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Prognoses for future acidification recovery of water, soils and forests:

Dynamic modeling of Norwegian data from ICP Forest, ICP Integrated Monitoring and ICP Waters

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REPORT

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Abstract

The project reported here was a cooperation between the National Focal Centers for four of the ICPs in Norway: ICP Mapping and Modeling, ICP Waters, ICP Forest and ICP Integrated Monitoring. Dynamic modeling was carried out using data from several sites in the ICP networks, with the aim of making predictions on the future acidification status for surface waters, forest and soils in Norway. Predictions are made for three different deposition scenarios. At two of the sites, the model predictions suggest that the Current Legislation scenario will not promote water qualities sufficient for sustainable fish populations, while the scenario seems sufficient for the other sites. Under the Maximum Feasible Reduction scenario one of the sites still will not reach a sufficiently high ANC. In general, modeling results for forest soils agree with results from previous investigations stating that surface water acidification is more severe than the soil acidification. However, the results suggest that there has been soil acidification at all sites as a result of acid deposition and that the base saturation will not be built up again to pre-industrial levels during the next 50 years at any of the sites, not even with the Maximum Feasible Reduction Scenario.

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1. forsuring	1. acidification
2. dynamisk modellering	2. dynamic modeling
3. gøteborgprotokollen	3. Gothenburg protocol
4. framtidsscenarier	4. Future scenarios



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Prognoses for future recovery from acidification of water, soils and forests:

Dynamic modeling of Norwegian data from ICP Forests, ICP IM and ICP Waters

A cooperative project between the
Norwegian Focal Centers for:
- ICP Mapping and Modeling
- ICP Forest
- ICP Waters
- ICP Integrated Monitoring

Preface

The work presented here was conducted in 2001-2002 as a contract to the Norwegian Directorate of Nature Management (DN) under the program "Naturens tålegrenser" (Critical loads). The project has been a cooperation between four national focal centers of the UNECE International Cooperative Programs (ICPs): ICP Forest, ICP Waters, ICP Integrated Monitoring and ICP Modeling and Mapping. Kjetil Tørseth (NILU) has contributed on issues related to deposition, Nicholas Clarke (Skogforsk) has contributed ion issues related to forest soils and forest growth, Brit Lisa Skjelkvåle (NIVA) contributed on surface waters issues. Thorjørn Larssen (NIVA) carried out the modeling.

Oslo, October 2002

Thorjørn Larssen

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Summary

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At Birkenes and Storgama, the model predictions suggest that the Current Legislation scenario is not sufficient to promote annual average ANC values in the future above about 10 µeq L^{-1}. By applying the Maximum Feasible Reduction scenario the ANC is predicted to reach 20 µeq L^{-1} at Storgama and 0 µeq L^{-1} at Birkenes in about 2020. Hence, there is a need for further reductions in the acid deposition in order to get water qualities sufficient for sustainable fish populations.

At Langtjern the ANC has already recovered to a relatively high value (30-35 µeq L^{-1}) and water quality acceptable for fish reproduction is expected in the future. From the model predictions for Langtjern it is uncertain whether further improvements from the observed values can be expected.

For Vikedal considerable improvements are predicted for the future. Large variations from year to year are observed due to variation in seasalt inputs, but the long term trend suggest average ANC of 20 µeq L^{-1} after implementation of the Current Legislation and an additional improvement on 5 µeq L^{-1} with implementation of the Maximum Feasible Reduction scenario.

In general, the modeling results for forest soils agree with results from previous studies stating that surface water acidification is more severe than the soil acidification. However, the results suggest that there has been soil acidification at all sites as a result of acid deposition. Of the modeled effects, perhaps the most serious and long-lasting is the decline in base saturation. It is predicted that base saturation will not be built up again to pre-industrial levels during the next 50 years at any of the sites, not even with the Maximum Feasible Reduction Scenario.

Data from comprehensive monitoring programs, such as the ICP-programs, are of great value in calibrating dynamic acidification models and applications of such models for predictions of future deposition scenarios.

1. Introduction

As acid deposition in Europe decreases as a result of reductions in sulfur and nitrogen emissions, there is an increased need to know whether these reductions are sufficient to promote ecosystem recovery and when this recovery will occur. In order to address these issues, dynamic acidification models are required. The predictive power of dynamic acidification modeling is a crucial issue for determining the usefulness of such models to support policymaking.

Dynamic models for surface water acidification have been developed during the last two decades. Models have been refined and enlarged through inclusion of additional processes as new data and new knowledge have become available (e.g.Cosby et al. 2001).

During the last two decades the policy goals of acidification modeling have changed. Initially, the objective was to illustrate the effects of anthropogenic acid deposition and the benefits of emission reductions. Recently, as substantial reductions in emissions have been implemented, the focus has increasingly changed to issues such as (a) whether the latest internationally-agreed measures are sufficient and (b) *when* waters and soils will recover from acidification (Bull et al. 2001). In order to address these questions policymakers have taken increasing interest in dynamic models, especially as the time component is lacking in steady state models (UNECE 2000).

Dynamic models must be calibrated against observed data. Hence, model predictive performance is related to the availability and quality of monitoring data. Necessary input data for dynamic modeling of acidification and recovery are available through the International Cooperative Programs (ICPs) which are part of the Working Group on Effects of the United Nations Economic Commission for Europe (UNECE) Convention on Long Range Transboundary Air Pollution (CLRTAP). Relevant data are collected in the ICP Waters (International Cooperative Programme on Assessment and Monitoring of Acidification of Rivers and Lakes), ICP Forest (International Cooperative Programme on Assessment and Monitoring of Air Pollution Effects on Forests) and ICP Integrated Monitoring (International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems) as well as EMEP (The Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe). Modeling are carried out through ICP Mapping and Modeling (International Cooperative Programme on Modelling and Mapping of Critical Loads and Levels and Air Pollution Effects, Risks and Trends).

The project reported here was a cooperation between the National Focal Centers for four of the ICPs in Norway: ICP Mapping and Modeling, ICP Waters, ICP Forest and ICP Integrated Monitoring. Dynamic modeling was carried out using data from several sites in the ICP networks, with the aim of making predictions on the future acidification status for surface waters, forest and soils in Norway. Predictions are made for three different deposition scenarios, illustrating the effect of implementing the Current Legislation (includes the Gothenburg protocol) compared to the older Oslo protocol and a Maximum Feasible Reduction scenario.

2. Material and methods

2.1 Site selection

Sites were selected from the Norwegian stations within ICP Waters, ICP Forest and ICP Integrated Monitoring. Focus was put on locations influenced by acid deposition, where surface waters and forest sites are located in the vicinity of each other. In addition, the locations were selected based on data availability, i.e. long time-series measurements and a maximum of necessary model inputs available. The selected locations cover the range of the typical acidified areas of southern Norway:

- Birkenes, Aust-Agder county (ICP Integrated Monitoring, ICP Waters, ICP Forest)
- Langtjern, Buskerud county (ICP Waters, ICP Forest)
- Vikedal/Nedstrand , Rogaland county(ICP Waters, ICP Forest)
- Storgama, Telemark county (ICP Waters)

The Vikedal river is limed, but the water monitoring station is located upstream the limed section. Hence the liming has no influence on the model calibration or predictions. Storgama was included for surface waters even though there is no comparable forest site, because the catchment is typical for the acidified southern Norwegian inland and hence an important sensitive ecosystem.

2.2 The MAGIC model

MAGIC is a lumped-parameter model of intermediate complexity, developed to predict the long-term effects of acidic deposition on surface water chemistry (Cosby et al. 1985; Cosby et al. 2001). The model simulates soil solution and surface water chemistry to predict average concentrations of the major ions. MAGIC calculates for each time step (month or year) the concentrations of major ions under the assumption of simultaneous reactions involving sulfate adsorption, cation exchange, dissolution-precipitation-speciation of aluminum and dissolution-speciation of inorganic carbon. MAGIC accounts for the mass balance of major ions in the soil by book-keeping the fluxes from atmospheric inputs, chemical weathering, net uptake in biomass and loss to runoff.

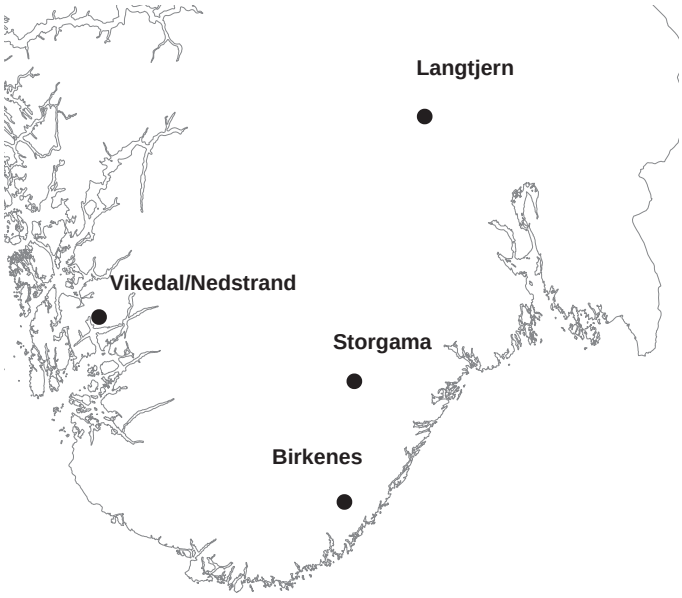

Figure 1. Map of southern Norway showing the locations of the sites.

At the heart of MAGIC is the pool of exchangeable base cations in the soil. As the fluxes to and from this pool change over time owing to changes in atmospheric deposition, the chemical equilibria between soil and soil solution shift to drive changes in surface water chemistry. The degree and rate of change of surface water acidity thus depend both on flux factors and the inherent characteristics of the affected soils.

Data inputs required for MAGIC comprise soil chemical and physical characteristics, input and output fluxes for water and major ions, and net uptake fluxes for vegetation.

MAGIC is usually applied at the catchment or lake scale, but can also be applied to soil profiles and soil solution.

2.3 Model input data and calibration

The deposition input was obtained from the NILU monitoring station closest to the site (data from Aas et al. 2002). The precipitation collector is not necessarily representative for a small catchment or forest plot. Therefore a number of adjustments to the observed data were carried out. Dry deposition must be estimated, as this is only to a limited extent measured. This may be dry deposition of seasalt aerosols and of gaseous SO_2. In the model calibration we used chloride as a conservative tracer, which means we assume that chloride does not take part in chemical processes and that it follows the water through the soil. The ratio between the chloride flux in the stream (or soil solution) and in precipitation is used to adjust the seasalt fractions of the other ions. For sulfate we assume that there is no weathering or uptake in the soil and that the adsorption is small. Hence the dry deposition flux of sulfur is the difference between net input and output fluxes. The assumption of limited sulfate adsorption is tested by comparing the shapes of the long-term trends in the sulfate input and output fluxes.

The entire time series of observed deposition data are given as model inputs. Prior to the period of monitoring, i.e., from 1850 to the 1970s, historic deposition for sulfate, nitrate and ammonium is calculated from European emission estimates using the EMEP model (Posch et al. 2002; Figure 2). This means that the model output is a smooth line for the time period up to the 1970s, after which monitoring data is available. Then the pattern is rather variable throughout the period with observations, as a result of the variations in inputs, both natural and anthropogenic, from year to year. For the forecast scenarios the model outputs become smooth again, as forecast are based on emission estimates and EMEP model calculations (Posch et al. 2002; Figure 2) (see below).

The soils data were lumped by mass weighting the different horizons and averaging the different profiles in the catchment. Soil depth, bulk density and exchangeable cations were based on measurements (Data from SFT 1993, SFT 1994, SFT 2000, SFT 2001). The sulfate adsorption parameters and the organic acid concentrations were calibrated by comparing the slopes of observations with the model outputs. For the soil CO_2 pressure, a default value of 10 times atmospheric CO_2 content was used. Data for vegetation uptake of base cations were taken from the ICP Forest forest sites (data from Kvindesland et al. 1994, Røsberg and Stuanes 1992, Stemsrud 1988). As the model input is the net uptake, i.e. the net removal over a long time span, only the uptake stored in the tree stem was included (Table 1). The data are summarized in Table 1.

The observed surface water, or soil water, chemistry data are not used as model inputs, but as targets in the model calibration. This means that the model is run with a certain set of inputs and the model outputs compared to the observations. The model is calibrated by adjusting several parameters until modeled stream water, or soil solution, chemistry, and soil base saturation, match observed data.

Table 1. Inputs and calibrated parameters for Birkenes (BIE), Langtjern (LAE), Vikedal (VIK) Nedstrand (NED) and Storgama (STE). n.a.: not applicable.

Parameter	Unit	BIE water	BIE soil water	LAE water	LAE soil water	VIK water	NED soil water	STE water
Fixed parameters								
Discharge, annual	m	1.15	1.15	0.57	0.57	3.11	3.11	0.91
Precipitation, annual	m	1.49	1.49	0.75	0.75	2.63	2.63	1.01
Soil depth	m	0.4	0.4	0.4	0.4	0.75	0.75	0.32
Bulk density	kg m^{-3}	773	773	814	814	1036	1036	555
CEC	meq kg^{-1}	106.8	106.8	90.2	90.2	23.0	23.0	142.6
SO_4 adsorption half saturation	meq m^{-3}	100	100	100	100	50	50	100
SO_4 maximum adsorption capacity	meq kg^{-1}	0.1	0.1	1	1	7	7	0.1
pCO_2 soil	atm	0.33	0.33	0.33	0.33	0.33	0.33	0.33
pCO_2 stream	atm	0.033	n.a.	0.033	n.a.	0.070	n.a.	0.070
Deposition Ca	meq m^{-2}	13.6	13.6	4.2	4.3	22.9	110.0	4.9
Deposition Mg	meq m^{-2}	30.3	30.3	2.6	3.1	73.2	490	6.4
Deposition Na	meq m^{-2}	131.1	131.1	9.2	11.4	315.9	2119	26.1
Deposition K	meq m^{-2}	5.7	5.7	2.5	2.5	8.6	46.4	1.8
Deposition NH4	meq m^{-2}	65.8	65.8	23.4	23.5	57.0	57.0	28.0
Deposition SO4	meq m^{-2}	136.3	136.3	36.8	37.0	127.1	467.0	55.6
Deposition Cl	meq m^{-2}	156.3	156.3	9.9	12.4	370.4	2472	30.7
Deposition NO3	meq m^{-2}	71.2	71.2	22.4	22.4	55.1	55.1	33.4
Organic acids, soil	mmol m^{-3}	65	35	65	65	65	10	100
Organic acids, stream	mmol m^{-3}	10	n.a.	30	n.a.	0	n.a.	16
Ca saturation	%	3.9	3.9	3.9	3.9	3.8	6.2	4.8
Mg saturation	%	2.1	2.1	1.2	1.2	2.6	4.2	2.9
Na saturation	%	1.1	1.1	0.6	0.6	2.0	7.4	1.0
K saturation	%	2.8	2.8	2.5	2.5	2.2	2.8	2.9
Total base saturation	%	9.9	9.9	8.2	8.2	10.6	20.6	11.6
Vegetation uptake Ca	meq m^{-2}	16.6	16.6	10.3	10.3	39.1	39.1	0.0
Vegetation uptake Mg	meq m^{-2}	3.8	3.8	3.0	3.0	13.0	13.0	0.0
Vegetation uptake Na	meq m^{-2}	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Vegetation uptake K	meq m^{-2}	7.3	7.3	15.0	15.0	56.0	56.0	0.0
Vegetation uptake N soil	%	93	93	50	50	63	63	78
Vegetation uptake N aquatic	%	0	0	50	50	0	0	30
Optimized parameters								
$Al(OH)_3$ solubility constant, soil	log 10	8.3	8.3	9.0	8.6	8.1	8.5	9.0
$Al(OH)_3$ solubility constant, stream	log 10	8.6	n.a.	8.5	n.a.	10.1	n.a.	8.2
Weathering Ca	meq m^{-2}	50	32	30	20	125	75	20
Weathering Mg	meq m^{-2}	5.0	2.5	8.0	9.0	27.0	15.0	1.5
Weathering Na	meq m^{-2}	4.0	4.0	5.5	12.0	0.0	0.0	6.0
Weathering K	meq m^{-2}	4.0	4.0	14.2	14.2	62.0	62.0	0.0
Selectivity coeff. Al-Ca	log	-0.72	-0.64	0.27	-0.20	-0.53	-1.21	-1.29
Selectivity coeff. Al-Mg	log	-0.17	-0.47	0.32	0.62	-0.42	1.20	-1.99
Selectivity coeff. Al-Na	log	-1.23	-1.30	-2.30	-1.71	-2.26	-1.45	-2.62
Selectivity coeff. Al-K	log	-6.89	-7.13	-6.49	-6.34	-6.17	-5.08	-7.79
Ca saturation (pre-industrial)	%	10.9	11.9	7.2	6.7	6.0	13.0	3.8
Mg saturation (pre-industrial)	%	7.0	9.0	2.4	2.7	4.6	7.3	2.6
Na saturation (pre-industrial)	%	1.9	2.4	1.0	1.1	2.8	10.0	2.0
K saturation (pre-industrial)	%	3.0	3.3	2.8	2.8	2.6	4.0	2.2
Total base saturation (pre-industrial)	%	22.8	26.6	13.4	13.3	16.0	34.3	10.6

The concentration of nitrate was calibrated by specifying nitrate uptake in the catchment at a constant fraction of the incoming flux such that modeled water concentrations matched observations. This assumption was kept constant over time in the present calibration, as there is little information supporting the choice of other mechanisms. However, the fate of nitrogen in the catchment in the future has strong influence on the forecast results. This is illustrated with an additional forecast simulation in Section 4.2.

For surface waters, long time trends of observations are available for model calibration (Birkenes, Langtjern and Storgama since 1974, Vikedal since 1982). This limits the choices of parameter values in the model calibrations and increases confidence in the model predictions (Larssen et al. 2002). For the soil solution chemistry the available time series are shorter, and there is in general larger variation from year-to-year and less clear changes over time. Hence model calibrations are less constrained.

The base cations were calibrated by adjusting the weathering rates for the individual ions, the initial values for the pre-industrial base saturation in the soils and the aluminum dissolution constant simultaneously until the modeled water chemistry (stream/river or soil solution) and the present base saturation matched the observations. The slope of base cation concentration is calibrated by indirectly adjusting the Gaines-Thomas exchange constants through setting the initial conditions. In addition to the base saturation and aluminum dissolution, the assumed initial organic acidity of the soil solution contributes to this. The soil solution acidity was adjusted using the organic acid concentration.

There are clear differences between the model applications to soil water and surface water, as shown in Table 1. Surface waters integrate runoff from a variety of soils in the entire catchment, while soil solution data represent only the forest stand. A higher deposition is expected at the forest sites because of the filter effect of the trees. For Birkenes the same deposition was used for both calibrations (stream and forest plots) and at Langtjern only slightly different deposition was used. This reflects the fact that the forest sites are representative for the entire catchment at these two sites. In Vikedal/Nedstrand the situation is very different. Here the river catchment is largely unforested and the forest site is located relatively close to the sea compared to the average for the Vikedal river catchment. Hence the total deposition at the forest site is much larger than for the catchment (Table 1). Similarly there are differences in the calibrated parameters. For Birkenes and Langtjern the differences are in general small, while for Vikedal and Nedstrand the differences are larger.

2.4 Recovery indicators and critical limits

In this report we discuss acidification impacts on surface waters, soils and forest in relation to future recovery. The outputs are limited to chemical parameters that must be related to possible impacts on indicator organisms or ecosystem functions. For surface waters we present results for acid neutralizing capacity (ANC), which correlates well with damage to fish populations in Norway. For trout in Norwegian waters an ANC_{limit} of 20 µeq L^{-1} has been suggested (Lydersen et al. 1994, Lien et al. 1992), and more recently a variable (i.e. catchment dependent) ANC_{limit} (Henriksen and Posch 2001). A limit of 30 µeq L^{-1} was recently suggested for salmon (Kroglund et al. 2002).

For forest acidification occurs in soil and the molar ratio of base cations to aluminum is most commonly used as a chemical limit for damage. In an extensive literature review a molar ratio of 1.0 was suggested as appropriate (Cronan and Grigal 1995), however, the use of this ratio has also been widely disputed (e.g. Løkke et al. 1996). Different combinations of base cations and different forms of aluminum have been suggested and reported from different experiments. Here we use the molar ratio of $(Ca^{2+}+Mg^{2+}+K^{+})/(Al^{3+})$ as chemical indicator for possible forest impacts from acidification with a critical limit of 1.0.

2.5 Forecast scenarios

The applied historic scenario for sulfur and nitrogen as well as the forecast scenarios are based on current knowledge (Posch et al. 2002). Emission and deposition have been estimated at the EMEP grid scale. The depositions used here are averages for the four EMEP grid cells covering most of southern Norway. The different scenarios are illustrated in Figure 2. Three future deposition scenarios have been applied:

- The Oslo protocol (2^{nd} sulfur protocol)
- Current Legislation
- Maximum feasible reduction

The Oslo protocol to the CLRTAP entails only reduction in sulfur emissions relative to base year 1980. Emissions in 2000 are already below what was proposed in the protocol and hence the forecast used here actually has a higher deposition in 2000 than observed. The Oslo protocol did not include any measures for nitrogen emission reduction and, as such, nitrogen deposition is kept constant at the 1990 level from 2001 and onwards. The scenario is included as a reference, although it is outdated and hence unrealistic in its high deposition figures.

The Current Legislation scenario is a combination of the present legislation in all Europe. This mainly includes the 1999 Gothenburg protocol of the CLRTAP and the NEC (National Emission Ceilings)-directive of the EU, in addition to other national legislation implemented or agreed. For Norway the scenario gives slightly lower deposition as expected with only the Gothenburg protocol (EU 2000). This is the most relevant scenario for evaluation of the Gothenburg protocol.

We have also included a Maximum Feasible Reduction scenario, which is based on best available technology being fully implemented. This scenario entails considerable reductions of both S and N emissions compared to the Current Legislation scenario.

For the base cations deposition was assumed to remain constant at present-day levels.

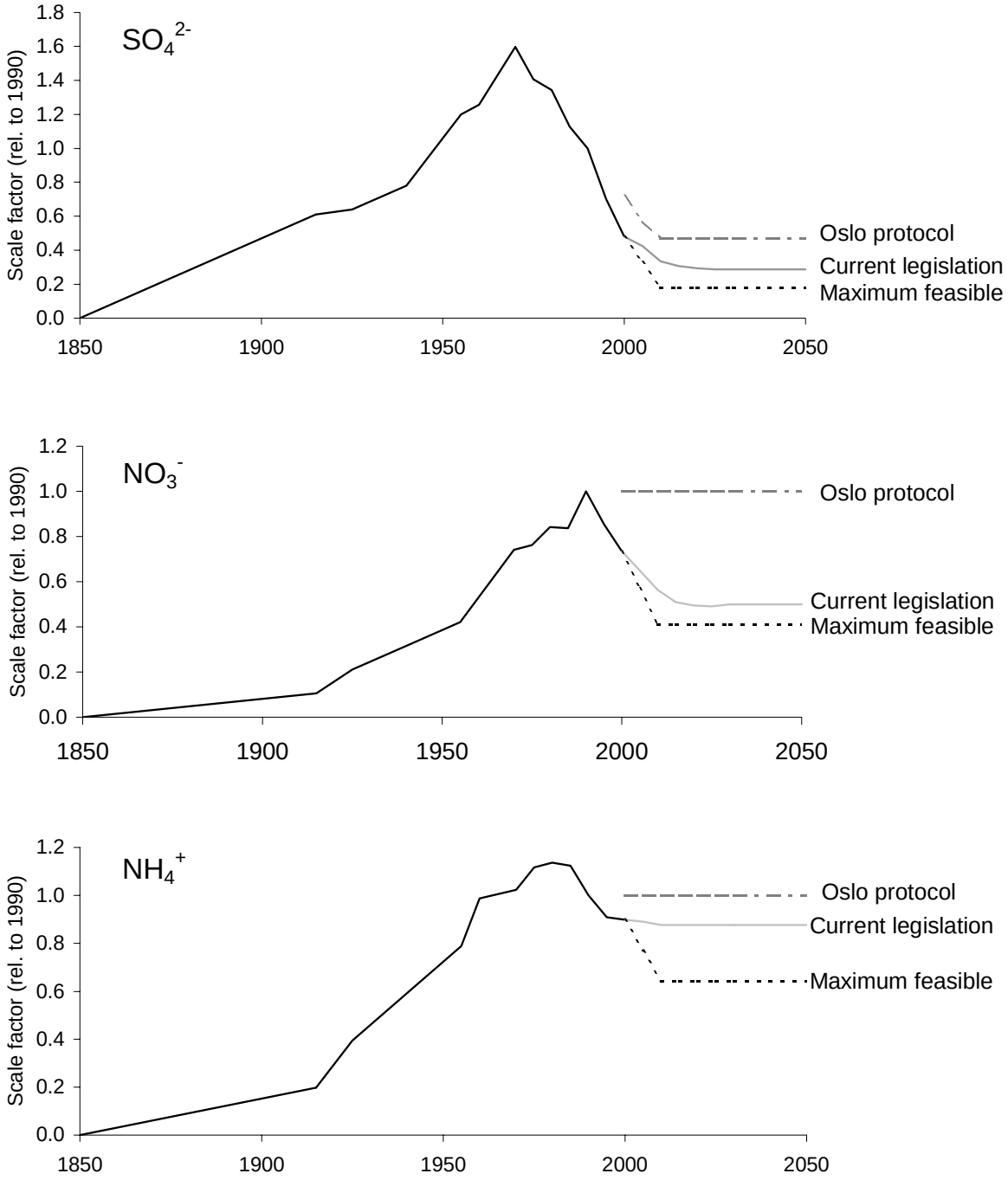

Figure 2. Scenarios for SO_4^{2-}, NO_3^- and NH_4^+ deposition used in the model expressed as scale factors relative to 1990 deposition. See text for explanation.

3. Results

3.1 Surface waters

We were able to calibrate the model well to the observed surface water chemistry data (Figure 3 - Figure 6). The sulfate deposition has decreased by about 60% over the period 1974-2001 (Aas et al. 2002). The sulfate concentration in surface water has decreased correspondingly at approximately the same rate. The decreased sulfate concentrations in the streams have been accompanied by decreased concentrations of calcium and magnesium, and to some extent also increased pH. The ANC shows considerable variations from year-to-year, but has increased in the 1990s.

Expected reductions in the sulfate concentrations under different scenarios follow the deposition changes. The response in surface water to reductions in sulfate deposition is fast because there is little sulfate adsorption in the soils. The present sulfate concentrations are already below the level predicted if the emission reductions only followed the Oslo protocol. Sulfate concentrations well below the present levels are expected after implementation of the Current Legislation scenario. Further reductions, beyond the Current Legislation, will lead to additional reductions in the surface water concentrations.

For the base cations, illustrated by calcium concentration in Figure 3 - Figure 6, the decrease observed so far is likely to level off and possibly increase in the future. The sources of base cations in the catchment are deposition, weathering and the exchangeable pool in the soil. As the ionic strength in the soil solution decreases with decreasing concentration of strong acid anions in the deposition, the release from the exchangeable pool will decrease. The predicted slow increase further into the future after the deposition has stabilized can be explained by the build up of the exchangeable base cation pool at a constant deposition ionic strength.

Increased ANC is expected as the deposition decreases in the future. At Birkenes, which has the lowest ANC among the four sites, the model calculations indicate that more than a decade is needed before positive ANC values are expected, even if the strictest scenario (i.e. the Maximum Feasible Reduction) is used. Under the Current Legislation scenario, model calculations suggest that ANC will remain negative for the next 50 years.

At Storgama the ANC is just below 0 µeq L^{-1} in 2001 and positive values are expected within a few years. The modeled long-term ANC is about 12 µeq L^{-1} for the Current Legislation scenario and 20 µeq L^{-1} for the Maximum Feasible Reduction scenario. These values are just in the range where impacts on trout are no longer expected.

At Langtjern the ANC has already recovered to a relatively high value (30-35 µeq L^{-1}). The model calibration to Langtjern is not as good as for the other surface water sites and we are not able to fully model the observed slopes in the sulfate concentration and the ANC in the recent years. The mismatch is related to the use of calendar years and not hydrological in the modeling. From the modeled present levels, further improvements are expected. Lack of model fit for the last years weakens confidence in the absolute values of the predictions.

In Vikedal we get a good fit for both sulfate and calcium and the general picture of the ANC is captured (Figure 5). The noisy curve for ANC is related to the high seasalt inputs in Vikedal, which vary considerably from year-to-year. Future improvements in the ANC are also expected in Vikedal under the Current Legislation scenario and additional improvements are expected with further reduction in deposition.

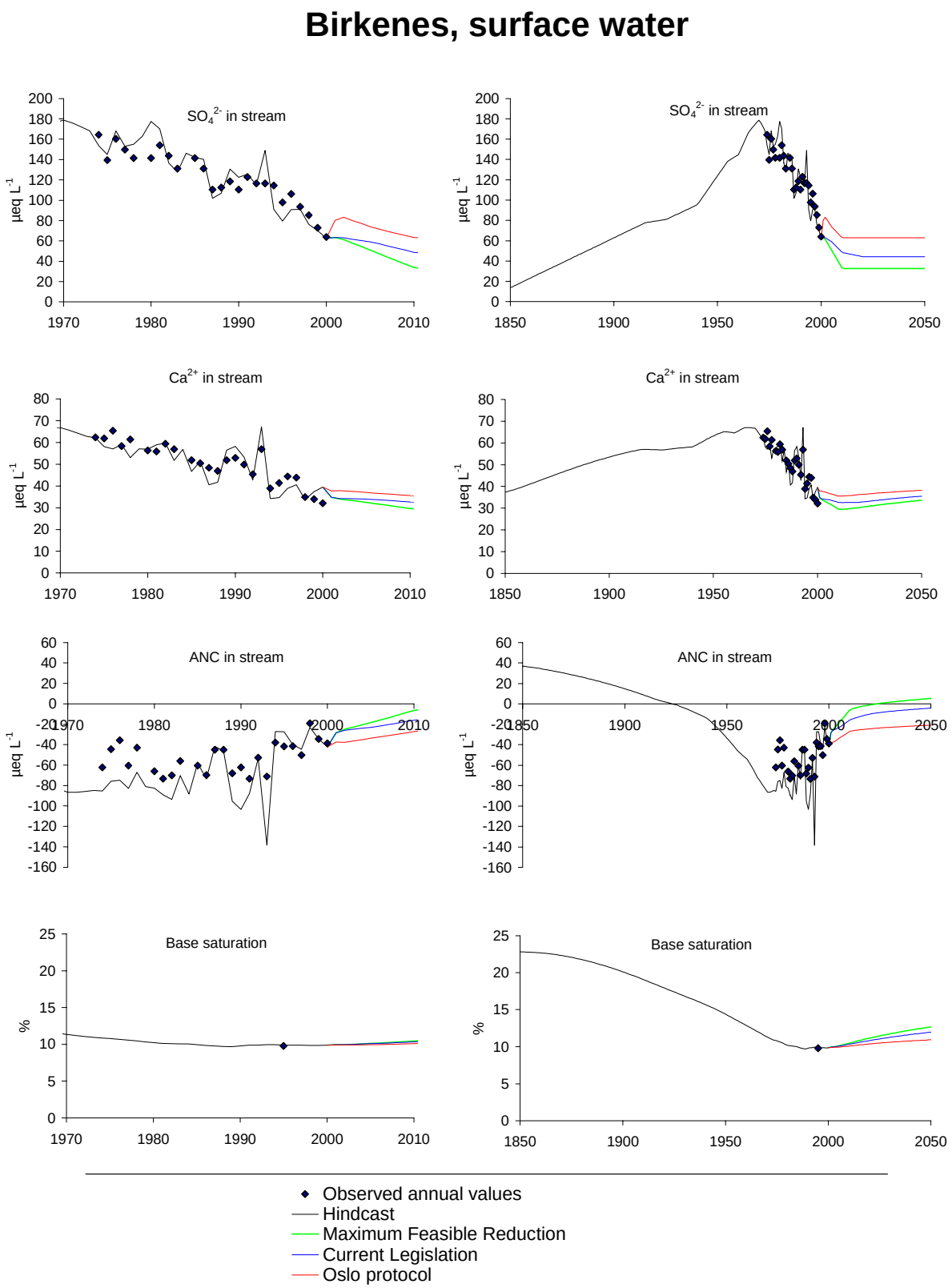

Figure 3. Modeled surface water chemistry and soil base saturation at the Birkenes calibrated catchment. Three different forecast scenarios are used, see text for description. Volume-weighted annual observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

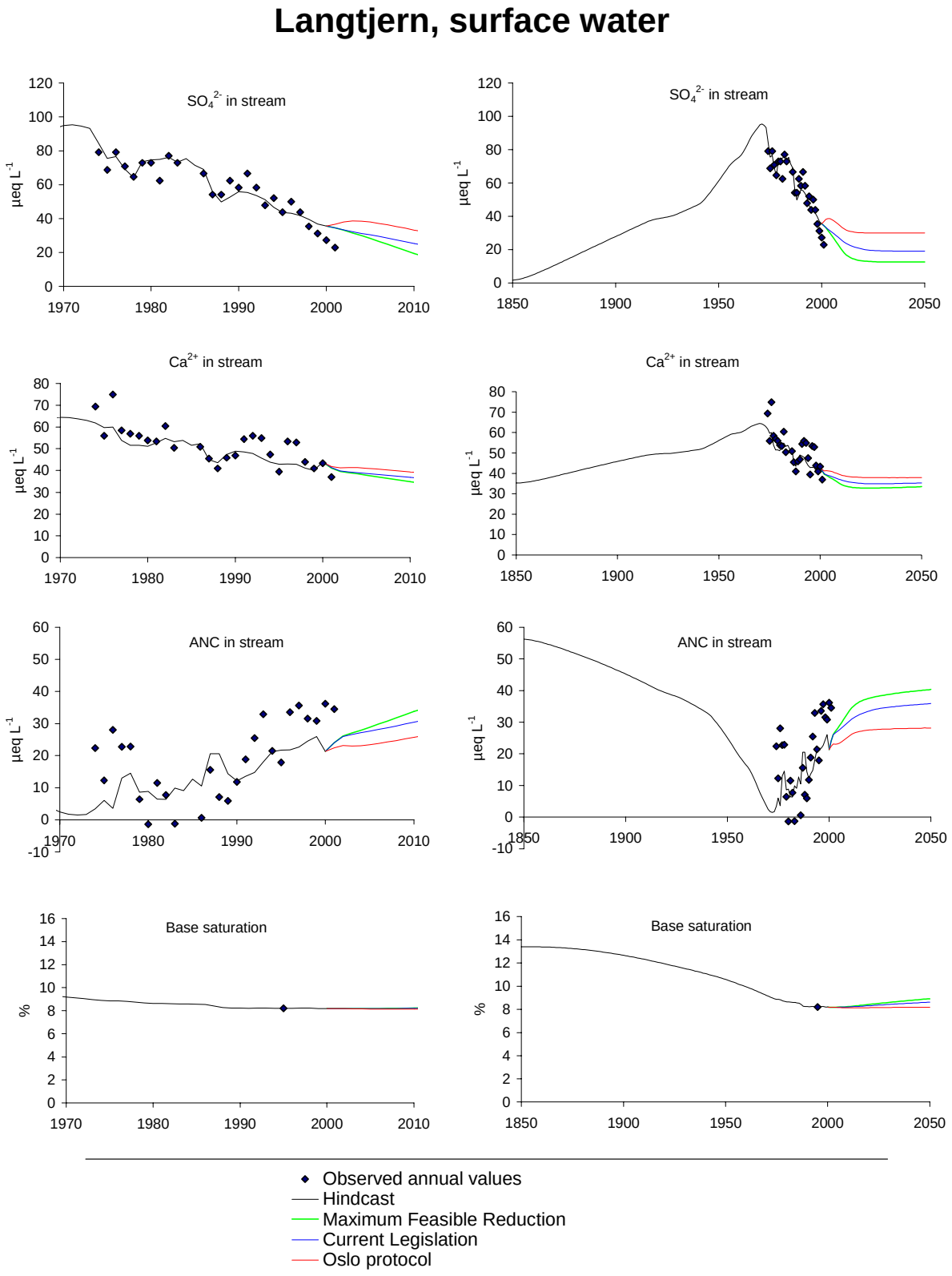

Figure 4. Modeled surface water chemistry and soil base saturation at the Langtjern calibrated catchment. Three different forecast scenarios are used, see text for description. Volume-weighted annual observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

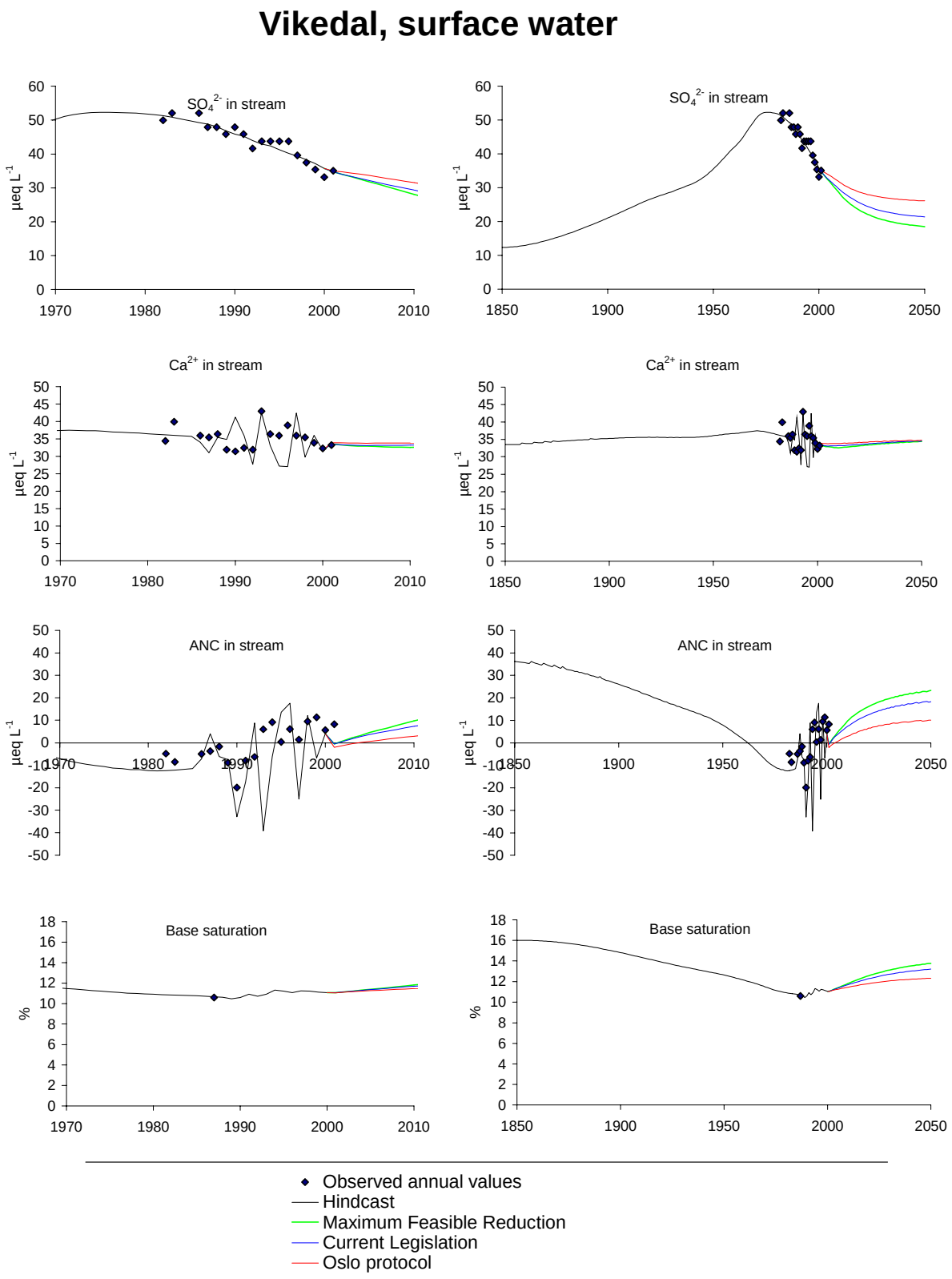

Figure 5. Modeled surface water chemistry and soil base saturation for the Vikedal river catchment. Three different forecast scenarios are used, see text for description. Volume-weighted annual observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

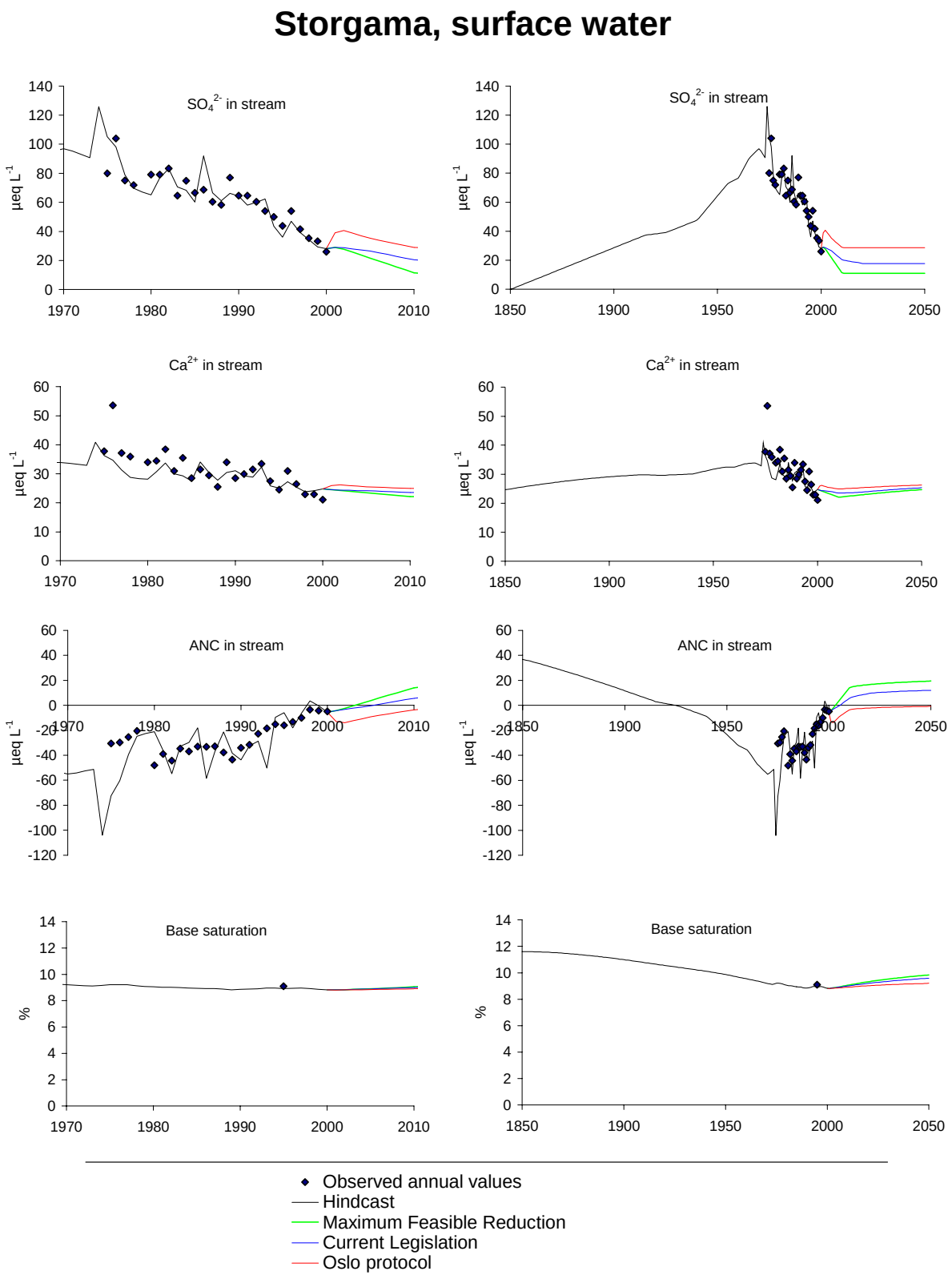

Figure 6. Modeled surface water chemistry and soil base saturation at the Storgama calibrated catchment. Three different forecast scenarios are used, see text for description. Volume-weighted annual observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

3.2 Forest soils

Soil solution data are typically highly variable over time and space and show less clear trends compared to surface waters. The model calibrations are thus less constrained. The measured calcium concentration at all three sites have one or two years in the beginning of the sampling period with very high concentrations compared to the other years (Figure 7 - Figure 9). These extreme values are assumed to be related to disturbance due to lysimeter installation (Moffat et al. 2002) and, as such, were ignored in the model calibration.

At Langtjern and Birkenes there is a tendency to a decreasing sulfate concentration trend over the approximately ten years with observations (Figure 7 - Figure 8). At the Nedstrand site there is no such trend in the observations (Figure 9). The sulfate concentration in soil solution at Nedstrand is particularly high, owing to the high deposition at this forested site. Compared to the calibration for the Vikedal river catchment, more than a six-fold increase in the seasalt deposition was needed for the forest site to match the observed concentrations in soil solution (Table 1). This is partly related to the fact that the forest site is located closer to the ocean than the average of the entire catchment and partly due to elevated deposition due to atmospheric scavenging by the trees.

The concentration of labile aluminum has been used in the calculations of the observed BC/Al molar ratio. Labile aluminum concentration was measured in soil solution only in recent years and hence there are only a few observations for the BC/Al ratio (Figure 7 - Figure 9). The observed values for the BC/Al ratio never fall below the assumed critical limit of 1.0. Birkenes has the lowest values, ranging from 1.5 to 2.1. There was one low value of 1.1 at Langtjern. At Nedstrand there is an apparent mismatch for the BC/Al ratio in the model simulation, but this is due to the unusually low Ca^{2+} concentrations measured during the few years with available labile aluminum concentrations (Figure 9). Relatively large changes from 1850 to 2000 are suggested for the BC/Al ratio. This is related to a suggested increase in the soil solution aluminum concentration as a response to acid deposition. The ratio is predicted to increase in the future and hence is not expected to fall below a critical limit for forest growth. Faster improvement is predicted for Nedstrand compared to the two other sites.

The model runs suggest that there has been soil acidification (expressed as decreased base saturation) at all sites owing to acid deposition. The largest change is at Birkenes, with a decrease in base saturation from a pre-industrial level of 26% to the present 10%. For Langtjern the change is from 13% to 8% and for Nedstrand from 34% to 20%. In the future the base cation pools are predicted to slowly build up again, but not to the pre-industrial levels.

The difference in the forecast between the three scenarios suggest that improvement in the BC/Al ratio and base saturation for the Maximum Feasible relative to the Current Legislation Scenario is about the same as seen from the Oslo Protocol to the Current Legislation.

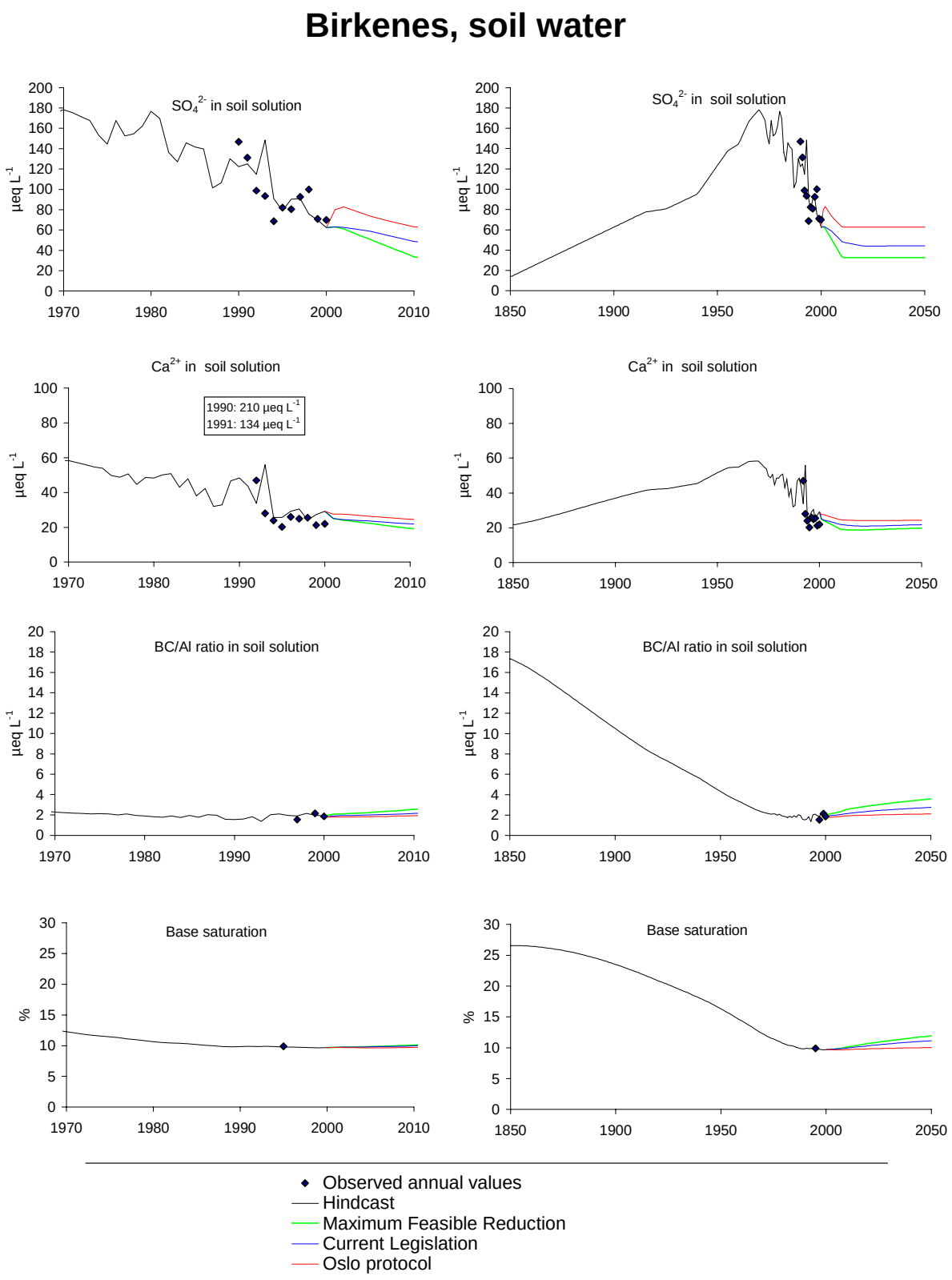

Figure 7. Modeled soil water chemistry and soil base saturation for the Birkenes forest site. Three different forecast scenarios are used, see text for description. Arithmetic annual average observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

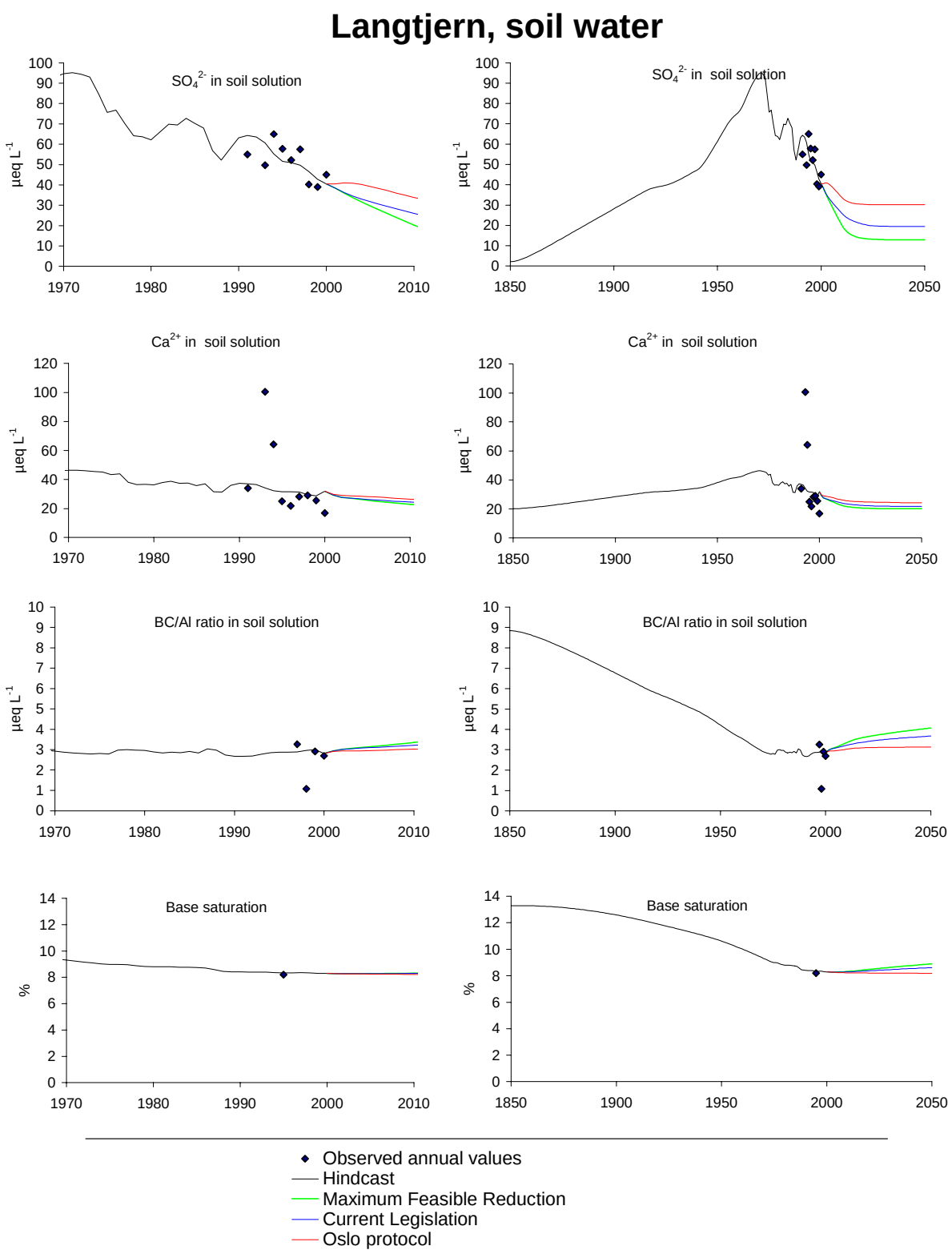

Figure 8. Modeled soil water chemistry and soil base saturation for the Langtjern forest site. Three different forecast scenarios are used, see text for description. Arithmetic annual average observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

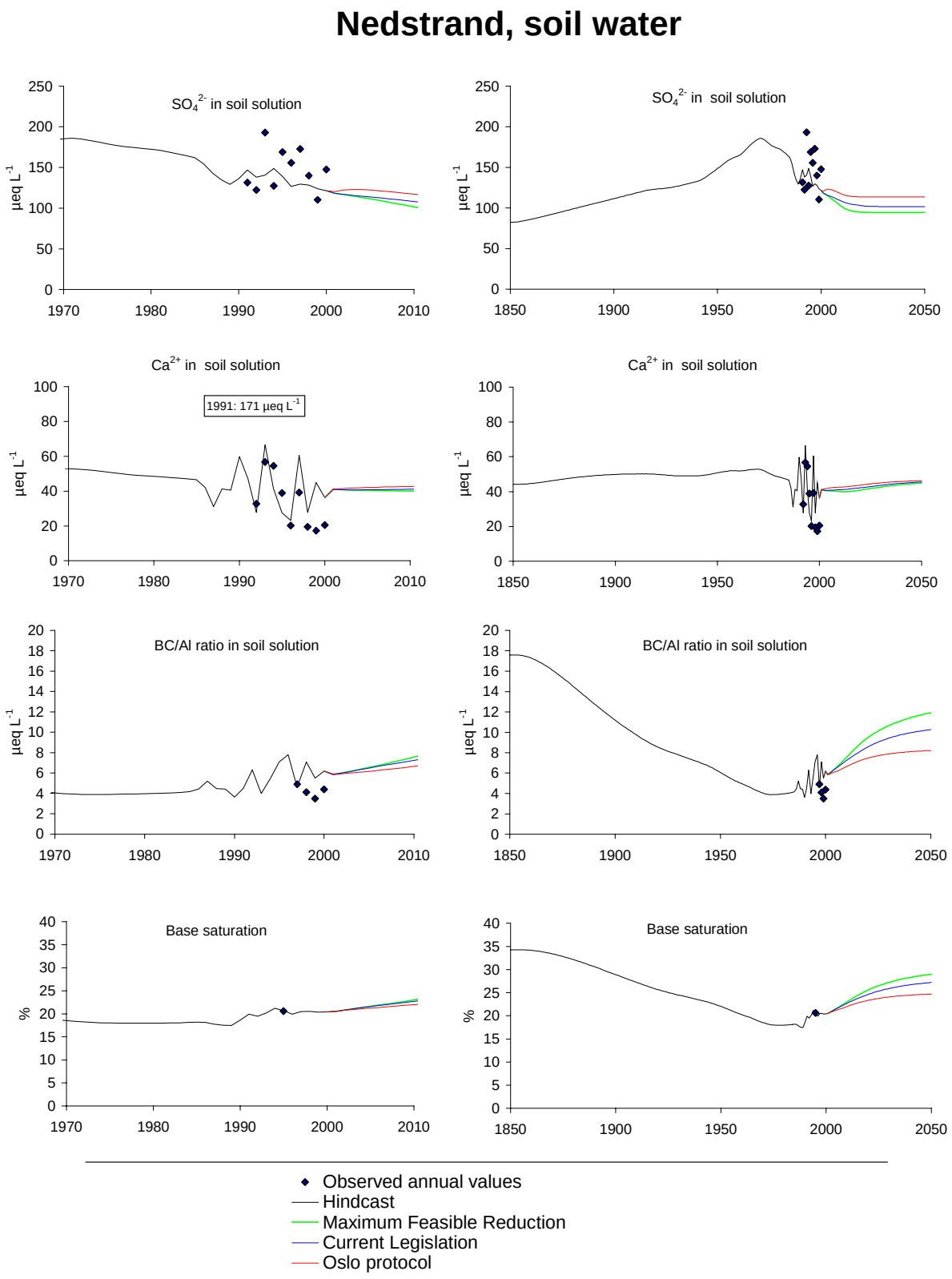

Figure 9. Modeled soil water chemistry and soil base saturation for the Nedstrand forest site. Three different forecast scenarios are used, see text for description. Arithmetic annual average observations are shown as dots. The panels to the left show the time segment around the observed data, while the panels to the right show the entire time period 1850 to 2050.

4. Discussion

4.1 Predicted responses to forecast scenarios

Considerable improvement in surface water quality has been observed during the period 1980-2000 in Norway in response to reduced acid deposition. For many lakes and rivers, however, further improvements are needed in order to get surface water quality that can support a sustainable population of trout or salmon. The predictions suggest that a considerable improvement in the surface water quality will be seen as a result of implementing Current Legislation in Europe. This illustrates the improvement from implementing the Gothenburg protocol compared to the older Oslo protocol. However, the predictions also show that the Current Legislation is not always sufficient to reach acceptable water quality.

At Birkenes and Storgama, the model predictions suggest that the Current Legislation scenario is not sufficient to promote annual average ANC values in the future above about 10 µeq L^{-1}. By applying the Maximum Feasible Reduction scenario the ANC is predicted to reach 20 µeq L^{-1} at Storgama and 0 µeq L^{-1} at Birkenes in about 2020.

At Langtjern the ANC is already relatively high. There are currently populations of stocked trout in Langtjern, but it is still uncertain whether the population is reproducing. At Langtjern the observed improvement in ANC has been faster than we are able to model, which results from the observed sulfate concentration falling faster than we can catch with model. This is related to the difference in water balance between hydrological and calendar years. The low sulfate and high ANC seen since 1997 to 2001 may be related to accidental circumstances around the combination of total ion input and precipitation amount, and we may expect to see a temporarily lower rate in the chemical recovery at Langtjern. Although further improvements can be expected if a scenario with further reduction is applied, the model predictions seems to be rather uncertain due to the discrepancy between modeled and observed concentrations. Due to the discrepancy between modeled and observed values for Langtjern, it is uncertain whether further increase in the ANC can be expected.

For Vikedal considerable improvements are predicted for the future. Rather large variations from year to year are observed due to variation in seasalt inputs, but the long term trend suggest average ANC of 20 µeq L^{-1} after implementation of the Current Legislation and an additional improvement on 5 µeq L^{-1} with implementation of the Maximum Feasible Reduction scenario. The difference between the Maximum Feasible and the Current Legislation scenarios at Vikedal is relatively small compared to the other sites. The predicted improvements are relatively large, but it will take a long time before the ANC stabilizes at the new level. This is related to the slow build up of base saturation in the soil. The ANC in the Vikedal river is predicted to increase to about 30 µeq L^{-1}, the limit for a sustainable salmon population suggested by Kroglund et al. 2002.

For all four surface waters the ANC in 2001 is at a level where reproducing fish populations are either unlikely or uncertain. The improvements predicted under the forecast scenarios suggest improvements at all sites, bringing the water chemistry to levels where reproducing fish populations are more likely than at present. Implementation of the Current Legislation is predicted to result in ANC values around the suggested limits for trout or salmon populations (except at Birkenes). The Maximum Feasible Reduction scenario result in even higher ANC values. At Birkenes, however, the improvements are still not large enough to give ANC values clearly above the suggested critical limits.

In general, the modeling results for forest soils agree with results from previous investigations on Norwegian forest soils (TVLF 2002). The results suggest that there has been soil acidification at all sites as a result of acid deposition. Of the modeled effects, perhaps the most serious and long-lasting is

the decline in base saturation. It is predicted that base saturation will not be built up again to pre-industrial levels during the next 50 years at any of the sites, not even with the Maximum Feasible Reduction Scenario. Concentrations of base cations in soil water will also remain lower than at pre-industrial levels. Recovery is predicted to be most complete at Nedstrand, possibly because of high deposition of base cations in seasalts and least complete at Birkenes, the most acidified catchment.

The risk for aluminum toxicity to forest trees appears small. Norway spruce has a far higher tolerance for aluminum than fish (Solberg et al. 2001) and modeled BC/Al ratios are not predicted to fall below the assumed critical limit of 1.0.

4.2 Future nitrogen dynamics as a confounding factor

In Europe deposition of sulfur is decreasing faster than deposition of nitrogen. The relative importance of nitrogen is thus increasing. If all deposited nitrogen leached to surface waters as nitrate, this would in many areas contribute as strongly to acidification as sulfate. However, most terrestrial systems are nitrogen deficient and consequently have a pronounced retention of inorganic nitrogen. When nitrogen deposition exceeds uptake capacities, and/or root damage caused by acidification reduces nitrogen uptake capacities, nitrate concentrations in runoff water increase (i.e. "nitrogen saturation") (Aber et al. 1989, Stoddard 1994)

The 1999 Gothenburg protocol (UNECE 1999) is based on, the precautionary principle, in which all nitrogen deposition over a certain catchment-specific threshold value is assumed to leach out in runoff water (FAB-model; Henriksen and Posch 2001). This leakage is the potential contribution of nitrogen to acidification. Today most catchments retain far more nitrogen than this hypothesis would suggest. In southern Norway the retention is typically 70%-95%, increasing from the southwest to the northeast (Larssen et al. 2001). The extent of N retention in the future, and consequently the future influence of nitrogen on surface water acidification, therefore represents a key uncertainty in future recovery from acidification.

In order to illustrate this uncertainty we ran the Current Legislation forecast for Birkenes assuming increased N leaching in the future. We assumed that until 2010, nitrogen will be retained at the same relative rate to the input as observed at present. We then assumed a linear decrease in the retention until zero in 2090. The impact of this is a linear increase in nitrate concentration and thus a decrease in ANC (Figure 10). The decrease is substantial, and prevents further recovery. In the worst case, increased nitrogen saturation may result in re-acidification even after implementing emission reductions.

4.3 Seasalt episodes as a confounding factor

Seasalt episodes are well known to have a possible negative impact on surface water chemistry (Hindar et al. 1994). The driving process is cation exchange in the soil in which sodium and magnesium in seasalt enriched rainwater exchange for aluminum. This mobilization of aluminum is most severe a few days or weeks after the seasalt event, but can also be seen on the annual average data. In the current study the MAGIC model is run with a yearly time step, based on calendar years, and hence is not ideal for studying seasalt events. The predicted ANC values in the most extreme seasalt years are predicted lower than measured by the model (seen most clearly in the model calibration for the years 1991 and 1993). This deviation is related to the use of calendar years and not hydrological years in the model. However, we have included a seasalt scenario here in order to illustrate the possible impacts.

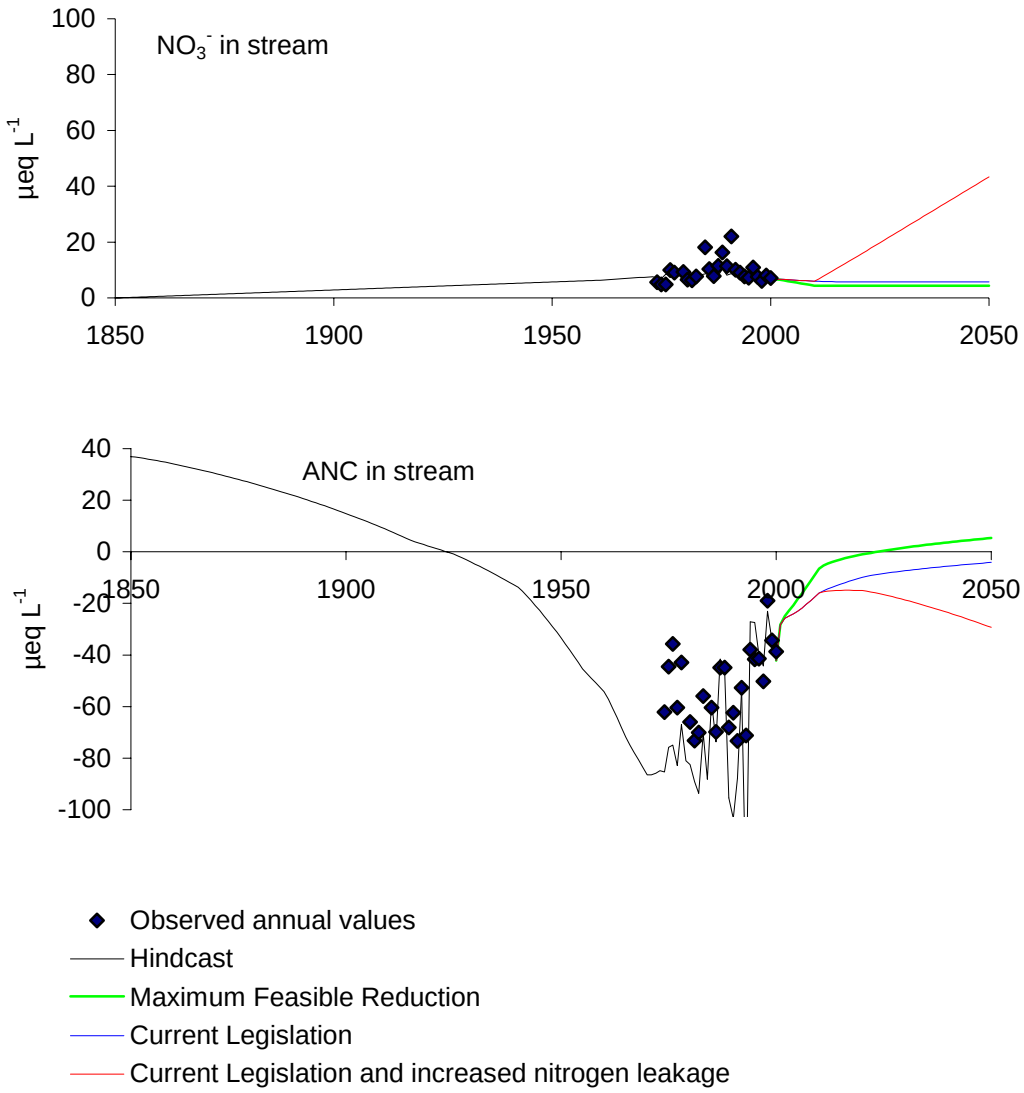

Figure 10. Illustration of the impact of future changes in catchment nitrogen retention for surface water at Birkenes assuming nitrogen leakage increasing linearly from present level to full leakage between 2010 and 2090.

Two forecast scenarios have been run. Both are based on the Current Legislation scenario deposition, but they differ in the assumed forecast scenario for seasalts. In one scenario the observed variation in seasalt deposition 1974-2001 was repeated into the future. In the other scenario we increased the observed seasalt deposition by 50%. This scenario is an extreme simplification to illustrate the possible case of increased event frequency (implying increased total seasalt deposition).

For the standard scenario, the model run suggests that the dip in ANC observed with seasalt events will be smaller as the acidification pressure decreases (Figure 11). The model run also illustrates that the large variation from year-to-year is related to the variation in seasalt inputs.

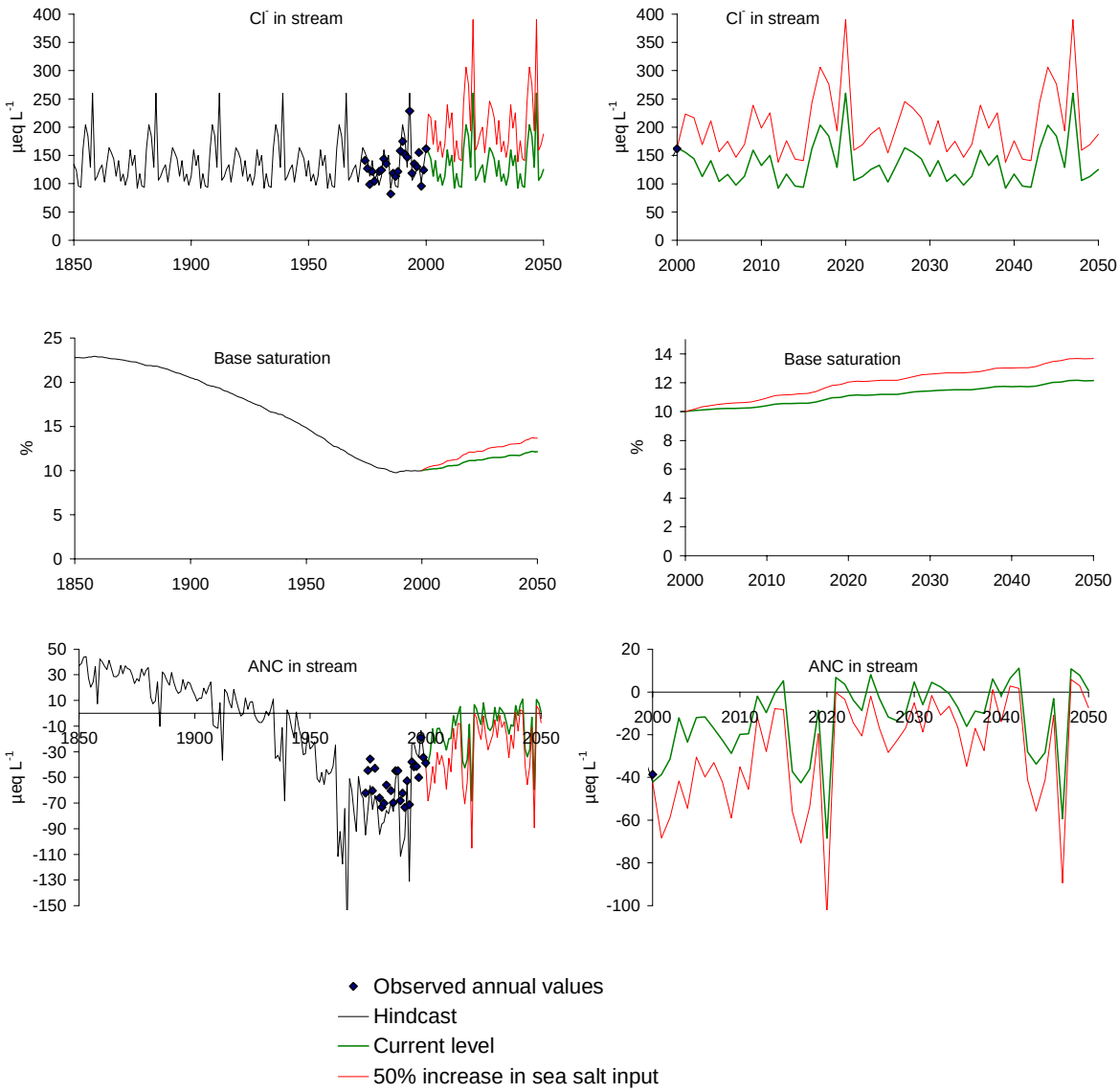

Figure 11. Illustration of the impact of seasalt variation included in the model run for Birkenes. The natural variation in seasalt deposition observed for the period 1974-2000 is repeated through the entire time series of deposition and annual runoff amount used as model inputs. Two different scenarios are run; one repeating the observed seasalt pattern in the future, another with 50% increased seasalt deposition. The panels to the left show the entire time period 1850 to 2050, while the panels to the right show the time segment from 2000 to 2050.

For the scenario with increased seasalt deposition (illustrated with Cl^- in Figure 11) soil base saturation is predicted to increase faster than for the standard scenario. The ANC is lower in the beginning, but as the base saturation builds up faster in the soil, the difference in ANC between the two scenarios decreases. Similar modeling results are reported for other sites in Scandinavia (Beier et al. 2002).

4.4 Uncertainties in predictions

The future dynamics of nitrogen and seasalt events are important sources of uncertainty in the predictions. However, there are also uncertainties related to model inputs and calibration data. The extensive data available from the ICP sites are of great value in exploring the predictive power of dynamic models and to evaluate the uncertainties in predictions.

In a separate project, data from Birkenes have been used to explore how uncertainties in predictions can be quantified. This has been reported separately (Larssen et al. 2002), but some of the results are included here to illustrate how uncertainties can be quantified and how they compare in size with the confounding factors discussed above. In a Monte Carlo setup a large number of model calibrations were conducted, giving several thousand different sets of model inputs. The model outputs were screened by stepwise addition of information from the time series of observations available. The results shown here represent the results after of stepwise filtering. The outputs can be viewed as a probability distribution of model predictions. Larssen et al. 2002) give further details on methodology, parameter selection and results.

In the uncertainty analysis an older scenario for deposition was used. The older scenario resulted in a slightly higher SO_4^{2-} concentration and lower ANC than the results presented here. In order to compare both studies, the model results for the seasalt scenario and the nitrogen scenario were scaled to match the median prediction from the uncertainty analysis.

The inclusion of the year to year variation for seasalts and runoff rate result in a variation in the modeled sulfate concentration much larger than the uncertainty study. This is because a smooth long term average is assumed when setting the uncertainty for sulfate deposition in the inputs.

The modeled calcium concentration with seasalt events is within the uncertainty band for most years, except the most extreme seasalt years. The situation is similar for the modeled ANC, the variation caused by the inclusion of seasalts is moderate compared to the uncertainty band, and only falls below the minimum line in the extreme seasalt years.

The uncertainty related to future retention of nitrogen falls within the uncertainty caused by the other factors. The predicted ANC for 2050 is about the same as the predicted minimum value from the uncertainty study.

This comparison of the magnitude of the uncertainty related to the inputs in general, the nitrogen dynamics and the seasalt deposition dynamics show they are predicted to be of similar magnitude.

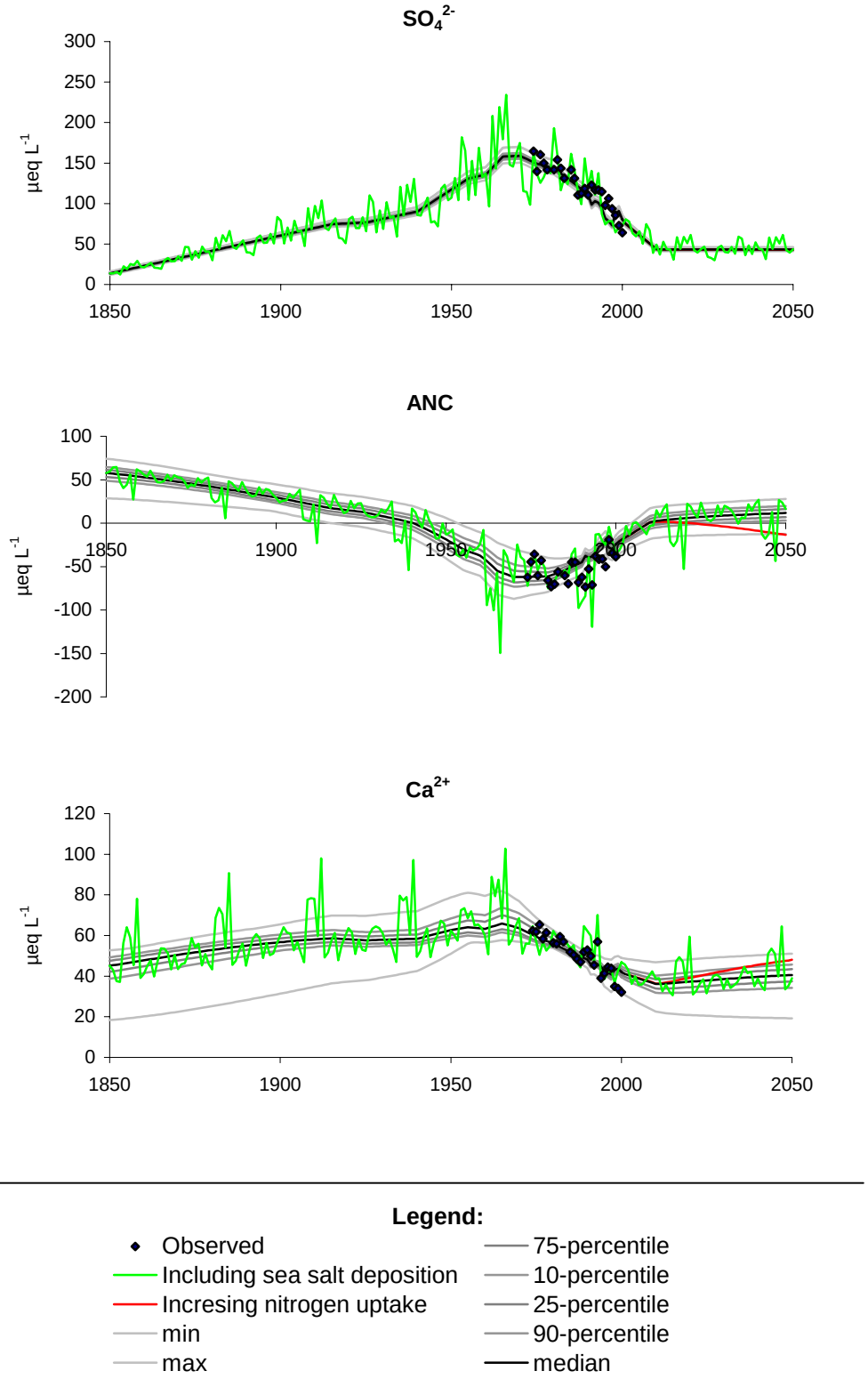

Figure 12. Illustration of uncertainties in model predictions (Larssen et al. 2002) compared with the modeled variation due to seasalt events (Figure 11) and the predicted effect of increased nitrogen deposition (Figure 10). The blue points show the annual average observations. The green line show the model results when including seasalt variation over the entire time period and the red line show the alternative nitrogen retention scenario.

4.5 Time horizons for soil base saturation recovery

Changes in the soil base saturation are slow relative to surface waters. The base cations in the exchangeable pool in the soil have been depleted over time. Natural processes contribute to soil acidification, but the deposition of strong acids is the important driver for depleting the soil base saturation. A relatively large decrease in soil base saturation was predicted for the hindcast period in the model calibration. In order to illustrate the long time needed for the soil to recover to the pre-industrial level, the MAGIC model was run with the Maximum Feasible Reduction scenario for 500 years into the future. The deposition was assumed constant at 2030 level until 2500. The base cation slowly builds up, but does not reach the pre-industrial level because the acid input is still larger than at pre-industrial time. This exercise illustrates that soil chemical recovery is very slow. In addition it shows that surface water recovery is the relevant target when discussing the need for future deposition reductions in Norway.

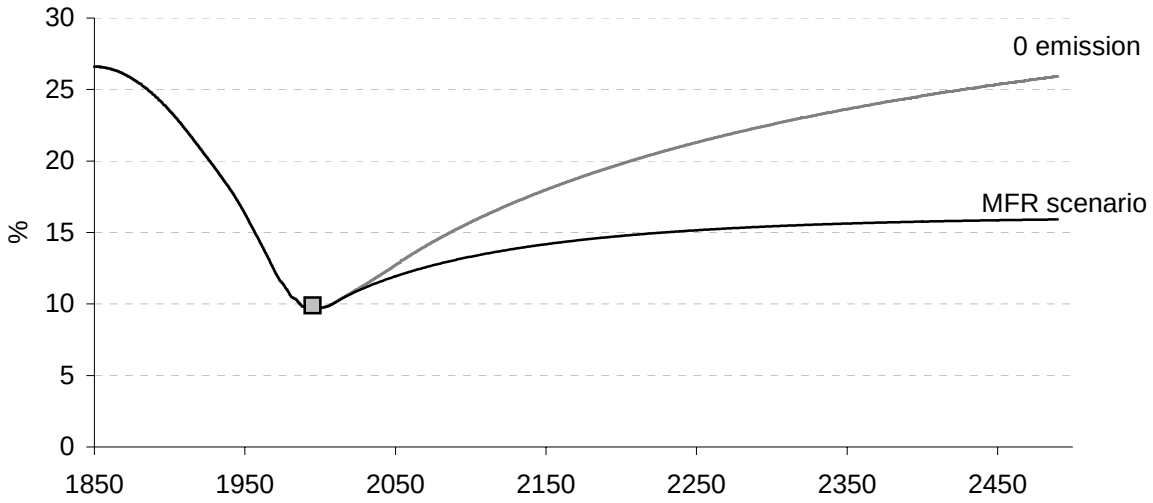

Figure 13. Long term time trend (1850-2500) in base saturation for the forest soil at Birkenes. The MFR scenario is applied. In one case the future deposition was held constant at the 2030 level for the MFR scenario. For the other case (called 0 emission in the figure) the deposition of S and N were set to 0 (except seasalts) from 2050 and onwards.

5. Conclusions

The results illustrate the following major points:

- Norwegian surface waters are considerably more sensitive to acidification than forests.
- Surface waters respond much faster to reduced acid deposition than soils
- There is still need for further reductions in acid deposition in order to reach surface water ANC levels sufficient for keeping sustainable fish populations.
- The magnitude and frequency of seasalt episodes, as well as the nitrogen dynamics in the future are important confounding factors for future predictions. The uncertainty in predictions related to these factors is of the same order of magnitude as the other uncertainties related to model inputs and calibration parameters.
- Data from comprehensive monitoring programs, such as the ICP-programs, are of great value in calibrating dynamic acidification models and applications of such models for predictions of future deposition scenarios.

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Naturens Tålegrenser

Programmet Naturens Tålegrenser ble satt igang i 1989 i regi av Miljøverndepartementet. Programmet skal blant annet gi innspill til arbeidet med Nordisk Handlingsplan mot Luftforurensninger og til pågående aktiviteter under Konvensjonen for Langtransporterte Grensoverskridende Luftforurensninger (Genevekonvensjonen). I arbeidet under Genevekonvensjonen er det vedtatt at kritiske belastningsgrenser skal legges til grunn ved utarbeidelse av nye avtaler om utslippsbegrensning av svovel, nitrogen og hydrokarboner.

En styringsgruppe i Miljøverndepartementet har det overordnete ansvar for programmet, mens ansvaret for den faglige oppfølgingen er overlatt en arbeidsgruppe bestående av representanter fra Direktoratet for naturforvaltning (DN) og Statens forurensningstilsyn (SFT).

Arbeidsgruppen har for tiden følgende sammensetning:

Tor Johannessen - SFT
Andre Kammerud - SFT
Else Løbersli - DN
Steinar Sandøy – DN

Henvendelse vedrørende programmet kan rettes til:

Direktoratet for naturforvaltning
7485 Trondheim
Tel: 73 58 05 00

eller
Statens forurensningstilsyn
Postboks 8100 Dep
0032 Oslo 1
Tel: 22 57 34 00

Naturens Tålegrenser - Oversikt over utgitte rapporter

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