processes for this ecosystem.

The future change in DIN deposition was due to changes in precipitation amount. The annual delta values for DIN deposition ranged from 5–23% with a threefold increase for MPI projections compared to HCs (Table 4), mainly caused by the higher rates during winter and spring (Figure 3). Due to slightly higher present-day (mean 1993–2004) N deposition rates for NO_3 compared to NH_4, the relative contribution of NO_3 to DIN deposition was higher (data not shown).

C and N processes

The two storylines gave large differences in simulated rates for microbial N immobilization. SLsoil was much higher than SLsoil + veg due to the lower rates for plant N uptake but comparable rates for mineralization that cause higher availability of N in soil solution (Table 4). Annual N immobilization rates for SLsoil were within the range of 750–990 mmol N m^{-2} yr^{-1} compared to 510–550 mmol N m^{-2} yr^{-1} for SLsoil + veg, with the highest relative increase for MPIA2 of 115% (SLsoil) and less than 20% increase for the scenarios of SLsoil + veg. Although the relative increase in immobilization for the SLsoil scenarios were large compared to the present, in magnitude gross mineralization rates were larger, ranging from 1,180–1,470 and 1,130–1,280 mmol N m^{-2} yr^{-1} for SLsoil and SLsoil + veg, respectively, with corresponding net mineralization rates of 430–480 and 610–740 mmol N m^{-2} yr^{-1}. For SLsoil + veg the largest relative increase in N rates compared to the present was for plant N uptake (and thus litterfall N) that ranged from 690–850 mmol N m^{-2} yr^{-1}. Thus the relative magnitude of the various N processes in the two storylines was (1) storyline SLsoil + veg: gross N mineralization > plant N uptake > immobilization N and (2) storyline SLsoil: gross N mineralization > immobilization N > plant N uptake.

Estimates of N processes were based directly on soil temperature (or soilGDD for plant N uptake and litter fall N), and thus their mean seasonal patterns for the respective scenarios (Figure 4) reflected the projected changes in temperature (Table 3). Among the different scenarios HCB2 had the smallest projected temperature increase, especially during spring and summer. Estimated rates of N processes for HCB2 for both storylines differed from the others with only minor increased or slightly reduced rates compared to

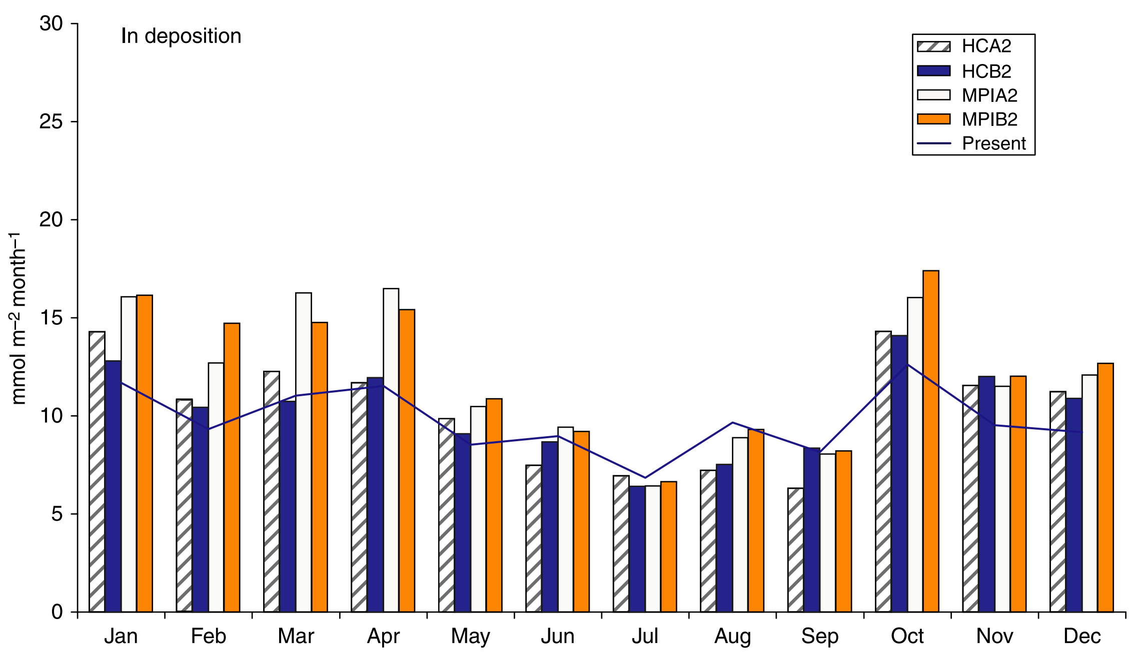

Figure 3 | Monthly present (solid line) mean (1993–2004) and estimated mean (2085–2096) DIN deposition (mmol m^{-2} $month^{-1}$) for each of the four climate scenarios (bars). Striped, black, white and grey bars represent HCA2, HCB2, MPIA2 and MPIB2 scenarios, respectively.

Table 4 | Comparison of annual mean N deposition and C and N processes (mmol m^{-2} yr^{-1}) and pools (mmol m^{-2}) between estimates from model calibration (present, 1993–2004) and the scenario (2085–2096) period for the two storylines, with calculated percent change (delta). Rates for immobilization, mineralization and organic C and N pools are taken from MAGIC simulations (i.e. model output) while the others are calculated (i.e. model input)

	Rates					Delta (%)			
Drivers	Present	HCA2	HCB2	MPIA2	MPIB2	HCA2	HCB2	MPIA2	MPIB2
NO_3 deposition	62	66	65	76	74	7	6	24	20
NH_4 deposition	56	58	58	68	66	5	4	23	19
IN deposition	117	124	123	144	140	6	5	23	19
SoilGDD	1,070	1,800	1,540	1,900	1,700	68	44	77	59
Storyline SLsoil									
Processes									
Plant N uptake	480	480	480	480	480	0	0	0	0
Immobilization N	460	920	750	990	880	100	63	115	91
Litter fall N	480	480	480	480	480	0	0	0	0
Litter fall C	21,600	21,600	21,600	21,600	21,600	0	0	0	0
Mineralization N (gross)	840	1,370	1,180	1,470	1,330	63	41	75	58
Decomposition C	21,400	28,800	26,000	30,100	28,100	35	21	41	31
Organic C pool	2,590	2,280	2,400	2,220	2,310	−12	−7	−14	−11
Organic N pool	102	109	109	109	109	7	7	7	7
NO_3 leaching	11.0	39.8	31.6	48.7	37.8	263	188	344	245
Storyline SLsoil + veg									
Processes									
Plant N uptake	480	800	690	850	760	68	44	77	59
Immobilization N	460	510	520	540	550	10	12	16	18
Litter fall N	480	800	690	850	760	68	44	77	59
Litter fall C	21,600	28,200	24,100	29,600	26,700	31	12	37	24
Mineralization N (gross)	840	1,220	1,130	1,280	1,210	45	35	52	44
Decomposition C	21,400	28,800	26,000	30,100	28,100	35	21	41	31
Organic C pool	2,590	2,605	2,540	2,620	2,565	1	−2	1	−1
Organic N pool	102	111	110	111	111	9	8	9	9
NO_3 leaching	11.0	15.7	20.3	15.8	17.8	43	85	44	62

the present for mineralization and immobilization. For the other scenarios in SLsoil + veg the monthly relative increases were consistent and showed only minor changes in the seasonal pattern for the N processes compared to present. The main difference between the future seasonal distribution of the different N processes of SLsoil and SLsoil + veg was the dramatic change in the seasonal pattern of microbial immobilization for SLsoil compared to the present with large increased rates during summer and autumn. Furthermore a slightly higher seasonal mineralization rate was obtained for SLsoil compared SLsoil + veg (Figure 4).

Model simulations

Long- and short-term changes

In general the differences among the scenarios for long-term changes in annual NO_3 leaching, soil organic C pool and the soil C/N ratios were larger for SLsoil compared to SLsoil + veg, although the largest differences were between the two storylines themselves (Figure 5). Soil organic N pools increased by 7% and 8–9% for all scenarios of SLsoil and SLsoil + veg, respectively (Table 4). Furthermore, for SLsoil the soil organic C pool was reduced by 7–14% compared to only minor changes for the scenarios in

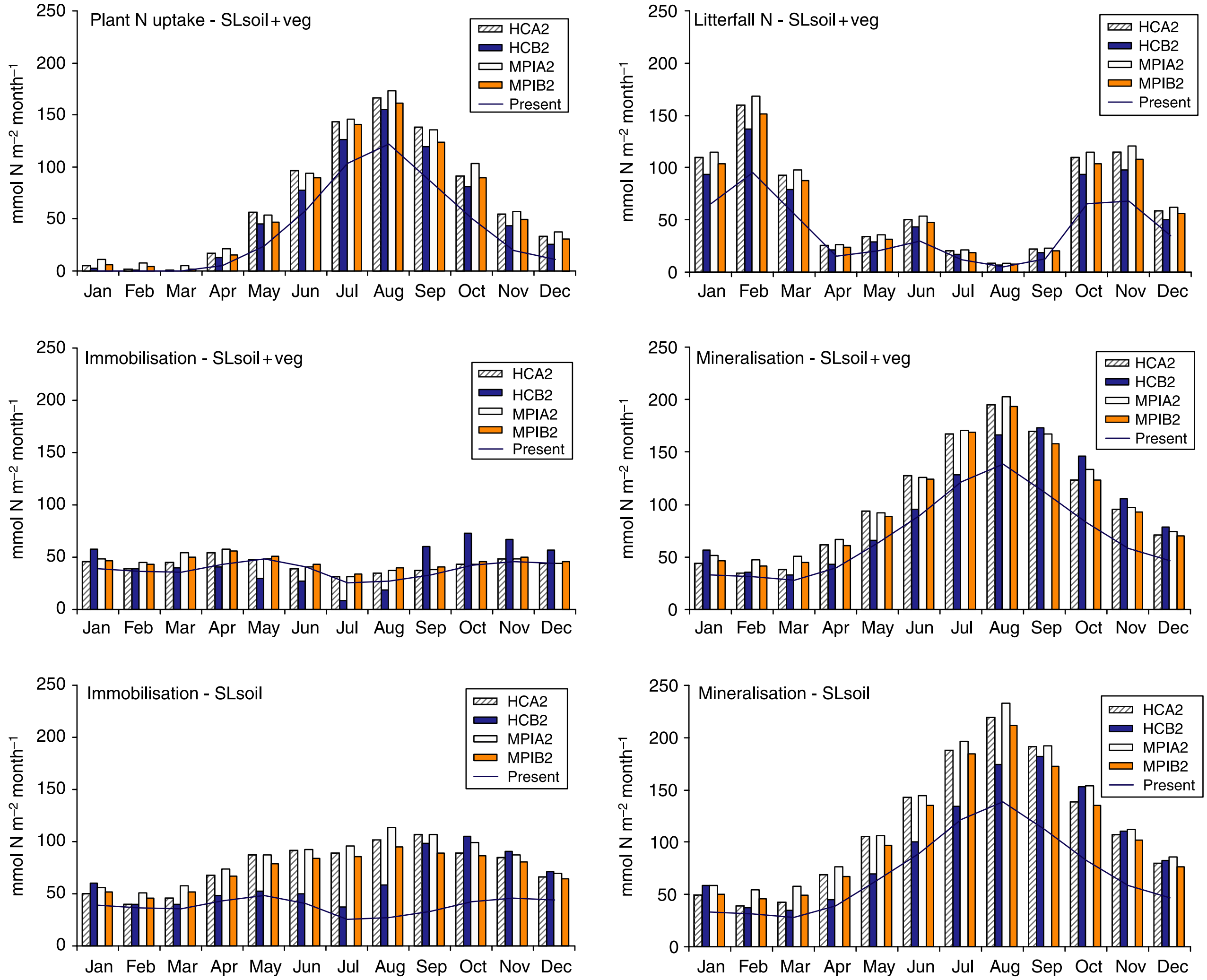

Figure 4 | Twelve-year mean monthly plant N uptake, litter fall N, immobilization and mineralization for the present-day (solid line) and the four scenario estimates of storylines SLsoil and SLsoil + veg, respectively. For SLsoil there was assumed no change in plant N uptake and litter N; thus all scenarios were assumed equal to present day. Mineralization and immobilization are results of the MAGIC simulations. Striped, black, white and grey bars represent HCA2, HCB2, MPIA2 and MPIB2 scenarios, respectively. Units are in mmol N m^{-2} $month^{-1}$.

SLsoil + veg (Table 4 and Figure 5). With the reduction and increase of the soil organic C and N pools, respectively, the simulated soil C/N ratios were projected to continue to decline under all scenarios, slightly more for SLsoil compared to SLsoil + veg (Figure 5). Moreover, NO_3 leaching to surface water was projected to increase for all scenarios, though much higher for SLsoil compared to SLsoil + veg. In the year 2100 the highest (MPIA2, SLsoil) and lowest (HCA2, SLsoil + veg) annual NO_3 leaching rates were 48.7 and 15.7 μmol l^{-1}, respectively (Table 4).

The simulated seasonal NO_3 pattern for the scenario period for HCB2 differed from the other scenarios in both storylines. Whereas simulated NO_3 leaching patterns for the other scenarios in SLsoil + veg were slightly skewed compared to present-day simulations (highest NO_3 leaching during spring compared to winter at present), HCB2 showed comparable dynamics to the present-day situation but with increased levels of NO_3 leaching (Figure 6, lower panel). Moreover, MAGIC simulated more than a twofold increase in NO_3 levels and the seasonal pattern levelled off for the other scenarios in SLsoil, while HCB2 showed the same seasonal pattern as the present-day simulation, but with a twofold increased in the NO_3 level (Figure 6, upper panel).

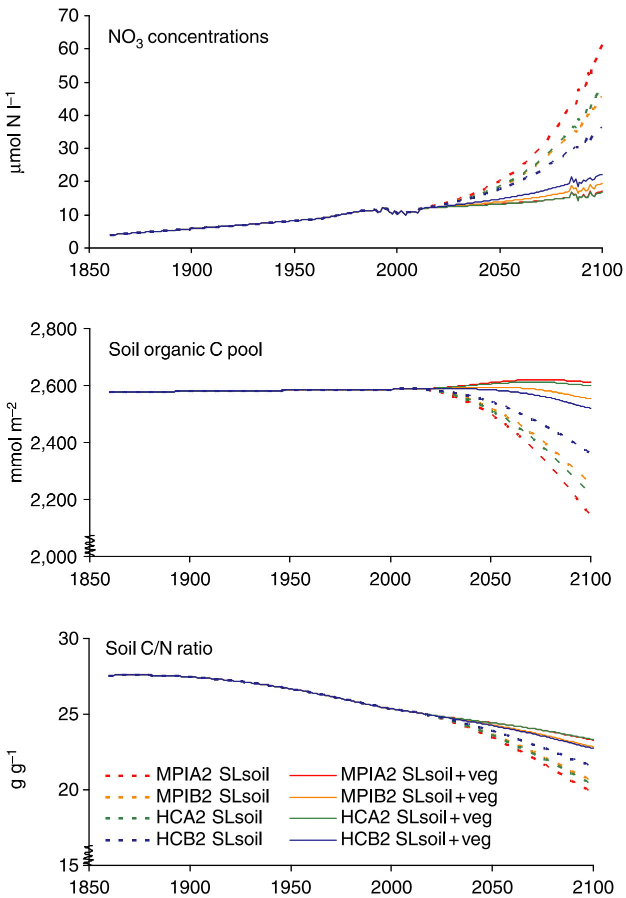

Figure 5 | Simulated mean annual NO_3 surface water concentration, $\mu mol\,l^{-1}$ (upper panel), soil organic C pool, $mmol\,m^{-2}$ (middle panel) and soil C/N ratio, $g\,g^{-1}$ (lower panel) for SLsoil (dashed lines) and SLsoil + veg (solid lines) during 1860–2100. Green, blue, red and orange lines represent HCA2, HCB2, MPIA2 and MPIB2 scenarios, respectively.

DISCUSSION

The storylines

Assumptions

The underlying assumption for SLsoil was that, in the future, elements other than N and temperature may limit vegetation growth, such as phosphorus (P) and potassium (K) (Britton *et al.* 2008), or water. P may be a key limiting resource for *Calluna* (Roem *et al.* 2002). Even with the future reduction in moisture, as projected for the summer at Øygardsbekken, water availability will probably be adequate and thus not limit plant growth. Furthermore, *Calluna* plants have proven to increase water use efficiency if water availability is scarce and combined with high N availability (Gordon *et al.* 1999).

For SLsoil + veg the underlying assumption was that plant foliage will become enriched in N due to increased N uptake, and consequently litter C/N will decrease (Norby & Cotrufo 1998; Hobbie 2000). These two storylines gave two contrasting results, i.e. large increased (SLsoil) and small increased (SLsoil + veg) NO_3 leaching to surface water. The simulated future NO_3 concentrations suggest larger differences and thus uncertainties between the two storylines compared to the differences between the scenarios (and thus the different GCMs) within each storyline. In terms of modelling N leaching, the implication for this is that the uncertainties in scientific understanding of N cycling exceed the uncertainties in the climate predictions.

Net primary productivity versus soil decomposition

Whether SLsoil or SLsoil + veg is the more likely can also be viewed as a battle between net primary production (NPP) and soil organic matter decomposition. NPP is the net result of CO_2 fixation by photosynthesis and CO_2 loss by plant respiration. The question as to whether climate warming will increase the rate of primary production (and hence the rate of litter C production) more than the increase in the rate of decomposition of soil organic matter has been the subject of many theoretical as well as modelling studies (e.g. Kirschbaum 1995, 2000, 2006 and references therein). This determines the future change in C stored in the soil, which in turn affects the C/N ratio. A literature survey on temperature sensitivities of NPP and decomposition concluded that laboratory incubations, soil warming experiments, field measurements of soil respiration and inference of organic C turnover times from isotope ratios all indicate that the temperature sensitivity of decomposition greatly exceeds that of NPP, particularly at low temperatures (Kirschbaum 2000). The strongest evidence came from laboratory incubations in which most confounding factors could be excluded, and the data suggested that a temperature increase of 1°C could ultimately lead to a loss of 10% of soil organic C in regions of the world with large C stores (Kirschbaum 1995). Thus, according to these analyses and in agreement with our model results from SLsoil,

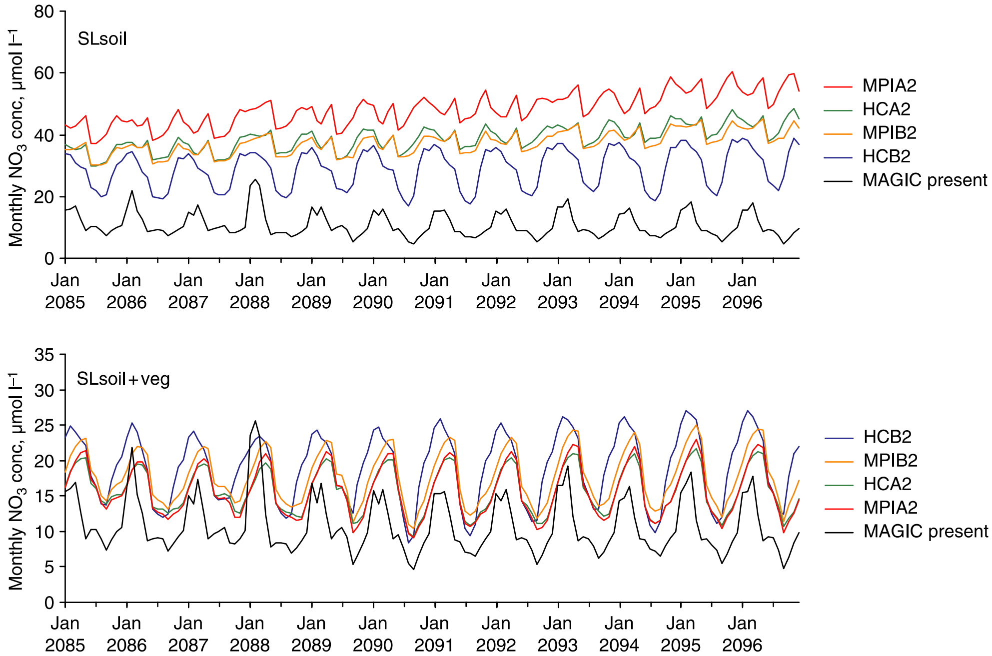

Figure 6 | Simulated monthly seasonal NO_3 surface water pattern for SLsoil (upper panel) and SLsoil + veg (lower panel) during the scenario period 2085–2096 for the different scenarios (coloured lines) compared to present-day simulations during 1993–2004 (black line). Green, blue, red and orange lines represent HCA2, HCB2, MPIA2 and MPIB2 scenarios, respectively. Units are in μmol l^{-1}year^{-1}. Note different scaling on the *y* axis. See text for details.

the Øygardsbekken catchment will probably become a net source of C with future warming.

Many ecosystem modelling studies have been driven by the need to estimate rates of C sequestration and storage in the future (e.g. Worrall *et al.* 2007). An increase in atmospheric CO_2 (the potential driver of global change) in combination with temperature is often considered (e.g. Norby & Lou 2004). Only in a few cases have studies also considered the effects on NO_3 in runoff (Rastetter *et al.* 1991; Wright *et al.* 1998; Aber *et al.* 2002). Rastetter *et al.* (1991) developed a generalized ecosystem model for C and N cycles that focuses on the changes in vegetation and soil under scenarios of climate change. The model was originally applied to a hardwood forest and an arctic tundra ecosystem. The results indicate that for the hardwood forest under a future warmer and CO_2-rich climate the rate of mineralization of soil organic matter will increase more than the increase in primary production; thus there will be net loss of soil organic matter and loss of NO_3 to runoff. This is analogous to the simulation obtained for SLsoil at Øygardsbekken. In the case of the arctic tundra the ecosystem was initially poor in N and all of the mineralized N was retained in the ecosystem.

Effect of soil moisture

Climate-change-related rainfall extremes (extremely wet or dry conditions) could periodically provide either a negative or positive feedback on temperature-induced increases in decomposition rate. For instance, the Øygardsbekken catchment may experience drought during extremely warm, dry summers in the future (and thus experience high evaporative loss of soil water). Jensen *et al.* (2003) reported contrasting results from a drought treatment on two heathland sites located in Denmark and the UK, respectively. At the drier site in Denmark, the drought treatment reduced soil microbial decomposition by 27% and also reduced soil solution C and N. However, the wet site in the UK seemed to benefit from the drought treatment, causing decomposition to increase by 22%. Øygardsbekken is comparable to the UK site, thus drought might give a positive feedback on temperature-induced increases in decomposition rate. This may in turn result in increased leaching of NO_3 in the future during very dry periods in the warm season, possibly increasing baseflow NO_3 concentrations when N is most limited for biota.

On the other hand, during high rainfall events soils may become water saturated, inducing anaerobic conditions favourable for few N processes (i.e. denitrification). This is relevant, however, only for the 6% of the Øygardsbekken catchment that is covered by peat. In general the catchment soils are thin and patchy, and the steep elevation gradient (180–540) causes relatively fast water movement through the soil (Kaste *et al.* 1997) preventing water logging elsewhere in the cathment.

Other ecosystem effects of climate change

Over the next 100 years there are likely to be other effects of increased temperature and N deposition that may occur at the Øygardsbekken catchment in addition to those already accounted for in the storylines.

For example, the vegetation type may change (Walther *et al.* 2002). In parts of the UK and the Netherlands in recent years, the characteristic ericaceous shrub vegetation of heath and moorland habitats has been replaced by grass species (Aerts & Heil 1993; Marrs 1993; Bobbink *et al.* 1998; Todd *et al.* 2000). Changes in traditional management practices (e.g. burning, cutting and grazing) have been partly responsible for these changes, but increased atmospheric deposition of N may be the main cause (Bobbink *et al.* 1998). Thus ecosystem succession from heather to grass-dominant vegetation may take place at the Øygardsbekken catchment. Over the longer term, increased temperature might result in conversion to mountain birch forest. This will result in substantial accumulation of C and N in biomass. Change in vegetation cover will probably affect the rates of many N and C processes in vegetation and soil, including plant uptake, litterfall, decomposition, mineralization and immobilization. It is not known how these changes might affect future NO_3 leaching.

Comparison with other model applications

The Norwegian CLIMEX experiment provided one of the few large-scale whole ecosystem manipulations with both increased temperature and increased CO_2 (van Breemen *et al.* 1998). The experiment was conducted at Risdalsheia, Norway, and ran for 3 years. The site is similar to Øygardsbekken; many of the C and N parameters used here come from the CLIMEX experiment. Data from the CLIMEX experiment have been used to evaluate the ability of MAGIC and the model MERLIN (Model of Ecosystem Retention and Loss of Inorganic Nitrogen) (Cosby *et al.* 1997) to simulate changes in NO_3 in runoff caused by the increase in temperature and CO_2. MERLIN is a model of C and N fluxes and pools that focuses on simulating inorganic N in runoff. MERLIN is similar in structure to MAGIC but incorporates two soil C pools, one with relatively rapid turnover (labile C) and one with slow turnover (recalcitrant C). Both the MERLIN (Wright *et al.* 1998) and MAGIC applications (Emmett *et al.* 1998) gave simulations consistent with the experimental data. Increased decomposition of soil organic matter with release of inorganic N to soil solution was the major process in the models that explained the changes in NO_3 in runoff. The CLIMEX experiment offers one of the few sets of relevant experimental ecosystem-scale data that can be used to evaluate effects of future climate change and the ability of models to simulate these changes. The two model applications at CLIMEX are consistent with the monthly MAGIC application made here to Øygardsbekken. All simulations invoke increased decomposition of soil organic matter, and all indicate that a fraction of the released N will appear as NO_3 in runoff, consistent with the experiment at CLIMEX. The heathland manipulation experiments of the CLIMOOR project also indicate increased net N mineralization in response to warming, but only under conditions when moisture is not limiting (Emmett *et al.* 2004). CLIMOOR is a plot-scale study with measurements of soil solution, and thus does not give direct information on changes in runoff chemistry.

One shortcoming of CLIMEX, CLIMOOR and other such experiments are their short duration relative to the long-term responses simulated by modelling exercises. It is unknown if the increased decomposition rate would have persisted over the long term, as assumed in the MAGIC applications here, or whether it is only a transient phenomenon that lasts until a relatively small pool of labile organic matter has been lost. For instance, in a single-factorial soil warming experiment at the Harvard Forest in Petersham, MA, USA initiated in 1991, the first four years of warming showed a dramatic 26–75% increase in soil respiration (Peterjohn *et al.* 1994). However, by 2000, 10 years after the initiation of treatments, soil respiration in the warmed plots was no longer significantly different from the control. Melillo *et al.* (2002) hypothesized that the reduced response in the warmed plots was due to a depletion of labile carbon stocks (e.g. consisting predominantly of simple sugars and amino acids), which may be more temperature sensitive than more recalcitrant carbon fractions (consisting of more complex aromatic compounds). Thus caution should be used when attempting to extrapolate short-term experimental responses to infer longer-term effects resulting from climate change (Rastetter 1996; Norby & Luo 2004).

N dynamics at Øygardsbekken have previously been modelled with the Integrated Nitrogen in Catchments model, INCA-N (Kaste *et al.* 2004, 2006). INCA-N is a process-based and semi-distributed model, which integrates hydrology, basin and river N processes, and simulates daily NO_3 and NH_4 concentrations in soil and surface water (Whitehead *et al.* 1998; Wade *et al.* 2002). In the study by Kaste *et al.* (2004), the INCA-N application simulated future N leaching based on available N deposition and climate scenarios for the next 50 years. As a result of climate change, the INCA-N model predicted markedly reduced duration and amounts of snow cover, resulting in more frequent winter floods and reduced or even absent snowmelt flooding in spring. In contrast to the present study, INCA-N predicted that the increased N mineralization due to climate change would be largely balanced by a corresponding increase in N retention, and relatively small increases in NO_3 leaching rates were indicated.

The dynamic relationship between N production and consumption in catchments strongly depends on the temperature responses of the key N transformation processes modelled. INCA-N lacks a dynamic soil N pool resulting from the long-term balance between N accumulation and release processes. Hence, it does not account for N accumulation in soils and possible effects on long-term N retention capacity. In a follow-up study by Kaste *et al.* (2006), a link between the MAGIC and INCA-N models was established to simulate the long-term balance between N accumulation and release processes. With this combination of models a 40–50% increase in streamwater NO_3 fluxes in the Bjerkreim river basin (in which Øygardsbekken is located) due to projected climate change between 1980–2000 and 2081–2100 was simulated. This is approximately the same magnitude of increase as simulated here under the scenarios of SLsoil.

The main advantages of INCA-N over MAGIC are the daily time step instead of monthly and the inclusion of a rainfall–runoff routine. Thus short-term changes (storm events or episodes) may be accounted for by the model, although the drawback is the need for higher resolution in data. On the other hand, INCA-N does not include the "long-term memory" of MAGIC. INCA-N on its own might have more success in simulating the year-to-year variations in the seasonal NO_3 leaching, while MAGIC simulates the long-term trends.

There have been few other modelling studies of NO_3 leaching from undisturbed boreal ecosystems. Collins & Jenkins (2001) used an earlier version of MAGIC (which did not include N processes) to simulate the seasonal pattern in NO_3 concentrations in runoff at Dargall Lane, Scotland, UK. Their study did not involve specification of the rates of the various N processes within the catchment but rather merely the apportioning of the observed N leached into months of the year. Aber *et al.* (2002) used PnET-CN to study the relative importance of several driving factors in explaining the 40-year record of NO_3 in runoff at the undisturbed forested catchment W6 at Hubbard Brook, NH, USA. PnET-CN is a process-oriented model of forest C, N and water dynamics. They found that the observed increase in annual NO_3 levels in the 1970s followed by the decline in the 1980s could be explained only by taking into account changes in N deposition, physical and biotic disturbances, as well as inter-annual variations in climate. In the Hubbard Brook case they hypothesized that a period of drought in the late 1960s caused a greater increase in mineralization than increase in plant activity, thus mobilizing more NO_3.

Seasonal changes and the effects on downstream watercourses

The overall higher NO_3 leaching rates simulated for SLsoil compared to SLsoil + veg were due to the lower modelled soil C/N ratios under SLsoil. SLsoil + veg entails a higher supply of organic C in litter, which acts to maintain the present-day soil C pool. For all scenarios except HCB2, the less pronounced sinusoidal pattern associated with the high leaching rates of the storylines of SLsoil is in accordance with the N saturation stage 3 from Stoddard (1994). The lower mineralization rate simulated by MAGIC during spring and summer for HCB2 (SLsoil) was the likely cause for summer depletion and thus a similar seasonal NO_3 pattern as the present-day (1993–2004) situation (Figure 6, upper panel). The simulated seasonal NO_3 pattern for the scenario period for SLsoil + veg HCB2 also differed from the other scenarios. Whereas simulated NO_3 leaching patterns for the other scenarios of SLsoil + veg were slightly delayed compared to present-day simulations, HCB2 showed comparable dynamics to the present-day situation but with increased levels of NO_3 leaching. This may be explained by the higher mineralization rates in late autumn and winter for this scenario relative to the others (Figure 4). The moderate NO_3 leaching rates for SLsoil + veg can be classified as N saturation stage 2 according to Stoddard (1994). For comparison, at present (1993–2004) the Øygardsbekken catchment is classified as saturation stage 1.

As a consequence of the projected increased in NO_3 leaching, acidification of surface waters will increase, thus offsetting ongoing recovery from acidification due to reductions in sulfur deposition. Furthermore, the slightly skewed seasonal NO_3 pattern (i.e. highest leaching rates during spring compared to winter at present) simulated for the scenarios of SLsoil + veg may have a detrimental effect on fish populations since the highest leaching rates coincide with the most acid-sensitive life stage of spawning and egg hatching. The increased N loading may also stimulate growth of possible N-limited benthic algae and macrophytes along the river channels and thus lead to undesirable eutrophication effects (Lindstrøm & Johansen 1995). As the N-rich water enters the coastal areas, eutrophication might also result in estuarine ecosystems (Howarth & Marino 2006). In a multi-model application to the Bjerkreim River, of which Øygardsbekken is a tributary, Kaste *et al.* (2006) found that future climate change would result in 15–20% increase in primary production in the summer in this estuary. The results from the present study indicate that effects on the estuary would be most pronounced during spring for the scenarios of SLsoil + veg due to the skewed seasonal surface water NO_3 pattern.

CONCLUSIONS

Implementing climate change scenarios for temperature and N deposition in the calibrated model indicated increased future NO_3 leaching to surface water. The scenarios were run with the MAGIC calibration that simulated the seasonal NO_3 pattern over the 12-year record; the model explained almost 70% of the variation in monthly NO_3 concentrations and nearly 90% of the variations in flux. With the highest projected leaching rates the future seasonal pattern was less pronounced (saturation stage 3). For the moderate future predictions the seasonal pattern was delayed, with the highest peak in spring instead of in winter. The simulated change in NO_3 leaching to surface water could offset the ongoing recovery from acidification due to reductions in acid deposition.

Modelling the effect of increased temperature and N deposition for the next 100 years is fraught with large uncertainty; thus the modelling results presented here are of a rather exploratory nature. The accumulating evidence from single- and multi-factor ecosystem-scale manipulation experiments has greatly improved our understanding of short-term responses of terrestrial ecosystems and many of their components to elevated atmospheric CO_2, warming, changes in water availability and N deposition (see, e.g., Emmett *et al.* 1998; Stitt & Krapp 1999; Rustad *et al.* 2001; Hyvönen *et al.* 2007). For long-term simulations, however, experimental data are still lacking. For instance, in areas that have experienced decades of elevated N deposition the long-term fate of the N retained and stored in soil is uncertain. Correspondingly, it is not know whether the NPP or soil decomposition will increase more with climate change in different types of ecosystems, and thus there is high uncertainty associated with the fate of the large store of C in soil.

Concern exists that the initial responses in ecosystem manipulations may be transitory, and caution should be used when attempting to extrapolate short-term experimental responses to infer longer-term effects (Norby & Luo 2004). Furthermore, data from these experiments may be difficult to generalize and transfer to other ecosystems as the responses induced by the perturbations may be site-specific and depend on the history and state of the ecosystem. To do these extrapolations and to provide estimates for inputs to ecosystem models it will be necessary to improve the understanding of the change in response patterns over time, including alterations in the magnitude, direction and rate of change of the response. According to Rustad (2006) these issues represent one of the biggest challenges in accurately predicting long-term changes in ecosystems and associated feedbacks to the atmospheric and climate system.

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REFERENCES

Aber, J. D., Ollinger, S. V., Driscoll, C. T., Likens, G. E., Holmes, R. T., Freuder, R. J. & Goodale, C. L. 2002 Inorganic nitrogen losses from a forested ecosystem in response to physical, chemical, biotic, and climatic perturbations. *Ecosystems* **5**, 648–658.

Aerts, R. & Heil, G. W. 1993 *Heatlands: Patterns and Processes in a Changing Environment*. Kluwer, Dordrecht.

Beniston, M., Diaz, H. F. & Bradley, R. S. 1997 Climatic change at high elevation sites: an overview. *Clim. Change* **36**, 233–251.

Bobbink, R., Hornung, M. & Roelofs, J. G. M. 1998 The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. *J. Ecol.* **86**, 717–738.

Britton, A. J., Helliwell, R. C., Fisher, J. M. & Gibbs, S. 2008 Interactive effects of nitrogen deposition and fire on plant and soil chemistry in an alpine heathland. *Environ. Pollut.* **156**, 409–416.

Christensen, J. H. & Christensen, O. B. 2007 A summary of the PRUDENCE model projections of changes in European climate by the end of this century. *Clim. Change* **81**, 7–30.

Collins, R. & Jenkins, A. 2001 Incorporation of seasonal nitrogen dynamics within a long-term acidification model. *Water Air Soil Pollut. Focus* **1**, 483–494.

Cormack, E. & Gimingham, C. H. 1964 Litter production by *Calluna vulgaris* (L) HULL. *J. Ecol.* **52**(2), 285–297.

Cosby, B. J., Ferrier, R. C., Jenkins, A., Emmett, B. A., Wright, R. F. & Tietema, A. 1997 Modelling the ecosystem effects of nitrogen deposition at the catchment scale: model of ecosystem retention and loss of inorganic nitrogen (MERLIN). *Hydrol. Earth Syst. Sci.* **1**, 137–158.

Cosby, B. J., Ferrier, R. C., Jenkins, A. & Wright, R. F. 2001 Modelling the effects of acid deposition: refinements, adjustments and inclusion of nitrogen dynamics in the MAGIC model. *Hydrol. Earth Syst. Sci.* **5**, 499–518.

Cosby, B. J., Hornberger, G. M., Galloway, J. N. & Wright, R. F. 1985*a* Time scales of catchment acidification: a quantitative model for estimating freshwater acidification. *Environ. Sci. Technol.* **19**, 1144–1149.

Cosby, B. J., Hornberger, G. M., Galloway, J. N. & Wright, R. F. 1985*b* Modelling the effects of acid deposition: assessment of a lumped parameter model of soil water and streamwater chemistry. *Water Resour. Res.* **21**, 51–63.

Davidson, E. A. & Janssens, L. A. 2006 Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature* **440**, 165–173.

Emmett, B. A., Beier, C., Estiarte, M., Tietema, A., Kristensen, H. L., Williams, D., Penuelas, J., Schmidt, I. & Sowerby, A. 2004 The response of soil processes to climate change: results from manipulation studies of shrublands across an environmental gradient. *Ecosystems* **7**, 625–637.

Emmett, B. A., Boxman, A. D., Bredemeier, M., Moldan, F., Gundersen, P., Kjønaas, O. J., Schleppi, P., Tietema, A. & Wright, R. F. 1998 Predicting the effects of atmospheric nitrogen deposition in conifer stands: evidence from the NITREX project. *Ecosystems* **1**, 352–360.

Gordon, C., Woodin, S. J., Mullins, C. E. & Alexander, I. J. 1999 Effects of environmental change, including drought, on water use by competing *Calluna vulgaris* (heather) and *Pteridium aquilinum* (bracken). *Func. Ecol.* **13**(Suppl. 1), 96–106.

Gudem, R. & Hovland, I. 1999 *The Influence of Natural Conditions and Production Costs in Agriculture*, Note 1999:19, Norwegian Agricultural Economics Research Institute (NILF), Oslo.

Hobbie, S. 2000 Interactions between litter lignin and soil nitrogen availability during leaf litter decomposition in a Hawaiian Montane forest. *Ecosystems* **3**(5), 484–494.

Hole, L. R., de Wit, H. A. & Aas, W. 2008 Influence of summer and winter climate variability on nitrogen wet deposition in Norway. *Hydrol. Earth Syst. Sci.* **12**, 405–414.

Hole, L. R. & Engardt, M. 2008 Climate change impact on atmospheric nitrogen deposition in northwestern Europe: a model study. *Ambio* **37**(1), 9–17.

Howard, D. M. & Howard, P. J. A. 1993 Relationships between CO_2 evolution, moisture-content and temperature for a range of soil types. *Soil Biol. Biochem.* **25**(11), 1537–1546.

Howarth, R. W. & Marino, R. 2006 Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. *Limnol. Oceanogr.* **51**(1, part 2), 364–376.

Hyvönen, R., Ågren, G. I., Linder, S., Persson, T., Cotrufo, M. F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I. A., Jarvis, P. G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R. J., Oren, R., Pilegaard, K., Ryan, M. G., Sigurdsson, B. D., Strömgren, M., Oijen, M. v. & Wallin, G. 2007 The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. *New Phytotol.* **173**, 463–480.

IPCC 2001 *Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC)*.

IPCC 2007 *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I In Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. (Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M. & Miller, H. L. (Eds)). Cambridge University Press, Cambridge.

Jonasson, S., Castro, J. & Michelsen, A. 2004 Litter, warming and plants affect respiration and allocation of soil microbial and plant C, N and P in arctic mesocosms. *Soil Biol. Biochem.* **36**(7), 1129–1139.

Jensen, K. D., Beier, C., Michelsen, A. & Emmett, B. A. 2003 Effects of experimental drought on microbial processes in two temperate healthlands at contrasting water conditions. *Appl. Soil. Ecol.* **24**, 165–176.

Kaste, Ø., Henriksen, A. & Hindar, A. 1997 Retention of atmospherically-derived nitrogen in subcatchments of the Bjerkreim River in southwestern Norway. *Ambio* **26**, 296–303.

Kaste, Ø., Rankinen, K. & Lepisto, A. 2004 Modelling impacts of climate and deposition changes on nitrogen fluxes in northern catchments of Norway and Finland. *Hydrol. Earth Syst. Sci.* **8**(4), 778–792.

Kaste, Ø., Wright, R. F., Barkved, L. J., Bjerkeng, B., Engen-Skaugen, T., Magnusson, J. & Sælthun, N. R. 2006 Linked models to assess the impacts of climate change on nitrogen in a Norwegian river basin and fjord system. *Sci. Total Environ.* **365**, 200–222.

Kirschbaum, M. U. F. 1995 The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biol. Biochem.* **27**, 753–760.

Kirschbaum, M. U. F. 2000 Will changes in soil organic carbon act as a positive or negative feedback on global warming? *Biogeochemistry* **48**, 21–51.

Kirschbaum, M. U. F. 2006 The temperature dependence of organic-matter decomposition–still a topic of debate. *Soil Biol. Biochem.* **38**, 2510–2518.

Körner, C. 1999 *Alpine Plant Life: Functional Plant Ecology of High Mountain Ecosystems*. Springer-Verlag, Berlin.

Leiros, M. C., Trasar-Cepeda, C., Seoane, S. & Gil-Sotres, F. 1999 Dependence of mineralization of soil organic matter on temperature and moisture. *Soil Biol. Biochem.* **31**(3), 327–335.

Lindstrøm, E. A. & Johansen, S. W. 1995 Factors controlling periphyton growth in rivers of low nutrient content. Experiments with nutrient-diffusing substrates in 1994. In: *Nitrogen From Mountains to Fjords* Newsletter 3/95.

Lloyd, J. & Taylor, J. A. 1994 On the temperature-dependence of soil respiration. *Funct. Ecol.* **8**(3), 315–323.

Marrs, R. H. 1993 An assesment on change in Calluna heathland. *Biol. Conserv.* **65**, 133–139.

Melillo, J. M., Steudler, P. A., Aber, J. D., Newkirk, K., Lux, H., Bowles, F. P., Catricala, C., Magill, A., Ahrens, T. & Morrisseau, S. 2002 Soil warming and carbon-cycle feedbacks to the climate system. *Science* **298**, 2173–2176.

Menzel, A. & Fabian, P. 1999 Growing season extended in Europe. *Nature* **397**(6721), 659.

Myneni, R. B., Keeling, C. D., Tucker, C. J., Asrar, G. & Nemani, R. R. 1997 Increased plant growth in the northern high latitudes from 1981 to 1991. *Nature* **386**(6626), 698–702.

Norby, R. J. & Cotrufo, M. F. 1998 Global change–a question of litter quality. *Nature* **396**(6706), 17–18.

Norby, R. J. & Luo, Y. Q. 2004 Evaluating ecosystem responses to rising atmospheric CO_2 and global warming in a multi-factor world. *New Phytol.* **162**(2), 281–293.

Peterjohn, W. T., Melillo, J. M., Steudler, P. A., Newkirk, K. M., Bowles, F. P. & Aber, J. D. 1994 Responses of trace gas fluxes and N availability to experimentally elevated soil temperatures. *Ecol. Appl.* **4**, 617–625.

Pilkington, M. G., Caporn, S., Carroll, J. A., Cresswell, N., Lee, J. A., Ashenden, T. W., Brittain, S. A., Reynolds, B. & Emmett, B. A. 2005 Effects of increased deposition of atmospheric nitrogen on an upland moor: leaching of N species and soil solution chemistry. *Environ. Pollut.* **135**(1), 29–40.

Pitcairn, C. E. R., Leith, I. D., Fowler, D., Hargreaves, K. J., Moghaddam, M., Kennedy, V. H. & Granat, L. 2001 Foliar nitrogen as an indicator for nitrogen deposition and critical load exceedance on a European scale. *Water Air Soil Pollut.* **130**, 1037–1042.

Posch, M., Schöpp, W., Johansson, M. & Mylona, S. 2003 *150 Years of Sulphur and Nitrogen Deposition in Europe (1880–2030)*. IIASA, Laxenburg, Austria.

Rastetter, E. 1996 Validating models of ecosystem response to global change. *Bioscience* **46**(3), 190–198.

Rastetter, E. B., Ryan, M. G., Shaver, G. R., Melillo, J. M., Nadelhoffer, K. J., Hobbie, J. E. & Aber, J. D. 1991 A general biogeochemical model describing the responses of the C and N cycles in terrestrial ecosystems to changes in CO_2, climate, and N deposition. *Tree Physiol.* **9**, 101–126.

REGCLIM 2005 *Norway's Climate in 100 years; Uncertainty and Risk*. (Iversen, T., Benestad, R., Haugen, J. E., Kirkevåg, A., Sorteberg, A., Debernard, J., Grønås, S., Hanssen Bauer, I., Kvamstø, N. G., Martinsen, E. A. & Engen Skaugen, T. (Eds)). RegClim Brochure, Met.no, Oslo, Norway (in Norwegian).

Roem, W. J., Klees, H. & Berendse, F. 2002 Effects of nutrient addition and acidification on plant species diversity and seed germination in heathland. *J. Appl. Ecol.* **39**(6), 937–948.

Rustad, L. E. 2006 From transient to steady-state response of ecosystems to atmospheric CO2-enrichment and global climate change: conceptual challenges and need for an integrated approach. *Plant Ecol.* **182**(1–2), 43–62.

Rustad, L. E., Campbell, J. L., Marion, G. M., Norby, R. J., Mitchell, M. J., Hartley, A. E., Cornelissen, J. H. C. & Gurevitch, J. 2001 A meta-analysis of the response of soil respiration, net nitrogen mineralization, and aboveground plant growth to experimental ecosystem warming. *Oecologia* **126**(4), 543–562.

Schmidt, I., Jonasson, S. & Michelsen, A. 1999 Mineralization and microbial immobilization of N and P in arctic soils in relation to season, temperature and nutrient amendment. *Appl. Soil Ecol.* **11**(2-3), 147–160.

Schöpp, W., Posch, M., Mylona, S. & Johansson, M. 2003 Long-term development of acid deposition (1880–2030) in sensitive freshwater regions in Europe. *Hydrol. Earth Syst. Sci.* **7**, 436–446.

Sjøeng, A. M. S., Wright, R. F. & Kaste, Ø. 2009 Modelling seasonal nitrate concentrations in runoff using MAGIC. *Hydrol. Res.* **40**(2).

Stitt, M. & Krapp, A. 1999 The interaction between elevated carbon dioxide and nitrogen nutrition: the physiological and molecular background. *Plant Cell Environ.* **22**(6), 583–621.

Stoddard, J. L. 1994 Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. In: Baker, L. A. (ed.) *Environmental Chemistry of Lakes and Reservoirs*. American Chemical Society, Washington, DC, pp. 223–284.

Tjomsland, T. & Rohrlack, T. 2008 *Simulated Effects on Hydrophysics and Water Quality in Lakes due to Climate Changes*. NIVA report no. 55573.

Todd, P. A., Phillips, J. D. P., Putwain, P. D. & Marrs, R. H. 2000 Control of Molinia caerulea on moorland. *Grass Forage Sci.* **55**(2), 181–191.

van Breemen, N., Jenkins, A., Wright, R. F., Arp, W. J., Beerling, D. J., Berendse, F., Beier, C., Collins, R., van Dam, D., Rasmussen, L., Verburg, P. S. J. & Wills, M. A. 1998 Impacts of elevated carbon dioxide and temperature on a boreal forest ecosystem (CLIMEX project). *Ecosystems* **1**, 345–351.

Wade, A. J., Durand, P., Beaujouan, V., Wessel, W. W., Raat, K. J., Whitehead, P. G., Butterfield, D., Rankinen, K. & Lepisto, A. 2002 A nitrogen model for European catchments: INCA, new model structure and equations. *Hydrol. Earth Syst. Sci.* **6**, 559–582.

Walther, G. R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T. J. C., Fromentin, J. M., Hoegh-Guldberg, O. & Bairlein, F. 2002 Ecological responses to recent climate change. *Nature* **416**(6879), 389–395.

Whitehead, P. G., Wilson, E. J. & Butterfield, D. 1998 A semi-distributed integrated nitrogen model for multiple source assessment in catchments (INCA): part I–model structure and process equations. *Sci. Total Environ.* **210**, 547–558.

Worrall, F., Burt, T., Adamson, J., Reed, M., Warburton, J., Armstrong, A. & Evan, M. 2007 Predicting the future carbon budget of an upland peat catchment. *Clim. Change* **85**, 139–158.

Wright, R. F. 1998 Effect of increased CO_2 and temperature on runoff chemistry at a forested catchment in southern Norway (CLIMEX project). *Ecosystems* **1**, 216–225.

Wright, R. F., Beier, C. & Cosby, B. J. 1998 Effects of nitrogen deposition and climate change on nitrogen runoff at Norwegian boreal forest catchments: the MERLIN model applied to Risdalsheia (RAIN and CLIMEX projects). *Hydrol. Earth Syst. Sci.* **2**, 399–414.

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