Moldan, Filip; Stadmark, Johanna; Fölster, Jens; Jutterström, Sara; Futter, Martyn N.; Cosby, Bernard J.; Wright, Richard F.. 2017. **Consequences of intensive forest harvesting on the recovery of Swedish lakes from acidification and on critical load exceedances.** *Science of the Total Environment*, 603-604. 562-569. https://doi.org/10.1016/j.scitotenv.2017.06.013

Consequences of intensive forest harvesting on the recovery of Swedish lakes from acidification and on critical load exceedances

Filip Moldan[1*], Johanna Stadmark[1], Jens Fölster[2], Sara Jutterström[1], Martyn Futter[2], Bernard J. Cosby[3] and Richard F. Wright[4]

[1] IVL Swedish Environmental Research Institute, Box 530 21, 400 14 Göteborg, Sweden
[2] Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, Lennart Hjelms väg 9, 750 07 Uppsala Sweden
[3] Centre for Ecology and Hydrology, Environment Centre Wales, Deiniol Road, Bangor, LL61 6HJ, United Kingdom
[4] Norwegian Institute for Water Research, Gaustadalléen 21 NO-0349 Oslo, Norway

[*] Corresponding author, email address: filip.moldan@ivl.se

Keywords
surface waters, silviculture, MAGIC model, leaching

Abstract
Across much of the northern hemisphere, lakes are at risk of re-acidification due to incomplete recovery from historical acidification and pressures associated with more intensive forest biomass harvesting. Critical load (CL) calculations aimed at estimating the amount of pollutants an ecosystem can receive without suffering adverse consequences are dependent on these factors. Here, we present a modelling study of the potential effects of intensified forest harvesting on re-acidification of a set of 3239 Swedish lakes based on scenarios with varying intensities of forest biomass harvest and acid deposition. There is some evidence that forestry would have caused a certain level of acidification even if deposition remained at 1860 levels. We show that all plausible harvest scenarios delay recovery due to increased rates of base cation removal. Scenario results were used to estimate critical loads for the entire population of lakes in Sweden. The forestry intensity included in critical load calculations is a political decision. After scaling calculations to the national level, it was apparent that a high but plausible forest harvest intensity would lead to an increase in the area of CL exceedances and that even after significant reductions in forest harvest intensity, there would still be areas with CL exceedances. Our results show that forest harvest intensity and regional environmental change must be carefully considered in future CL calculations.

Background
In many regions of Europe and North America, there is an on-going legacy of surface water acidification related to historic acid deposition (Evans et al. 2001, Garmo et al. 2014). In the second half of 20th century much of Fennoscandia received large amounts of sulphur (S) emitted from fossil fuel combustion and industrial processes in northern and central Europe. As a consequence, soils and surface waters in southern Sweden, Norway and Finland gradually acidified. Many lakes lost fish populations (Tammi et al. 2003) and the long-term fertility of soils has been put at risk (e.g. Tamm

1976, Akselsson et al. 2006). While acid deposition is well below historical highs, modelling studies have suggested that more intensive forest harvesting for bioenergy production may slow or counteract recovery (Akselsson et al. 2006, Iwald et al. 2013). Removal of the essential base cations (BC; Ca + Mg + K) in forest biomass will reduce the buffering capacity of the catchment soils and may make surface waters more sensitive to acidification.

In Sweden, the criterion for surface water acidification is based on the estimate of change in lake (or stream) pH between reference conditions, assumed to exist in 1860 when there were only minor industrial impacts on the environment, and the present. A decrease of pH (ΔpH) of more than 0.4 units is considered indicative of unacceptable biological damage and is used for the classification of ecological status in Sweden (Naturvårdsverket 2007). This criterion is derived from empirical data for sensitive fish populations and littoral invertebrates (Fölster et al. 2007). Reference condition pH is modelled either directly with the dynamic model MAGIC (Model of Acidification of Groundwater In Catchments; Cosby et al. 1985a, b, 2001) or indirectly by comparison with a similar water body that has been modelled by MAGIC and stored in the MAGIC library (Moldan et al. 2013a, b).

The extent to which surface water pH has changed between the reference condition and the present depends on both past air pollution and land management in the catchment. The MAGIC model uses present-day observed lake (or stream) water chemistry and soil chemistry for calibration of several soil parameters such as mineral weathering and pre-industrial soil base saturation. Historical changes in acid deposition and forestry practices must be specified to reconstruct time series of water and soil chemistry between reference conditions and the present day. Credible future projections are dependent on both realistic descriptions of the past to calibrate the model and on realistic projections of the future acid deposition and land use.

The 2009 European Renewable Energy Directive requires the EU to fulfil at least 20% of its total energy needs with renewables by 2020 – to be achieved through the attainment of individual national targets. Several European countries, including Sweden, have interpreted the Directive to promote a greater reliance on bioenergy from trees. Estimates of the effects of acid deposition and current and future forest harvesting on surface water acidification are needed to ensure that more intensive forest harvest does not lead to unacceptable environmental consequences.

The Convention on Long-range Transboundary Air Pollutants (CLRTAP) is an international body that among other things seeks to reduce the emissions of acidifying air pollutants including sulphur and nitrogen (N). Protocols have been “effects- based” and aim to reduce the deposition of S and N compounds such that the critical loads (CL) to terrestrial and aquatic ecosystems are not exceeded (UNECE 2015). The CL concept is based on the idea that an ecosystem has a threshold for the amount of pollutants it can receive before suffering unacceptable damage (Nilsson and Grennfelt 1988, CLRTAP 2004). Thus, the CL concept provides a link between air pollution and effects. The CL concept makes the implicit assumption that land use is static while in reality higher BC removal rates associated with more intensive forest harvesting will leave less buffering capacity in the soils to counteract acidifying atmospheric deposition, and if included in the calculations will result in lower CLs.

Within the CLRTAP, each country can choose the method by which the critical loads are determined to best suit the national conditions. This includes decisions about the future intensity of forest harvesting and other possible land use in CL calculations. Declines in acid deposition since the peak in the 1980s means that assumptions about the intensity of forest harvesting used in CL calculations have become increasingly important. This is because BC loss from soils due to acid deposition and leaching to runoff has declined relative to BC removal associated with forest harvesting. Since the 1980s when the first critical load calculations for Sweden were made, S deposition has decreased by more than 80% while the intensity of forest harvesting, especially whole tree harvesting, has increased. The relative importance of forest harvesting for BC removal from soils has therefore become much larger and consequently, the choice of forest harvest scenarios has become more important for the outcome of CL calculations. The projected increasing intensity of forest harvesting implies increasing exceedance of critical loads at constant - or even at decreasing – acid deposition.

While future forest harvesting practices are subject to many economic, technical and environmental constraints, and thus are by no means certain, most scenarios suggest significant increases in harvest intensity (Claesson et al. 2015). Here we used five different forest harvest scenarios as inputs to the MAGIC model and calculated critical loads from these scenarios for a dataset of 5084 Swedish lakes.

Materials and methods

Lakes in this study

In Sweden, there are about 100 000 lakes larger than 1 ha (http://www.smhi.se/k-data/hydrologi/sjoar_vattendrag/sjo_SVAR_2009.pdf). A set of 3239 Swedish lakes were calibrated with MAGIC when building the 2012 version of the MAGIC library (Moldan et al. 2013a). The MAGIC library (Moldan et al. 2013b) regionalizes individual MAGIC simulations using an analogue matching procedure based on 10 parameters describing lake geographical position, surface area, measured or estimated annual discharge and observed lake water chemistry. The MAGIC library consists of two key components: a library of the existing MAGIC model runs and an analogue matching routine which selects the library lake which is most similar to an evaluation lake described by the 10 parameters. The acidification assessment modelled by MAGIC at the library lake is then assumed valid for the evaluation lake as well (http://magicbiblioteket.ivl.se/, Moldan et al. 2013b). The MAGIC library version 2012 (MAGIC library $_{2012}$) was used in this study.

Water chemistry data for the 3239 library lakes comes from three separate lake surveys. These are 163 “time series” lakes, 1625 “liming reference” lakes and 1451 “national survey” lakes (Fölster et al. 2014). The liming reference lakes and most of the time series lakes were selected because they are acid sensitive. Therefore, the 3239 modelled lakes represent a subset of Swedish lakes biased towards acid sensitive lakes.

To estimate CLs for the whole of Sweden we used the entire 5084 lakes in the national lake survey (Fölster et al. 2014). The national survey lakes are a stratified random selection such that they provide the basis for making estimates for the entire population of Swedish lakes. Stratification was based on lake size class and geographic location (Grandin 2007).

The MAGIC model

MAGIC is a lumped-parameter model of intermediate complexity, developed to predict the long-term effects of acidic deposition on soils and surface water chemistry (Cosby et al., 1985a, b, 2001). Details of the soil data aggregation and deposition calibration procedure of the MAGIC application used here are given in Moldan et al. (2013a).

Scenarios

MAGIC was used to simulate lake and catchment soil chemistry under five different future scenarios (Table 1) at each of the 3239 lakes in the MAGIC library. Four of the scenarios shared the same description of historical development of air pollution and forest harvest from 1860 to 2010 (Moldan et al. 2013a). For those 4 scenarios, MAGIC was calibrated to observed soil chemistry (year 1995) and lake water chemistry (variable years between 1995 and 2010). The historical part of the fifth scenario (Constant, cf Table 1) is hypothetical and does not lead to present-day observed catchment status since it neglects forest harvest history by assuming no change in forestry practices since 1860. The results for each lake and each scenario were evaluated by comparing simulated water and soil chemistry for the years 1860, 2010 and 2030. These years represent reference, present day and future conditions regarding surface water and soil chemistry. For future conditions, 2030 was chosen since it is sufficiently close to the present to ensure that projections of air pollution and forest harvest intensity are realistic.

The impact of forestry practices on soils and waters is simulated in MAGIC by BC and N uptake needed to support forest growth. In the results presented here, the fraction of uptake associated with forest growth which returns to the soil as litter or remains on site after thinning or harvest is considered to be fully compensated by the release of nutrients back to the soil after the organic matter is decomposed. Therefore only the net uptake is considered, i.e. the fractions of BC and N which are incorporated in the part of biomass that is removed from the catchment following harvest.

Table 1 Description of the forest harvesting intensity in the scenarios. Air pollution is considered the same in all scenarios in the hindcast, and follows current legislation (Current Legislated Emissions; CLE) in the future for all scenarios except for Maximum recovery, where air pollution from 2011 and onwards is at the level of air pollution in 1860.

<table>
<tr><th>Scenario</th><th colspan="2">Forest harvesting intensity</th></tr>
<tr><td></td><td>Historical (1860-2010)</td><td>Future (2011-2030)</td></tr>
<tr><td>Low harvest</td><td rowspan="4">Actual forest harvesting as in the MAGIC library$_{2012}$ (Moldan et al. 2013a,b)</td><td>Reference condition (1860) rates</td></tr>
<tr><td>Medium harvest</td><td>Stem harvest only at 2010 level</td></tr>
<tr><td>High harvest</td><td>Increase in harvest intensity including branches, tops and ash recycling) from 2010 to 2020 then constant to 2030. This is the scenario in the MAGIC library$_{2012}$</td></tr>
<tr><td>Maximum recovery</td><td>Reference condition (1860) rates</td></tr>
<tr><td>Constant</td><td>Reference condition (1860) rates held constant 1860-2010</td><td>Reference condition (1860) rates</td></tr>
</table>

Forest harvesting scenarios

The scenarios "Low harvest", "Medium harvest", "High harvest" and "Maximum recovery" are forced in the calibration to run through the most recently observed lake water and soil chemistry. These four harvest scenarios share the same historical uptake trajectory based on development of forestry in Sweden as described above and in Moldan et al. (2013a). All five scenarios (Table 1) have identical 1860 chemical status of lake water and catchment soils. They have the same deposition, BC uptake, climate and hydrology for the year1860. Prior to 1860, all scenarios assumed a forest harvest of "light selective cutting" at a level based on historical estimates. The "Constant" scenario was added to illustrate the chemical status of lakes and catchment soils if forest harvesting intensity had stayed the same between 1860 and 2010. Further increases in harvest intensity between 2010 and 2020 are assumed in the High harvest scenario. The increase in BC uptake in the High harvest scenario is substantial (Figure 1) due to an assumed further increase in timber production and also an increasing forest area with whole tree harvesting and associated removal of BC rich needles and branches. After 2010, the Medium harvest scenario retains 2010 stem only harvest levels, and the Low harvest scenario drops back to 1860 harvest levels. After 2020 all scenarios are kept constant up to 2030. The Low harvest and Maximum recovery scenarios use the same BC uptake but differ in their assumptions about air pollution as the Maximum recovery assumes 1860 levels of deposition while the Low harvest uses CLE (Current Legislated Emissions) values from 2011 onwards.

In Sweden the recent (2013) annual area of felling is 183 000 ha or 0.8% of the 23.2M ha productive forest area with an average of 4.3 ha per felling (Skogsstyrelsen 2014). We therefore assumed that forest harvesting is such that the forest in any lake catchment will be comprised of patches of young, medium-age and harvestable trees. Consequently the uptake fluxes used in MAGIC were set equal to the long-term average annual removal of nutrients in biomass (tree harvesting). For each lake catchment the modelled forestry scenarios are translated to differences in annual net BC and N uptake, weighted by percentage of forest in the catchment. Region and forest type specific uptake rates were calculated by the Swedish ASTA program (International and national abatement strategies for transboundary air pollution; Akselsson et al. 2006). BC net uptake (annual net accumulation in biomass) was assumed to be zero for non-forest areas. The annual average BC uptake rates for the three harvest intensities thus follow the same trajectory from 1860 to 2010 but then differ substantially (Figure 1).

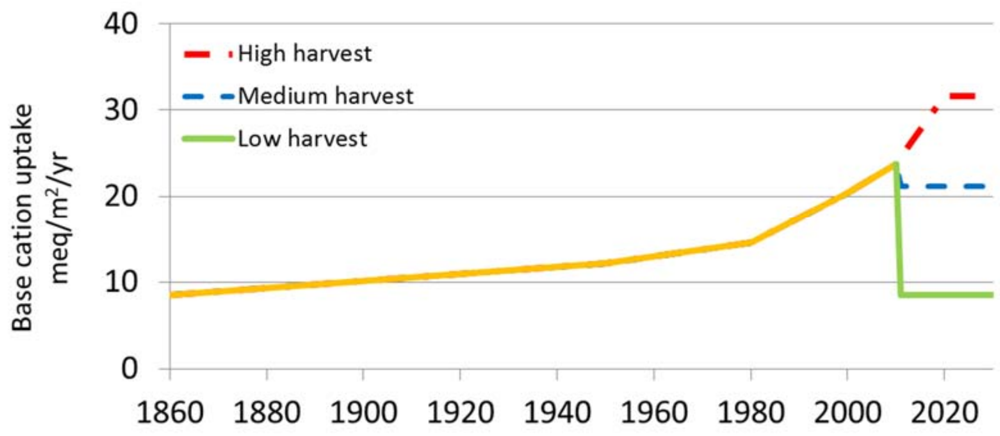

Figure 1 Average BC (Ca+Mg+K) loss through forest net uptake in catchments of 3239 modelled lakes between years 1860 and 2030. Only the part of the uptake which is removed through harvesting is accounted for. The Maximum recovery scenario follows the Low harvest scenario.

For the hindcast period (1860 – 2010), it was assumed that net annual uptake of nutrients in the forest was equal to the amount removed from the catchment at harvest divided by the mean rotation time (70 - 90 years in southern and 90 - 120 years in northern Sweden). Biomass (and thus nutrient) removal from forest land was estimated from historic data on forest area and information on harvesting of stems, branches and tops as follows: years 1860-1980 stems only were removed; years 1980-1999 stems plus an increasing percentage of branches and tops; years 2000-2010 a fraction of the nutrients removed was compensated by recycling of ash from biomass combustion. Thus due to increasing timber production and onset and increase of branches and tops (whole tree) harvest, the total annual BC removal from soils in the modelled catchments gradually increased. Over the whole hindcast period, removal of base cations due to harvesting almost tripled (Figure 1). The data used for the hindcast scenario were retrieved from the Swedish Forest Agency (www.skogsstyrelsen .se) and Statistics Sweden (www.scb.se).

Deposition of acidifying compounds

Gridded 50X50 km sulphur and nitrogen deposition rates for each decade between the late 1800s and 2010 were provided by the Coordination Centre of Effects International Cooperative Programme on Modelling & Mapping (CCE ICP M&M, a part of the Working Group on Effects (WGE) of the UNECE Convention on Long-range Transboundary Air Pollution). These data are in part based on Schöpp et al. (2003). Future deposition (for the Low, Medium and High harvest and Constant scenarios) was assumed to follow the CLE scenario, given full implementation of the revised CLRTAP Gothenburg protocol (UNECE 2013). These data were also provided by the CCE ICP M&M. Thus, any differences in MAGIC model outcomes for these scenarios are solely due to differences in forestry scenarios.

Critical load calculations

Data

CL calculations were made for 5084 lakes within the national lake survey program (Fölster et al. 2014) of which approximately 850 lakes were sampled each year from 2007 to 2012. A subset of these 5084 lakes have been calibrated with MAGIC and are included in the MAGIC library. For the lakes with no MAGIC calibrations, estimates for CL were obtained by use of the analogue matching routines in the MAGIC library. For those of the 5084 lakes that were limed, water chemistry was corrected using the Ca/Mg ratio from non-limed reference lakes either upstream within the catchment or outside the catchment within a 20-km distance (Fölster et al. 2011).

Calculations

Critical loads were calculated using the First-order Acidity Balance (FAB) model (Henriksen and Posch 2001) with modifications as described below. The procedures were the same as used for previous Swedish national CL calculations, as submitted to the CCE ICP M&M. The lake water chemical threshold, ANC_{limit}, was calculated individually for each lake to a value corresponding to a change in pH of 0.4 units from reference ANC conditions (1860) calculated by MAGIC. Delta pH was calculated from ΔANC by using the model of Hruska et al. (2003) for organic acids and assuming that total organic carbon (TOC) has been constant over time. The pressure of CO_2 was calculated from a linear relationship with TOC (Sobek et al. 2011). This lake-specific value of ANC_{limit} takes better account of individual lake properties including the large range of TOC in Swedish lakes (5 and 95 percentiles of 1.5 and 26 mg/l, respectively). It assumes, however, that there has been no change in TOC concentrations between 1860 and the present. The $\Delta pH \geq 0.4$-criterion, as opposed to fixed pH targets, also allows the appropriate treatment of naturally acidic lakes

In the FAB model calculations of N immobilisation in lake catchments were based on Gundersen et al. (1998). Excess N deposition was calculated as deposition minus forest uptake. Immobilisation was set to 100% for excess deposition up to 2 kg N ha^{-1} yr^{-1}, 50% for excess deposition between 2 and 10 kg N ha^{-1} yr^{-1} and 0% for excess deposition above 10 kg N ha^{-1} yr^{-1}. Organic N leaching was calculated from measured lake total organic nitrogen (TON) concentrations and was regarded as non-acidifying. On average about 1 kg N ha^{-1} yr^{-1} is lost as TON in runoff at the modelled lakes.

The BC leaching used in the FAB-model was based on MAGIC-calculated BC concentrations for 1860. pH_{1860} is calculated from the modelled ANC_{1860} and TOC in the lake as described above. The ANC_{limit} was calculated from pH_{1860} minus 0.4 according to the criterion for acidification. Results from the 5084 lakes were used to estimate the acidification state for all Swedish lakes larger than 1 ha using interpolation procedures described elsewhere (Posch et al. 2012; Curtis et al. 2015).

Results and Discussion

Response in soils and surface waters

Since reference conditions in 1860, there has been a general decline in soil and surface water pH in response to acid deposition and forest harvesting practices, followed by a recovery after the period of peak acid deposition in the 1980s. The simulated changes in lake water pH and ANC and in soil base saturation indicate that less intense forest harvesting results in less acidification (Table 2). The High harvest scenario projects the most widespread acidification in the future (Table 2).

Table 2 Median soil base saturation (BS), lake ANC, lake pH and % acidified lakes in the years 1860, 1980, 2010 and the projections 2030 for 3239 MAGIC calibrated lakes for scenarios Low, Medium and High harvest and the Maximum recovery (Max rec) scenario. Area (in %) in which CLs are exceeded is extrapolated to the whole country based on 5084 lakes representing all Swedish lakes.

	Years			Projections			
Median	**1860**	**1980**	**2010**	**Low 2030**	**Medium 2030**	**High 2030**	**Max rec 2030**
BS (%)	49.3	38.7	35.5	36.2	35.0	33.8	36.6
ANC (μeq/l)	190	145	159	164	161	159	168
pH	6.69	6.43	6.52	6.56	6.55	6.53	6.59
Acidified lakes (%)	0	43	29	20	24	27	13
Exceeded area (%) in Sweden	0	58.7	20.0	3.4	14.4	22.1	3.4

The four scenarios have different impacts on lake water chemistry. The results for the Medium and Low harvest show that pH continues to increase through the period 2010 - 2030, whereas under the High harvest scenario average improvement slows to a stop (Figure 2). Differences between scenarios are larger for lakes with lower pH.

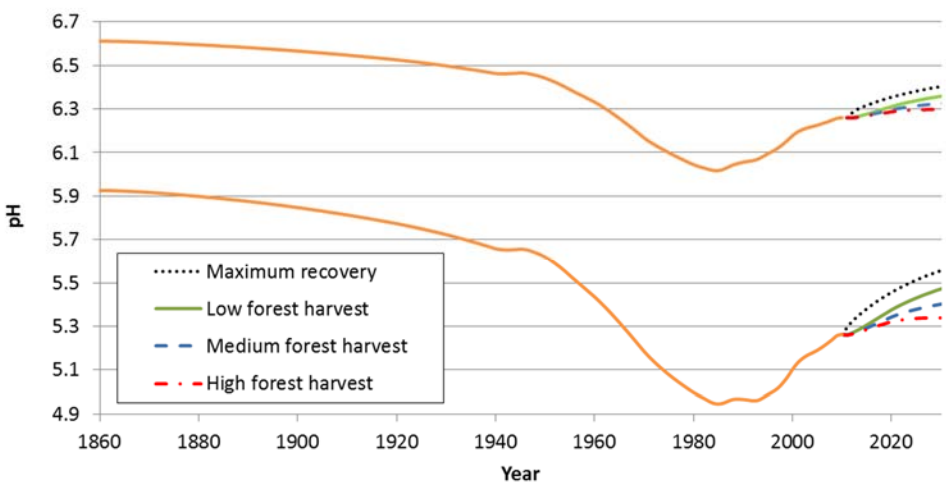

Figure 2 The median pH of 3239 lakes for the different scenarios from 1860 to 2030 modelled with MAGIC. The lakes are grouped by pH with subsets of 2161 higher pH lakes (pH>6 in 2010, upper set of lines) and 1078 low pH lakes (pH<6 in 2010, lower set of lines), respectively.

The Maximum recovery scenario resulted in the fastest average pH increase and least number of acidified lakes in 2030. Of the modelled 3239 lakes 13% were acidified (ΔpH > 0.4) in 2030 under the Max recovery scenario, and 20%, 24% and 27% under the Low, Medium and High harvest scenarios, respectively (Table 2).

The alternative scenario with constant forestry at the 1860 level (Constant, cf Table 1) suggests that without active silviculture Sweden would have had fewer problems with lake acidification despite the high levels of air pollution. Without the historical increase in forest harvesting intensity in the catchments of the 3239 modelled lakes, the median ANC in 2010 would have been 165 µeq l^{-1}, pH 6.56 and BS 38.5%. These values are all higher than the observed median values in 2010; ANC by 6 µeq l^{-1}, pH by 0.04 units and BS by 3% .

The difference in BS was larger in lakes that had catchments with low soil BS (Figure 3). The difference in lake water ANC was more evenly distributed across the ANC span of the 3239 modelled lakes (Figure 3). ANC would have been up to 40 µeq l^{-1} higher in year 2010 with constant 1860 forestry. Due to the nature of the ANC/pH relationship, the difference in pH was largest for pH values between 4.5 and 6, where the median 2010 modelled lake water pH was 0.12 pH units higher with constant 1860 forestry. For lakes outside the pH 4.5 – 6 interval, the difference was only 0.02 pH units. In 2010, 689 of the 3239 modelled lakes had a pH more than 0.4 pH units below the modelled 1860 pH, and therefore were classified as acidified. When the intensification of forestry since 1860 was not included in the model runs, the number of lakes with a pH more than 0.4 below the 1860 value decreased to 514, that is 25% fewer acidified lakes.

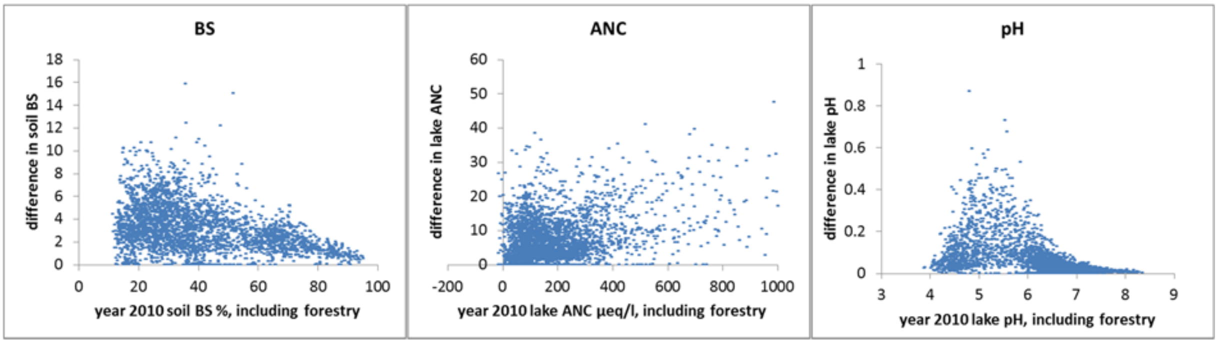

Figure 3. The difference in year 2010 between ‘Reference condition (Constant 1860 forestry)’ and ‘Actual forest harvesting’ (Table 1) for base saturation (BS), ANC (93 lakes with ANC>1000 µeq l^{-1} not shown) and pH plotted against the 2010 values for the respective parameters.

The scenario results put in perspective

Throughout the period of simulation (1860 – 2030), leaching associated with sea salt deposition and weathering is responsible for the largest proportion of BC fluxes (Figure 4). Due to acidifying deposition, resulting in cation exchange, BC leaching increased from a median value of 69 meq m^{-2} yr^{-1} in 1860 to a maximum of 89 meq m^{-2} yr^{-1} in 1984, and is projected to decrease to 65 meq m^{-2} yr^{-1} in 2030. Forest uptake increased steadily from 8 meq m^{-2} yr^{-1} in 1860 to 24 meq m^{-2} yr^{-1} in 2010. In the Medium harvest scenario it decreased after 2010 to 21 meq m^{-2} yr^{-1} and remained at that level through the rest of the modelling period. The High and Medium harvest scenarios (Tables 1 and 2) suggest that the future acidifying effect of forest harvesting will be large relative to the effects of future acid deposition.In 2010 the average non marine SO_4^{2-} deposition at the modelled lakes was 9.3 meq m^{-2} yr^{-1}; this is expected to further decrease to an average of 4.9 meq m^{-2} yr^{-1} by 2020 (Moldan et al. 2013a). The average includes areas with low acid deposition and in most cases few acidified lakes. At the 10% of the lakes which receive thehighest sulphur deposition, the sea-salt corrected sulphur deposition in 2010 was between 20 and 61 meq m^{-2} yr^{-1} and is expected to decrease to 10 – 37 meq m^{-2} yr^{-1} in 2020 (90th percentile and maximum, 3239 lakes). In comparison, the current and future forest harvest practices in most of the modelled catchments result in several times larger BC removal from soils than BC leaching to surface waters due to acid deposition. This will mean slower recovery from acidification, lack of recovery or even re-acidification in lakes depending on the mineral weathering rates, BC deposition, actual acid deposition, forestry practices and potential increases in inorganic nitrogen leaching at each site.

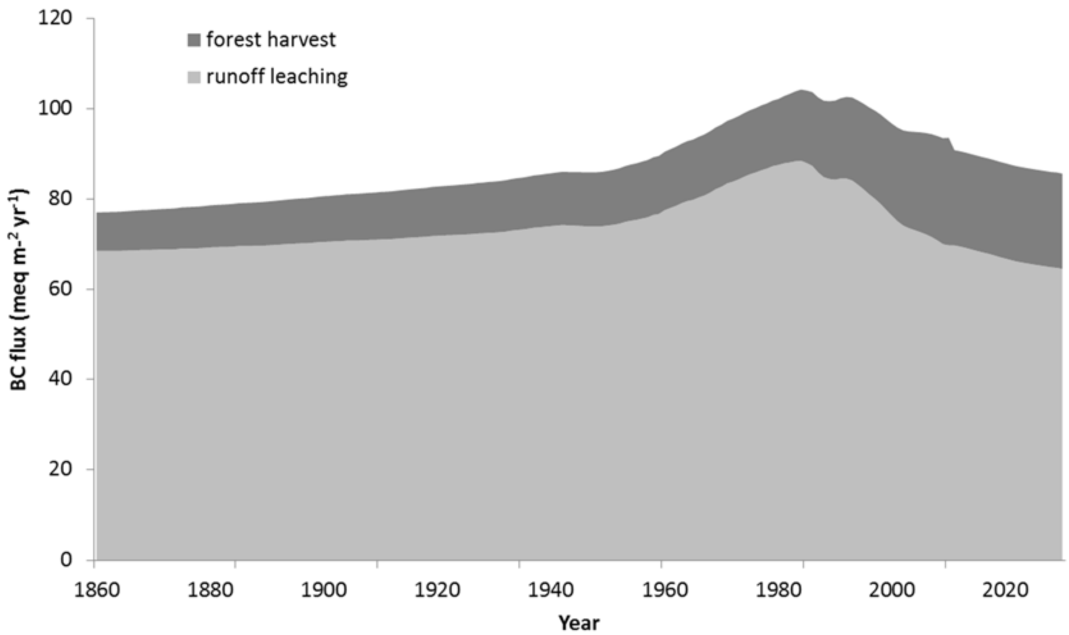

Figure 4 Median base cation (Ca+Mg+K) loss from the modelled catchments by leaching in runoff, and forest uptake (BC removed due to harvesting from the Medium harvest scenario) from 1860 to 2030 based on the 3239 MAGIC modelled lakes.

For the future the Medium harvest scenario is projected to result in decreased BC leaching to runoff. In 2030 the BC runoff leaching will be 65 meq m^{-2} yr^{-1} (median value), i.e. 4 meq m^{-2} yr^{-1} lower than in 1860. In comparison the 2030 forest harvest will remove BC from soils at an average rate of

21 meq m^{-2} yr^{-1}, i.e. 13 meq m^{-2} yr^{-1} higher than in 1860. Thus the future situation is different from the acidification period in the second half of the 20th century. Leaching of BC remains the largest BC flux, but in 2030 it is now lower than in 1860, whereas the removal of BC by forest harvest in 2030 is now more than twice as high as in 1860.

The impact of forest harvesting on exceedances of CLs in Sweden 1980-2030

There has been a large decrease in the area in which acid deposition exceeds the CL for acidification of lakes, from 58% of the country in 1980 to an expected 3-22% in 2030 for the four different scenarios (Table 2). Also the decrease in CL Average Accumulated Exceedance (AAE, Figure 5) is considerable. The 2030 AAE is lowest in Maximum recovery and Low harvest scenarios with some regional differences between the two due to the underlying geographical pattern of deposition. The year 1860 deposition of S and N, to which Maximum recovery changes after 2010, is slightly higher in northern part and lower in the southern part of Sweden relative to the expected deposition in 2030. For the Medium harvest and High harvest, the AAE gradually increases across the whole country (Figure 5). Neither the area exceeded nor AAE decreases to zero by 2030 in any of the scenarios. This means that using the current CL calculation methodology and based on realistic air pollution (much decreased since 1980) and forestry scenarios, the Swedish national environmental goal of no exceedance of CL for acidity cannot be achieved by 2030.

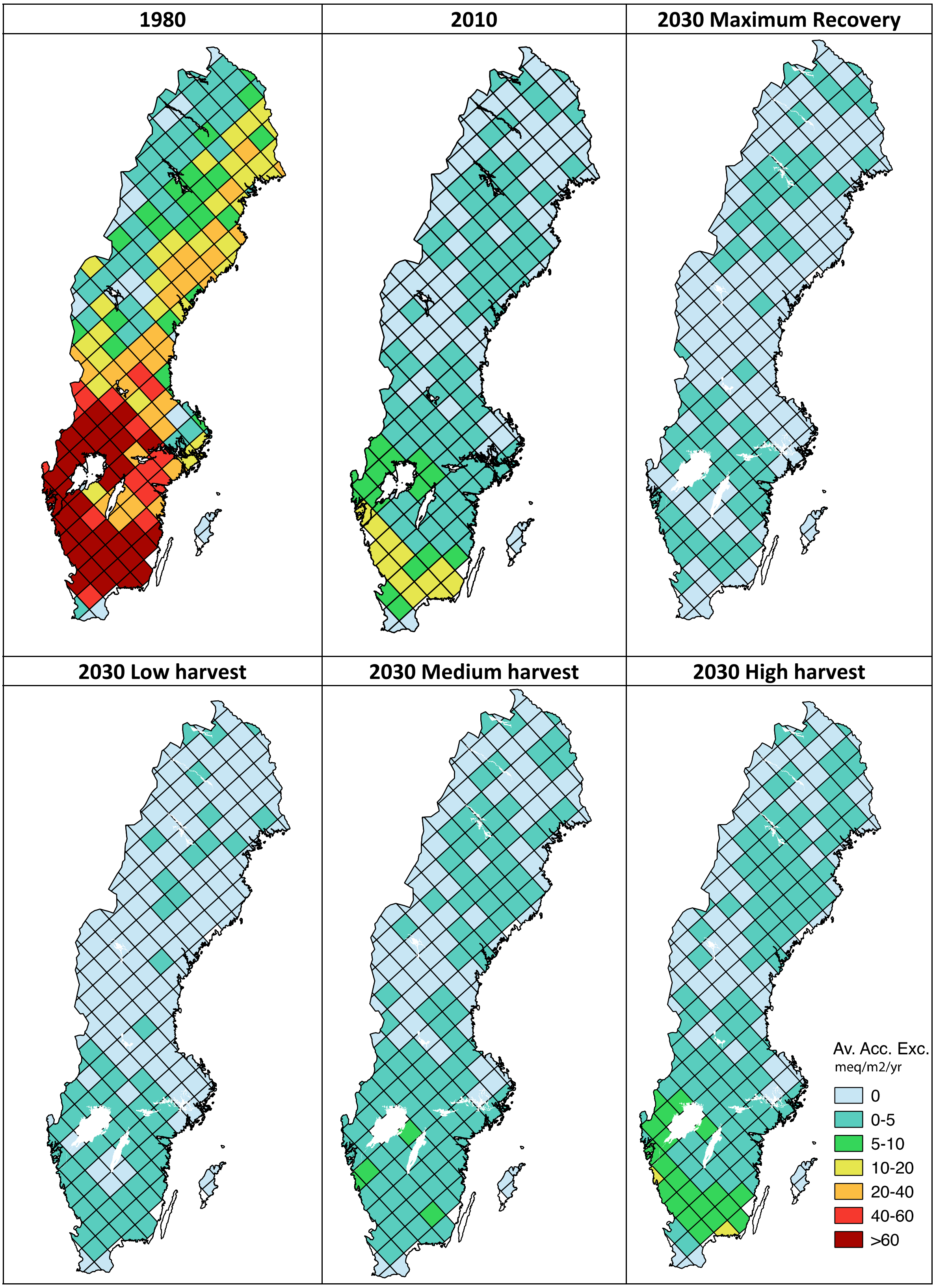

Figure 5 Exceedance of critical loads for surface waters in Sweden given as Average Accumulated Exceedance for EMEP50 squares (meq m^{-2} yr^{-1}) for the different scenarios.

The Low, Medium and High harvest scenarios follow the expected pattern where the exceeded areas increase with harvest intensity (Table 2, Figure 6). The Maximum recovery scenario gives a result nearly identical to the Low harvest scenario. This is because both use the same forest scenario, but the Maximum recovery has a slightly lower future acid deposition (5 vs. 9 meq m^{-2} yr^{-1}). The difference between the Maximum recovery and Low harvest scenarios is driven by a difference of on average only 4 meq m^{-2} yr^{-1} of deposited SO_4^{2-}, too small to have much effect on the recovery of the lakes by the year 2030.

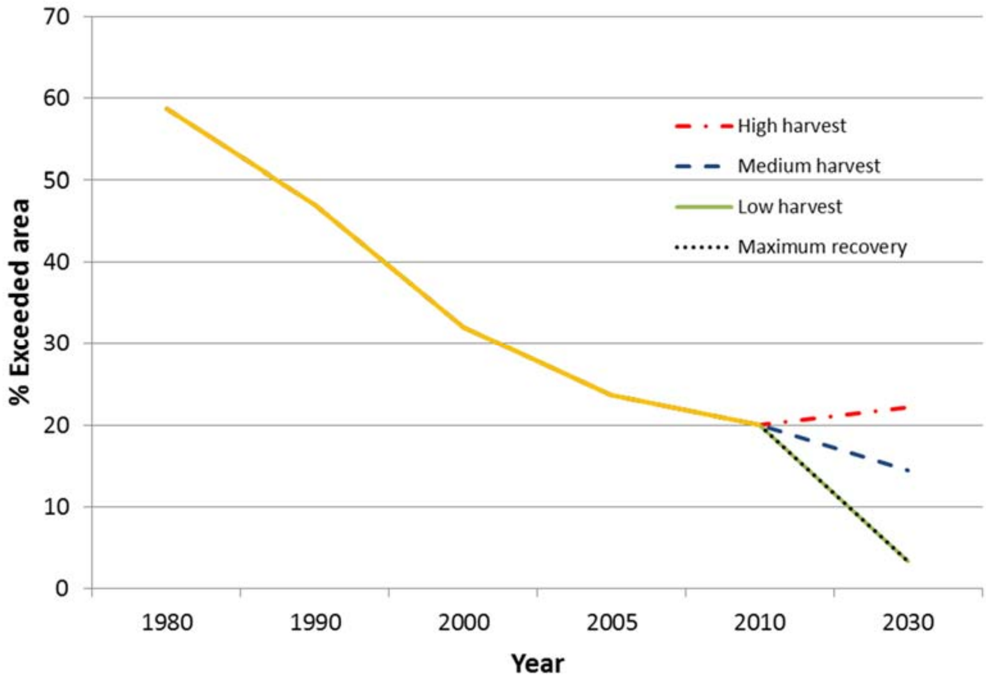

Figure 6. Changes in exceeded area after 2010 due to differences in the future forest harvesting (for the four scenarios using actual forest harvesting up to 2010 in hindcast).

The current official Swedish CL calculations are based on realistic descriptions of past forestry and a fairly conservative estimate of the future, considering stem-only harvest at the 2010 level.

Forestry scenarios are continually revised in light of e.g. changing world demand, climate and silvicultural practices. The rate of BC uptake embedded in the forestry scenarios is the major reason for the differences in exceedances. In practice, however, the acidifying potential of forest harvest versus acid deposition is still inadequately understood. Several experimental studies indicate that BC removal through forest harvest is less likely to cause soil acidification than atmospheric deposition (Thiffault et al. 2011, Brandtberg and Olsson 2012, Zetterberg et al. 2013). Uptake of BC by trees is dependent not only on demand but also on availability. Trees may actively affect weathering rates (e.g. Palviainen et al. 2012). Such factors are difficult to quantify and neither of these or other biological feedbacks is included in the MAGIC model. Thus, the exceedances associated with different intensities of harvesting should be considered maximum estimates, and the actual effect of increased forest harvesting may be lower.

Separation of the effects of forestry from those of air pollution on soil and water acidification is a difficult task. In a study of harvesting effects on soil solution chemistry, Zetterberg et al. (2013) noted that the greatest decline in calcium concentrations occurred in a well-buffered site which was unlikely to acidify. Furthermore, Akselsson et al. (2013) suggest that the on-going soil acidification is more closely related to the legacy of acid deposition than to forestry.

Conclusions

Simulations indicate that in the future more intensive forest harvesting will cause higher rates of BC removal from catchment soils as compared to the leaching of base cations to surface waters due to acidifying deposition. The intensity of forest harvest has a major impact on CL exceedance for any given deposition. Including intensive forest harvesting practices where branches, tops and stumps are removed along with stems in CL calculations will result in very low CLs which will be exceeded even at very low acidifying deposition. These results should be interpreted with caution, however, as experimental studies suggest that base cation uptake by forest may have a lower acidifying potential than acid deposition (Zetterberg et al. 2013).

Model results suggest that recovery from acidification in acid-sensitive lakes may continue, stabilize or reverse depending on the intensity of forest harvesting and acid deposition (Figure 2). Both Low and Medium harvest intensities are projected on average to lead to a continuing recovery from acidification. Recovery stops with the High harvest scenario, and in some lakes re-acidification starts. Despite the uncertainty associated with the acidifying effects of forest harvesting, the precautionary principle should be applied and the potential for re-acidification must be taken seriously. The environmental services and disservices related to bioenergy production and re-acidification of surface waters must be evaluated in sensitive catchments.

Critical loads are expected to be exceeded at 14% of Sweden by the year 2030 under the scenario of CLE deposition and Medium harvest forestry. That is a significant improvement from the year 2010 exceeded area of 20%. Under the scenario Low harvest, the CL exceeded area would decrease to 3.4%. On the other hand, under the High harvest scenario, the CL exceeded area would increase to 22% despite decreasing deposition. There is, however, a fundamental difference between acidification due to air pollution as opposed to acidification due to land use: air pollution means damage due to uncontrolled emissions, caused by often distant polluters. Acidification associated with forest harvesting could be seen as use of resources (soil, mineral weathering, and base cation deposition) that produce ecosystem services such as timber and biomass.

Acknowledgements

This work was supported by a grant from the Swedish Environmental Protection Agency (NV-08191-12) and by the MISTRA FutureForests programme (Martyn N. Futter). Forest soils and lake water chemistry data were collected in the national monitoring programs coordinated by the Swedish University of Agricultural Sciences and financed by the Swedish Environmental Protection Agency and the Swedish Agency of Marine and Water Management. Deposition data originate from EMEP (www.emep.int) and were further refined by IIASA and provided by Maximilian Posch (RIVM, the Netherlands). We are grateful to two anonymous reviewers whose comments improved this manuscript.

References

Akselsson, C., Westling, O., Sverdrup, H., Holmqvist, J., Thelin, G., Uggla, E., 2006. Impact of harvest intensity on long-term base cation budgets in Swedish forest soils. Water, Air, & Soil Pollution: Focus, 7: 201–210.

Akselsson, C., Hultberg, H., Karlsson, P. E., Pihl Karlsson, G., Hellsten, S., 2013. Acidification trends in south Swedish forest soils 1986–2008—slow recovery and high sensitivity to sea-salt episodes. Science of the Total Environment, 444: 271-287.

Brandtberg, P.-O., Olsson, B.A., 2012. Changes in the effects of whole-tree harvesting on soil chemistry during 10 years of stand development. Forest Ecology and Management, 277: 150-162.

Claesson, S., Duvemo, K., Lundström, A., Wikberg, P.-E., 2015. Skogliga konsekvensanalyser 2015 – SKA 15. Rapport 102015. Skogsstyrelsen, http://shop.skogsstyrelsen.se/shop/9098/art17/31291417-90daf1-Skogliga_konsekvensanalyser_webb.pdf (in Swedish)

CLRTAP, 2004. Manual on methodologies and criteria for modelling and mapping critical loads and levels and air pollution effects, risks and trends, ECE Convention on Long-range Transboundary Air Pollution. http://www.icpmapping.org/Mapping_Manual

Cosby, B. J., Hornberger, G. M., Galloway, J. N., Wright, R. F., 1985a. Modeling the effects of acid deposition: Assessment of a lumped parameter model of soil water and streamwater chemistry. Water Resources Research, 21(1): 51-63.

Cosby, B. J., Wright, R. F., Hornberger, G. M., Galloway, J. N., 1985b. Modeling the effects of acid deposition: Estimation of long-term water quality responses in a small forested catchment. Water Resources Research, 21(11): 1591-1601.

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. Hydrology and Earth System Sciences, 5(3): 499-517.

Curtis, C.J., Posch, M., Aherne, J., Fölster, J., Forsius, M., Larssen, T., Moldan, F., 2015. Assessment of Critical Loads of Acidity and Their Exceedances for European Lakes. In *Critical Loads and Dynamic Risk Assessments,* de Vries, W., Hettelingh, J.-P., Posch, M. (Eds.) pp. 439-462. Springer Netherlands.

Evans, C.D., Cullen, J.M., Alewell, C., Kopácek, J., Marchetto, A., Moldan, F., Prechtel, A., Rogora, M., Veselý, J., Wright, R., 2001. Recovery from acidification in European surface waters. Hydrology and Earth System Sciences, 5(3): 283-297.

Fölster, J., Andrén, C., Bishop, K., Buffam, I., Cory, N., Goedkoop, W., Holmgren, K., Johnson, R., Laudon, H., Wilander, A., 2007. A novel environmental quality criterion for acidification in Swedish lakes – An application of studies on the relationship between biota and water chemistry. Water, Air, & Soil Pollution: Focus, 7: 331-338.

Fölster, J., Johnson, R. K., Futter, M. N., Wilander, A., 2014. The Swedish monitoring of surface waters: 50 Years of adaptive monitoring. Ambio, 43: 3–18.

Fölster, J., Köhler, S., von Brömssen, C., Akselsson, C., Rönnback, P., 2011. Korrigering av vattenkemi för kalkningspåverkan - val av referenser och beräkning av osäkerheter. Rapport 2011:1, Institutionen för vatten och miljö, Sveriges Lantbruksuniversitet, . https://www.slu.se/globalassets/ew/org/inst/vom/_vatten-och-miljo_dok/miljoanalys/sjoar-och-vattendrag/keu/rapport-2011-1-camg.pdf (in Swedish)

Garmo, Ø.A., Skjelkvåle, B.L., de Wit, H.A., Colombo, L., Curtis, C., Fölster, J., Hoffmann, A., Hruška, J., Høgåsen, T., Jeffries, D.S., Keller, W.B., Krám, P., Majer, V., Monteith, D.T., Paterson, A.M., Rogora, M., Rzychon, D., Steingruber, S., Stoddard, J.L., Vuorenmaa, J., Worsztynowicz, A., 2014. Trends in surface water chemistry in acidified areas in Europe and North America from 1990 to 2008. Water, Air, & Soil Pollution, 225: 1880. doi:10.1007/s11270-014-1880-6

Grandin, U., 2007. Strategier för urval av sjöar som ska ingå i den sexåriga omdrevsinventeringen av vattenkvalitet i svenska sjöar. Rapport 2007:10. Institutionen för Miljöanalys, Sveriges Lantbruksuniversitet, http://pub.epsilon.slu.se/12399/7/grandin_u_pbasen_150626.pdf (in Swedish)

Gundersen, P., Emmett, B. A., Kjonaas, O. J., Koopmans, C. J., Tietema, A., 1998. Impact of nitrogen deposition on nitrogen cycling in forests: a synthesis of NITREX data. Forest Ecology and Management, 101(1-3): 37-55.

Henriksen, A., & Posch, M., 2001. Steady-state models for calculating critical loads of acidity for surface waters. Water, Air and Soil Pollution: Focus, 1(1-2): 375-398.

Hruska, J., Kohler, S., Laudon, H., Bishop, K., 2003. Is a universal model of organic acidity possible: Comparison of the acid/base properties of dissolved organic carbon in the boreal and temperate zones. Environmental Science & Technology, 37(9): 1726-1730.

Iwald, J., Löfgren, S., Stendahl, J., Karltun, E., 2013. Acidifying effect of removal of tree stumps and logging residues as compared to atmospheric deposition. Forest Ecology and Management, 290: 49-58.

Moldan, F., Cosby, B. J., Wright, R. F., 2013a. Modeling past and future acidification of Swedish lakes. Ambio, 42(5): 577-586.

Moldan, F., Cosby, J., Wright, R.F., Stadmark, J., 2013b. MAGIC library for Swedish lakes: evaluation of multiple calibrations. IVL Report B2104. IVL Swedish Environmental Research Institute, http://www.ivl.se/download/18.343dc99d14e8bb0f58b7660/1449742851369/B2104.pdf

Naturvårdsverket, 2007. Bedömningsgrunder för sjöar och vattendrag. Bilaga A till Handbok 2007:4. Swedish Environmental Protection Agency, http://www.naturvardsverket.se/Documents/publikationer/620-0148-3.pdf (in Swedish)

Nilsson, J., Grennfelt, P. (Eds.), 1988. Critical loads for sulphur and nitrogen. Report from a workshop held at Skokloster, Sweden, UN-ECE/Nordic Council workshop report, Report 1988:15, Nordic Council of Ministers, Copenhagen.

Palviainen, M., Starr, M., Westman, C.J., 2012. The effect of site fertility and climate on current weathering in Finnish forest soils: results of a 10-16 year study using buried crushed test rock material. Geoderma. 183-184 58-66.

Posch, M., Slootweg, J., Hettelingh, J.-P. (Eds). 2012. Modelling and mapping of atmospherically-induced ecosystem impacts in Europe, CCE Status Report 2012, Coordination Centre for Effects, RIVM, Bilthoven, The Netherlands 144 pp.

Schöpp, W., Posch, M., Mylona, S., Johansson, M., 2003. Longterm development of acid deposition (1880-2030) in sensitive freshwater regions in Europe. Hydrology and Earth System Sciences, 7: 436-446.

Skogsstyrelsen, 2014. Swedish Statistical Yearbook of Forestry 2014, Swedish Forest Agency, Sweden, https://www.skogsstyrelsen.se/globalassets/statistik/historisk-statistik/skogsstatistisk-arsbok-2010-2014/skogsstatistisk-arsbok-2014.pdf

Sobek, S., Nisell, J., Fölster, J., 2011. Predicting the volume and depth of lakes from map-derived parameters. Inland Waters, 1(3): 177-184.

Tamm, C.O., 1976. Acid precipitation: Biological effects in soil and on forest vegetation. Ambio, 5(5/6): 235-238.

Tammi, J., Appelberg, M., Beier, U., Hesthagen, T., Lappalainen, A., Rask, M., 2003. Fish status of Nordic lakes: effects of acidification, eutrophication and stocking activity on present fish species composition. Ambio, 32: 98–105.

Thiffault, E., Hannam, K.D., Paré, D., Titus, B.D., Hazlett, P.W., Maynard, D.G., Brais, S., 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests — A review. Environmental Reviews, 19: 278–309.

UNECE, 2013. 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone to the Convention on Long-range Transboundary Air Pollution, as amended on 4 May 2012, United Nation, Geneva, ECE/EB.AIR/114, 98 pp.

UNECE, 2015. Updated handbook for the 1979 Convention on Long-range Transboundary Air pollution and its protocols. United Nations, Geneva, ECE/EB.AIR/131, 255 pp.

Zetterberg, T., Olsson, B. A., Löfgren, S., von Brömssen, C., Brandtberg, P. O., 2013. The effect of harvest intensity on long-term calcium dynamics in soil and soil solution at three coniferous sites in Sweden. Forest Ecology and Management, 302: 280-294.