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# 1 **Storm Disturbances in a Swedish Forest—a case study Comparing**

## 2 **Monitoring and Modelling**

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### 8 **Abstract**

9 A Norway spruce (*Picea abies* Karst) forest site in southwest Sweden was chosen to study the  
10 effects of storm disturbances over the period 1997–2009, during which two storms, ‘Lothar’ (Dec.  
11 1999) and ‘Gudrun’ (Jan. 2005), affected the area. Monitored deposition data, soil water  
12 chemistry data and forest inventory data were compared with the predictions of an integrated  
13 ecosystem model, ForSAFE, in an effort to reveal and understand the effects of storms on  
14 acidification/recovery in forest soils. Both storms caused windthrow loss leading to increased  
15 nitrate and sulphate concentrations in soil water as a result of stimulated mineralization. *Lothar*  
16 led to increased concentrations of Na<sup>+</sup>, Mg<sup>2+</sup>, and Cl<sup>-</sup> in soil water due to sea-salt episode. No  
17 general sea-salt episode was seen following *Gudrun*, but small sea-salt episodes were observed in  
18 2007 and 2008. Each sea-salt episode caused a temporary decrease of pH, and a subsequent  
19 recovery, but overall, the soil water pH decreased from 4.54 to 3.86 after *Lothar*. Modelling  
20 suggested that the site was recovering from acidification from 1990s, and would continue to  
21 recover in future. Both modelled and monitored data showed that storm caused disturbances in  
22 the recovery; monitored data even suggested that soil acidification happened due to storm  
23 disturbances. Sea-salt episode does not increase soil acidity in the long term, and will probably  
24 decrease the soil acidity by replenishing the base saturation. The modelled data also suggested  
25 that storms with only windthrow would not have effects on soil acidification recovery in the long  
26 term, but they may influence the soil fertility by losses of base cations.

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27 *Keywords:*

28 Forest soil, Dynamic model, Storm disturbance, Sea-salt episode, Acidification recovery

29

## 30 **1. Introduction**

31 Swedish forests generally grow on nutrient-poor soils, mainly tills from nutrient-poor bedrocks such as  
32 granites and gneisses (Wastenson et al. 1990). The fertility of these soils has been further affected by acid  
33 deposition (Reuss & Johnson 1986). Furthermore, whole-tree harvesting has become more common in  
34 recent decades, leading to further removal of base cations (Iwald et al. 2013; Akselsson et al. 2007;  
35 Brandtberg & Olsson 2012). One main effect of acid deposition (S and N) on forest soils is the increased  
36 leaching of sulphate and nitrate coupled with nutrient cations such as  $Mg^{2+}$ ,  $Ca^{2+}$  and  $K^+$  causing base  
37 cations (BC:  $Ca^{2+}$ ,  $Mg^{2+}$ ,  $K^+$ ) depletion from the soil (Reuss & Johnson 1986). Base cations depletion in  
38 forest soils due to high acid deposition has been simulated in several model approaches in regional scale  
39 (Sverdrup & Rosen 1998; Akselsson et al. 2007) and in catchment scale (Navrátil et al. 2007; Hruška et al.  
40 2012; Oulehle et al. 2007). However, the N deposition in most of Sweden is generally low (Simpson et al.  
41 2011), and northern forest ecosystems are normally nitrogen (N) limited (Tamm 1991; Jonard et al. 2015).  
42 Thus, leaching of inorganic nitrogen is generally very low in Swedish forests, except for the  
43 southwesternmost part with the highest present and historical N deposition, where the nitrate concentration  
44 in soil water is often elevated (Akselsson et al. 2010).

45 S deposition to forest ecosystems has strongly decreased since the 1990s throughout Sweden, but recovery  
46 from acidification has been reported to be very slow (Pihl Karlsson et al. 2011; Akselsson et al. 2013).  
47 This has also been demonstrated through dynamic modelling (Sverdrup et al. 2005), and can be explained  
48 by the slow replacement, through weathering and deposition, of the base cations that were lost during  
49 acidification. Although N deposition has decreased in some areas of Sweden, it has generally remained  
50 constant (Pihl Karlsson et al. 2011). N accumulation in forest soils due to high loads of N deposition is  
51 also preoccupying for soil acidification recovery. In forest soils with high N content, intense N leaching  
52 episodes may occur following ecosystem disturbances such as clear-cutting (Gundersen et al. 2006;  
53 Zanchi et al. 2014), or windthrow from storms (Legout et al. 2009). Intense N leaching episodes, for  
54 which no causal explanation was found, have also been reported (van der Heijden et al. 2011). The  
55 consequences of such episodes are increased acidity and increased aluminum concentrations in soil  
56 solution, along with increased leaching of base cations and aluminum (e.g. Lundell et al. 2001). In the  
57 long term, N leaching although episodic may strongly impact the recovery from soil acidification.

58 Natural ecosystem disturbances, such as storms, droughts and fire, may also have considerable effects on  
59 the physical, chemical and biological properties of the soil, and thus the soil fertility. Storms are one of the  
60 most common types of ecosystem disturbances in Swedish forests (Schlyter et al. 2006). Storms are high-  
61 wind episodes, sometimes accompanied by heavy rain, which may lead to a reduction in canopy closure  
62 due to windthrow and stem breakage, thereby also increasing solar radiation levels in stands (Vodde et al.  
63 2011). Storms may also lead to an increase in the availability of nutrients and water (Vygodskaya et al.

64 2002; Legout et al. 2009), as a result of the reduced uptake and the increase in woody debris which will  
65 decompose and release nutrients (Vodde et al. 2011). In fact, the stimulation of mineralization of organic  
66 matter, particularly the rate of nitrification, has been reported to be an important change resulting from  
67 ecosystem disturbances (Attiwill & Adams 1993; Dahlgren & Driscoll 1994; Legout et al. 2009). This is  
68 often accompanied by the leaching of nitrate and BC (Dahlgren & Driscoll 1994).

69 Apart from the effect of wind on trees, storms with high precipitation often cause sea-salt episodes. Sea-  
70 salt episodes affect the cation exchange in soil; and is known as the 'sea-salt effect' (Hindar et al. 1995).  
71 The cations transported by sea-salt episodes, primarily  $\text{Na}^+$  and  $\text{Mg}^{2+}$ , cause the displacement of absorbed  
72 acid ions: hydrogen ( $\text{H}^+$ ), labile aluminum ( $\text{Al}^{3+}$ ) (Wright et al. 1988; Evans et al. 2001), and other BC:  
73  $\text{Ca}^{2+}$  and  $\text{K}^+$  (Hindar et al. 1994; Hindar et al. 1995). The relationship between soil acidity and the leaching  
74 of  $\text{H}^+$ ,  $\text{Al}^{3+}$  and base cations is complex, but it is often observed that the pH falls and  $\text{Al}^{3+}$  increases and  
75 sea salt ions ( $\text{Na}^+$ ,  $\text{Mg}^{2+}$ , and  $\text{Cl}^-$ ) increase shortly after sea-salt episodes (Hindar et al. 1995; Pedersen &  
76 Bille-Hansen 1995). Disturbances in acidification recovery as a result of sea-salt episodes have been  
77 observed in European and Swedish forests (Wright 2008; Akselsson et al. 2013; Laudon 2008). Wright  
78 observed that sea-salt episodes were responsible for more than one-third of the low ANC ( $\text{ANC} < -50$   
79  $\mu\text{eq l}^{-1}$ ) episodes, which were considered as strong indicator of acid episodes, at Birkenes in Norway  
80 during the period 1975–2004. Akselsson et al. (2013) and Laudon (2008) pointed out that recovery from  
81 acidification was slow at some coastal forest sites in Sweden, as these sites are susceptible to sea-salt  
82 episodes.

83 The aim of this study was to investigate the effects of storms on the acidification/recovery processes in  
84 forest soils. We hypothesized that: the high deposition of sea salt and increased rate of nutrient  
85 mineralization caused by storms slow down the recovery trend of forest soils from past acidification. For  
86 this purpose, a site in southwest Sweden that suffered the effects of the storms 'Lothar' and 'Gudrun' was  
87 studied. ForSAFE, a numerical model simulating biogeochemical cycling in forest ecosystems, was  
88 applied with monitoring data from the site. Both monitoring data and modelled data were used to study the  
89 acidification/recovery trends and the influence of the storms 'Lothar' and 'Gudrun' on these trends.  
90 Finally, ForSAFE was used to simulate the influence of frequently occurring storms in the future on soil  
91 acidification recovery.

## 92 **2. Materials & Methods**

### 93 **2.1. Site description**

94 The study site, was a  $30 \times 30$  m square plot located in a forest called Klintaskogen ( $55^\circ 62'\text{N}$ ,  $13^\circ 44'\text{E}$ ) in  
95 the region of Scania (southwest Sweden), and is part of the Swedish Throughfall Monitoring Network  
96 (SWETHRO) (Pihl Karlsson et al. 2011). It is also one of 223 intensive monitoring plots managed by the

97 Swedish Forest Agency. The average annual precipitation at Klintaskogen is 780mm and the average  
98 annual temperature is 7.2 °C (average over 1961–2010). The site is about 25 km from the west coast and  
99 about 30 km from the south coast of Sweden. The elevation of Klintaskogen is about 105 m, and it is on  
100 the southwestern slope of the horst Romeleåsen.

101 Klintaskogen is a managed Norway spruce (*Picea abies* Karst) forest that was planted on a juniperous  
102 grass-land in the 19<sup>th</sup> century. The latest final felling occurred in 1957, and the site was replanted with  
103 Norway spruce. Klintaskogen suffered from the severe wind storm *Lothar* in Dec. 1999, which caused  
104 windthrow of about 15% of the trees in the study plot. Another wind storm, *Gudrun*, also caused damage  
105 at the study site in January 2005, but tree loss (about 5% of the trees) was less severe.

106 The forest floor has a thin organic layer (3.5 cm) with 48% organic matter content (Loss on Ignition, LOI)  
107 (Table 1). The soil type is dystric regosol. The top 50 cm of the mineral soil profile is sandy and acidic  
108 with very low base saturation, and the exchange sites are mainly occupied by Al<sup>3+</sup> (Table 1). The organic  
109 matter content decreases downwards from 10.8% to 2.3%, and the pH<sub>H2O</sub> increases downwards from 4.2 to  
110 5.0, within the top 50 cm of soil (Table 1).

## 111 2.2. Monitoring data

112 The following parameters are monitored at the Klintaskogen site:

113 a) **Deposition:** Throughfall was collected monthly between 1997 and 2009 with a total of ten  
114 throughfall collectors along two sides of the study plot. Bulk rainfall was collected monthly in an  
115 open-field collector between 1997 and 2001.

116 b) **Soil solution chemistry:** Soil solution was collected at 50 cm depth using tension cup lysimeters  
117 (5 replicates), three times per year from 1997 to 2009. The 5 sample obtained were combined into  
118 a composite sample on each sampling occasion for the analysis of: sulphate (SO<sub>4</sub><sup>2+</sup>), Cl<sup>-</sup>, nitrate  
119 (NO<sub>3</sub><sup>-</sup>), ammonium (NH<sub>4</sub><sup>+</sup>), Ca<sup>2+</sup>, Mg<sup>2+</sup>, Na<sup>+</sup>, K<sup>+</sup>, Mn<sup>2+</sup>, Fe<sup>3+</sup>, inorganic Al (iAl), organic Al (oAl)  
120 and dissolved organic carbon (DOC). Composite samples with volumes <50 ml were not analyzed.  
121 The ANC was computed as follows:

$$122 \text{ANC} = [\text{Ca}^{2+}] + [\text{Mg}^{2+}] + [\text{K}^{+}] + [\text{Na}^{+}] + [\text{NH}_4^{+}] - [\text{SO}_4^{2-}] - [\text{Cl}^{-}] - [\text{NO}_3^{-}] \quad (1)$$

123 where the concentrations are in the units eq l<sup>-1</sup>.

124 c) **Forest inventories:** Inventories were carried out on three occasions (1996, 1999 and 2004) within  
125 the intensive monitoring program led by the Swedish Forest Agency, and these inventories were  
126 complemented with an inventory in 2010, using the same methodology. On each occasion, the  
127 diameter at breast height of each tree was measured, and the tree and crown heights were  
128 measured in a selected aliquot representing 20% of the trees in the plot.

129 d) **Soil characteristics:** In 2010, soil samples were collected and C, N, pH, exchangeable cations,  
 130 grain size distribution, bulk density and total element contents were determined. Total element  
 131 analysis was accomplished by ALS Scandinavia AB in Agrilab Uppsala in 2010.

132 The properties of the soil at the study site are given in Table 1, and a detailed description of the  
 133 SWETHRO network monitoring equipment and analysis protocols has been given in Pihl Karlsson et al.  
 134 (2011).

135 Table 1  
 136 Soil properties at Klintaskogen

Horizon	Layer thickness m	Stoniness <sup>a</sup> Fraction	pH		Exchangeable ions, $\mu\text{eq/g}$								CEC <sup>b</sup>	BS	Tot-C	Tot-N	Grain size <sup>c</sup>		
			H <sub>2</sub> O	KCl	Fe	Mn	Al	H	Na	K	Mg	Ca	$10^{-5}$ kmolc/kg	%	g/Kg DS	g/Kg DS	Clay	Silt	Sand
O	0.035	0	4.1	3.2	3.5	12.1	27.6	47.9	<0,1	4.6	11.7	71.1	17.9	49.0	312.9	15.0	–	–	–
A	0.06	0.15	4.2	3.3	3.3	0.7	55.4	24.6	<0,1	1.4	1.8	7.5	9.47	11.2	61.5	3.8	4	24	72
AB	0.08	0.15	4.5	3.8	0.8	0.3	41.5	11.0	<0,1	0.7	0.6	1.8	5.67	5.0	32.4	2.2	4	24	72
B	0.16	0.15	4.8	4.2	0.2	0.1	21.1	3.9	<0,1	0.4	0.1	0.9	2.66	5.4	13.8	1.1	8	20	72
C	0.2	0.15	5.0	4.2	0.1	0.1	15.4	3.2	<0,1	0.5	0.4	2.0	2.17	13.4	10.4	0.9	6	18	76

137 <sup>a</sup> The fraction of stones and boulders was assessed visually.

138 <sup>b</sup> CEC: cation exchange capacity; BS: base saturation.

139 <sup>c</sup> Grain size is expressed in percent.

## 140 2.3. Calculations

### 141 2.3.1. Tree biomass

142 The biomass of stems, branches, needles, and fine roots was estimated for each tree using Marklund  
 143 functions (Marklund 1988). The biomass of each tree compartment (wood, needles, and fine roots) of each  
 144 tree within the study plot were calculated and summed to provide an estimate of the standing biomass per  
 145 square meter, where wood referred to the sum of stems and branches.

### 146 2.3.2. Atmospheric deposition in study plot

147 The total atmospheric deposition (TD) was calculated as the sum of wet (bulk precipitation deposition)  
 148 and dry deposition. For sulphate, nitrate, Na<sup>+</sup> and Cl<sup>-</sup>, canopy exchange is assumed to be zero. Dry  
 149 deposition is thus the difference between throughfall deposition and bulk precipitation deposition. The dry  
 150 deposition of BC was computed using the Na dry deposition factor (Staelens et al. 2008):

$$151 \quad DD_x = \frac{(TFD - PD)_{Na}}{PD_{Na}} PD_x \quad (2)$$

152 where  $DD_x$  is the dry deposition of element x, TFD is the throughfall deposition, and PD is the open-field  
 153 precipitation deposition.

154 Annual deposition was computed, except for years in which values of monthly throughfall and/or bulk  
155 precipitation data were missing. The total annual deposition of BC and Na<sup>+</sup> was only available for 1998,  
156 1999 and 2001; the total annual deposition of ammonium was only available from 1996 to 2001. Missing  
157 values were estimated as follows:

- 158 1) The main source of Mg<sup>2+</sup> and Na<sup>+</sup> deposition was assumed to be sea salt. Therefore, the average  
159 values of  $\frac{TD_{Mg}}{TD_{Cl}}$  and  $\frac{TD_{Na}}{TD_{Cl}}$  in 1998, 1999 and 2001 were used to estimate missing TD of Mg<sup>2+</sup> and  
160 Na<sup>+</sup> from measured TD of Cl<sup>-</sup>.
- 161 2) Ca<sup>2+</sup> and K<sup>+</sup> deposition was assumed to be constant over the period. The average TD for the years  
162 1998, 1999 and 2001 were used to estimate missing values.
- 163 3) No PD measurements of BC or Na<sup>+</sup> were available from 2002 to 2009, so it was not possible to  
164 calculate the TD of ammonium by canopy exchange model. Therefore, the downscaled data for  
165 ammonium from the European Monitoring and Evaluation Programme (EMEP) were used.

### 166 2.3.3. *Statistical tools*

167 All statistical tests were carried out with the RStudio software (R Core Team 2013). Temporal trends in  
168 atmospheric deposition and soil solution concentration were tested with the Mann-Kendall test of the  
169 Kendall R package (McLeod 2011). Correlations between major element concentrations in soil solution  
170 were tested with the Pearson test. For both tests, p<0.05 indicates statistical significance, and p<0.001  
171 indicates strong statistical significance.

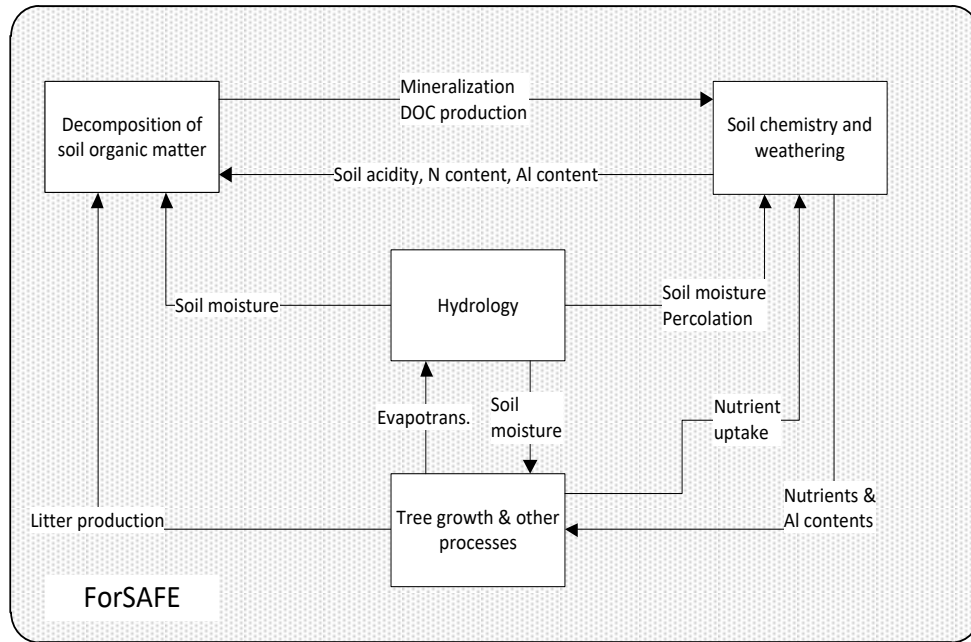
## 172 **2.4. The ForSAFE model**

173 ForSAFE is a mechanistic model designed to simulate the dynamic responses of forest ecosystems to  
174 environmental changes (Figure 1). A full description of the model can be found in the literature (Belyazid  
175 et al. 2006; Wallman et al. 2005).

176 Simulations with the ForSAFE model were run over the period 1800 to 2100, and the model was  
177 calibrated over the period 1997–2009 using the monitoring data collected from the Klintaskogen site  
178 described above. The ForSAFE inputs (soil properties, climate, deposition and silviculture scenario) are  
179 described below, and the values used are given in Table 2 and Table 3.

180





**Figure 1.** Layout of the ForSAFE model, showing the connections/interactions between the four sub-modules: decomposition, soil chemistry, hydrology and tree.

181  
182

### 183 2.4.1. Soil inputs

184 The 50 cm soil profile was divided into 5 layers in accordance with the soil measurements (Table 2).  
185 Exposed surface areas of minerals were calculated using bulk density and grain size (Warfvinge &  
186 Sverdrup 1995; Sverdrup & Warfvinge 1993). The partial pressure of CO<sub>2</sub> (pCO<sub>2</sub>) and the aluminium  
187 solubility (K<sub>gibb</sub>) were derived from Warfvinge & Sverdrup (1993) and Sverdrup & Warfvinge (1995).  
188 Root fractions were derived for each layer based on the empirical data by Rosengren & Stjernquist (2004).  
189 Mineralogical regions from Warfvinge & Sverdrup (1995) were used to assess the potential mineral  
190 occurrence in the geographical area. Based on this, and the total element analysis, soil mineral  
191 composition was determined using A2M model (Posch & Kurz 2007) (Table 3).

192 The soil field saturation points for each soil layer were calculated as follows:

$$193 \quad FS = 1 - \frac{BD \cdot (1 - HML) \cdot LOI}{\rho_{organic}} - \frac{BD \cdot (1 - LOI)}{\rho_{particle}} \quad (3)$$

194 where FS is the field saturation (m<sup>3</sup><sub>water</sub> m<sup>-3</sup><sub>soil</sub>), BD is the bulk density (kg m<sup>-3</sup>), HML is the heat mass loss  
195 from 40 °C to 105 °C (fraction of weight), LOI is the loss on ignition (fraction of weight), ρ<sub>organic</sub> is the  
196 density of organic matter, assumed to be 600 kg m<sup>-3</sup>, and ρ<sub>particle</sub> is the density of sand particles, assumed to  
197 be 2670 kg m<sup>-3</sup> (Milsom 2003). The field capacity (FC), wilting point (WP) were calculated using the

198 method described by Balland et al. (2008). The cation exchange capacity and base saturation were  
199 calculated from measured soil properties in Table 1.

#### 200 2.4.2. *Climate inputs*

201 The climate input data included precipitation, maximum, minimum & average temperature, atmospheric  
202 CO<sub>2</sub> concentration and radiation for the period 1800–2100. A weather station belonging to the Swedish  
203 Meteorological and Hydrological Institute (SMHI) situated about 12km southwest of the Klintaskogen site  
204 measured daily precipitation and temperature for the period 1961 to 2010 period. Precipitation and  
205 temperature data for the period 2010–2100 were derived from the global climate model HADLEY (Johns  
206 et al. 2006) follows the SRESA2 emission story-line of the Intergovernmental Panel on Climate Change  
207 (Nakicenovic et al. 2000). For the period 1900–1960, data were estimated by repeating the measured  
208 values for 1960–1990 twice. For the period 1800–1900, data were estimated by repeating the measured  
209 climate data for the year 1960. The CO<sub>2</sub> concentrations from 1970 to 2100 were derived from the SRESA2  
210 scenario. The CO<sub>2</sub> concentration in 1800 was obtained by extrapolation, assuming that it remained  
211 constant before 1970. Solar radiation data for 1960–2100 were derived from a study performed by the  
212 Regional Climate Group at the University of Gothenburg (David Rayner, personal communication). Solar  
213 radiation data were extrapolated back to 1800 using the same scheme for precipitation and temperature  
214 data. To avoid the need to calibrate the modeled interception of precipitation, the precipitation data were  
215 further scaled to give the throughfall precipitation using the monitored throughfall precipitation factor:

$$216 \quad TFprec_x = \frac{TFprec_{1997-2009}}{Prec_{1997-2009}} Prec_x \quad (4)$$

217 where  $TFprec_x$  is the throughfall precipitation for month  $x$ ,  $Prec_x$  is the precipitation during month  $x$ ,  
218  $TFprec_{1997-2009}$  is the total throughfall precipitation monitored by the SWETHRO network from 1997 to  
219 2009, and  $Prec_{1997-2009}$  is the total precipitation obtained from the SMHI station from 1997 to 2009.

#### 220 2.4.3. *Deposition inputs*

221 The deposition input data included the atmospheric deposition of sulphate, nitrate, and ammonium, Ca<sup>2+</sup>,  
222 Mg<sup>2+</sup>, K<sup>+</sup>, Cl<sup>-</sup> and Na<sup>+</sup> in intervals of one year. The Current Legislation (CLE) scenario and the  
223 Gothenburg protocol for the UN Convention on Long-Range Transboundary Air Pollution provided by the  
224 EMEP on a 50 × 50 km grid was used to estimate atmospheric deposition data during the period 1800–  
225 2100. The EMEP data include deposition of sulphate, nitrate and ammonium from 1900 to 2100. EMEP  
226 data were downscaled by the average ratio of the measured deposition to the EMEP deposition (1997–  
227 2009) for sulphate, nitrate and ammonium. Atmospheric deposition was assumed to be constant and equal  
228 to the deposition in 1900 during the period 1800–1900. The deposition of Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>, Cl<sup>-</sup> and Na<sup>+</sup> was  
229 assumed to be constant over the simulation period (1800–2100) and equal to the average measured

230 deposition for each element. The yearly deposition input was converted to monthly amounts in ForSAFE  
231 assuming that the monthly deposition is proportional to the monthly amount of precipitation.

#### 232 2.4.4. Storm disturbance and the future storm disturbances scenario

233 Windthrow caused by the storms was implemented in ForSAFE by simulating thinning at the end of 1999  
234 to model the effects of *Lothar*, reducing the standing biomass by 15%, and thinning in 2005 to account for  
235 *Gudrun*, reducing the standing biomass by 5%. The stimulation of organic matter mineralization in soils  
236 was implemented in the model by reducing the nutrient immobilization in microbial biomass after each  
237 storm. It was assumed that the reduction was greatest after storm, and the initial immobilization level was  
238 recovered after three years.

239 A scenario involving future storm disturbances (FSS) was included in the study for comparison with the  
240 baseline scenario (BLS). In the FSS scenario, it was assumed that a storm would occur every 20 years  
241 from 2005 onwards, causing 10% windthrow loss and stimulating organic matter mineralization (as  
242 described above) at each storm occurrence. All the storms are assumed not to be accompanied with sea-  
243 salt episodes. The difference in soil nutrient input-output budgets between the BLS and the FSS was  
244 determined using the model. No silvicultural thinning or harvesting was included in either scenario after  
245 the clear-cut in 1957 in order that only effects of storm are identified.

#### 246 2.5. Nutrient input-output budget calculation

247 Nitrogen and base cations input-output budgets were calculated over the period 1970–2070 for the two  
248 scenarios using the methodology applied in many other studies (Sverdrup et al. 2006; Akselsson et al.  
249 2007; van der Heijden et al. 2013). Nutrient budgets were calculated as the difference between soil inputs  
250 and outputs. At the forest plot scale, nutrient budgets can be adapted from Ranger & Turpault (1999):

$$251 \quad B(N) = AD(N) + F(N) - U(N) - L(N) \quad (5)$$

$$252 \quad B(BC) = AD(BC) + W(BC) - U(BC) - L(BC) \quad (6)$$

253 where B(x) is the nutrient budget of x, AD(x) is total annual atmospheric deposition of x, F(N) is the  
254 fixation of N in the soil, assumed to be  $0.12 \text{ kmolc ha}^{-1} \text{ y}^{-1}$  (DeLuca et al. 2002), and W(BC) is the  
255 weathering of BC. U(x) is the net accumulation of x in wood biomass, and L(x) is the leaching flux of x at  
256 50 cm depth. All fluxes and budgets are expressed in  $\text{kmolc ha}^{-1} \text{ y}^{-1}$ . The plant uptake of N and BC,  
257 leaching of N and BC and weathering of BC were computed with ForSAFE.

258

259 Table 2  
 260 ForSAFE inputs of soil properties for the plot at Klintaskogen

Layer	z <sup>a,b</sup> (m)	Density (kg m <sup>-3</sup> )	Area (10 <sup>6</sup> m <sup>-2</sup> )	pCO <sub>2</sub>	K <sub>gibb</sub>	FC (m <sup>3</sup> water m <sup>-3</sup> soil)	WP (m <sup>3</sup> water m <sup>-3</sup> soil)	LP (m <sup>3</sup> water m <sup>-3</sup> soil)	FS (m <sup>3</sup> water m <sup>-3</sup> soil)	Roots fraction
1	0.035	195	0	10	6.5	0.410	0.150	0.258	0.687	0.2
2	0.051	709	754052	10	6.5	0.299	0.099	0.212	0.565	0.2
3	0.068	957	1017944	20	7.6	0.338	0.093	0.203	0.542	0.28
4	0.136	1393	1805718	20	8.6	0.332	0.089	0.180	0.481	0.22
5	0.17	1258	1389149	20	9.2	0.306	0.076	0.160	0.428	0.1

261 <sup>a</sup> Data are based on the results of lab analysis of the soil samples listed in Table 1 (see text for detailed explanation).

262 <sup>b</sup> z is the corrected layer depth; Density is bulk density; Area is exposed surface area of minerals; The other abbreviations have been  
 263 defined in the text.

264 Table 3  
 265 ForSAFE inputs of soil minerals for the Klintaskogen plot

Layer	Plagioclase														
	Quartz <sup>a</sup>	Albite	Anorthite	Hornblende	Illite	Epidote	Vermiculite1	Vermiculite2	Chlorite1	Chlorite2	Muscovite	Apatite	K-Feldspar	Hematite	Rutile
1	16.59 <sup>b</sup>	5.15	1.02	1.01	0.63	0.86	1.17	0.31	0.53	0.34	2.45	0.49	3.44	1.03	0.32
2	55.07	9.34	1.84	1.04	1.72	1.63	1.52	0.59	0.9	0.61	3.3	0.26	6.87	2.34	0.72
3	55.95	10.07	2.12	1.93	1.29	1.79	2.32	0.64	1.09	0.69	4.7	0.24	6.37	2.05	0.73
4	58.11	11.77	3.09	2.67	2.64	1.32	0.5	0.39	2.02	1.7	2.86	0.26	9.42	2.05	0.65
5	50.19	14.25	1.92	1.35	2.09	2.35	4.48	0.74	1.27	0.79	4.11	0.28	10.81	1.03	0.5

266 <sup>a</sup> Quartz: SiO<sub>2</sub>, Albite: NaAlSi<sub>3</sub>O<sub>8</sub>, Anorthite: CaAl<sub>2</sub>Si<sub>2</sub>O<sub>8</sub>, Hornblende: K<sub>18</sub>Na<sub>54</sub>Ca<sub>166</sub>Mg<sub>210</sub>Fe<sub>180</sub>Ti<sub>11</sub>Al<sub>216</sub>Si<sub>606</sub>O<sub>2146</sub>(OH)<sub>188</sub>,  
 267 Illite: K<sub>0.6</sub>Al<sub>2</sub>(Al<sub>0.6</sub>Si<sub>3.4</sub>O<sub>10</sub>)(OH)<sub>2</sub>, Epidote: Ca<sub>80</sub>Fe<sub>30</sub>Al<sub>96</sub>Si<sub>124</sub>O<sub>495</sub>(OH)<sub>44</sub>, Vermiculite1: Ca<sub>20</sub>Mg<sub>103</sub>Fe<sub>182</sub>Al<sub>162</sub>Si<sub>293</sub>O<sub>832</sub>(OH)<sub>804</sub>,  
 268 Vermiculite2: Ca<sub>10</sub>Mg<sub>103</sub>Fe<sub>22</sub>Al<sub>68</sub>Si<sub>123</sub>O<sub>249</sub>(OH)<sub>490</sub>, Chlorite1: Na<sub>2</sub>Ca<sub>3</sub>Mg<sub>107</sub>Fe<sub>124</sub>TiAl<sub>124</sub>Si<sub>138</sub>O<sub>540</sub>(OH)<sub>442</sub>,  
 269 Chlorite2: Mg<sub>103</sub>Fe<sub>58</sub>TiAl<sub>100</sub>Si<sub>87</sub>O<sub>365</sub>(OH)<sub>302</sub>, Muscovite: K<sub>44</sub>Na<sub>2</sub>Mg<sub>8</sub>Fe<sub>12</sub>Ti<sub>2</sub>Al<sub>96</sub>Si<sub>120</sub>O<sub>390</sub>(OH)<sub>94</sub>, Apatite: Ca<sub>10</sub>(PO<sub>4</sub>)<sub>6</sub>(OH)<sub>2</sub>, K-Feldspar: KAlSi<sub>3</sub>O<sub>8</sub>,  
 270 Hematite: Fe<sub>2</sub>O<sub>3</sub>, Rutile: TiO<sub>2</sub>. (Warfvinge & Sverdrup 1995)

271 <sup>b</sup> the values given are the percent of the total weight in the soil layer. Water content and organic material content are not presented in  
 272 the table.

273

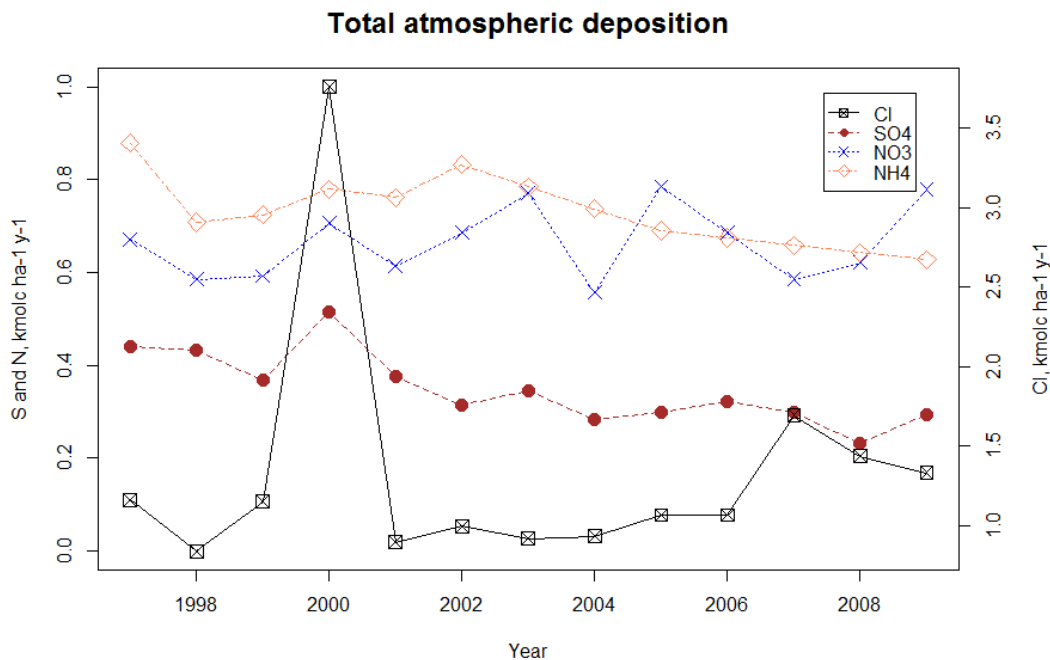
274

275 **3. Results**

276 **3.1. Monitoring data from the study plot**

277 *3.1.1. Atmospheric deposition 1997–2009*

278 The total deposition of  $\text{Cl}^-$ , sulphate, nitrate and ammonium is shown in Figure 2. Chloride was the most  
279 abundant anion in atmospheric deposition, with an average deposition rate of  $1.33 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ . Sulphate  
280 was the second most abundant anion with an average deposition rate of  $0.35 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ . The total N  
281 deposition was  $1.39 \text{ kmolc ha}^{-1} \text{ y}^{-1}$  (nitrate:  $0.66 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ ; ammonium:  $0.73 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ ).  $\text{Cl}^-$   
282 deposition showed a peak in 2000 ( $3.76 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ ), and a small increase after 2006. Sulphate  
283 deposition decreased significantly ( $\tau=-0.684$ ,  $p=0.001$ ) from about  $0.5 \text{ kmolc ha}^{-1} \text{ y}^{-1}$  in 1997 to about  
284  $0.25 \text{ kmolc ha}^{-1} \text{ y}^{-1}$  in 2009. Ammonium deposition also decreased significantly, from  $0.88 \text{ kmolc ha}^{-1} \text{ y}^{-1}$   
285 to  $0.63 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ , over the same period ( $\tau=-0.615$ ,  $p=0.004$ ). However, nitrate deposition and total N  
286 deposition were relatively stable over the period.



287

**Figure 2.** Total atmospheric deposition of  $\text{Cl}^-$ , sulphate, nitrate and ammonium over the period 1997–2009 at the study plot in Klintaskogen.

288

289 Total atmospheric deposition of  $\text{Na}^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$  and  $\text{K}^+$  is given in Table 4. The average  $\text{Na}^+$  deposition  
290 for the years 1998, 1999 and 2001 was  $0.84 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ , while the average deposition of  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$   
291 and  $\text{K}^+$  over the same period were 0.095, 0.10 and  $0.057 \text{ kmolc ha}^{-1} \text{ y}^{-1}$ , respectively. The monthly  
292 deposition (data not shown) of  $\text{K}^+$  showed a decreasing tendency over the period 1998–2001 ( $\tau=-0.306$ ,

293  $p=0.007$ ), and the monthly deposition of  $\text{Ca}^{2+}$  also seemed to decrease however this visual trend was not  
 294 statistically significant ( $\tau=-0.215$ ,  $p=0.06$ ). No significant trends were seen in the monthly  $\text{Na}^+$  and  $\text{Mg}^{2+}$   
 295 deposition over the same period.

296 Table 4

297 Total atmospheric deposition of Na, Ca, Mg and K ( $\text{kmolc ha}^{-1} \text{y}^{-1}$ )

Year	Na	Ca	Mg	K
1998	0.72	0.092	0.092	0.065
1999	1.07	0.10	0.14	0.067
2001	0.73	0.094	0.080	0.039
<b>Average</b>	0.84	0.095	0.10	0.057

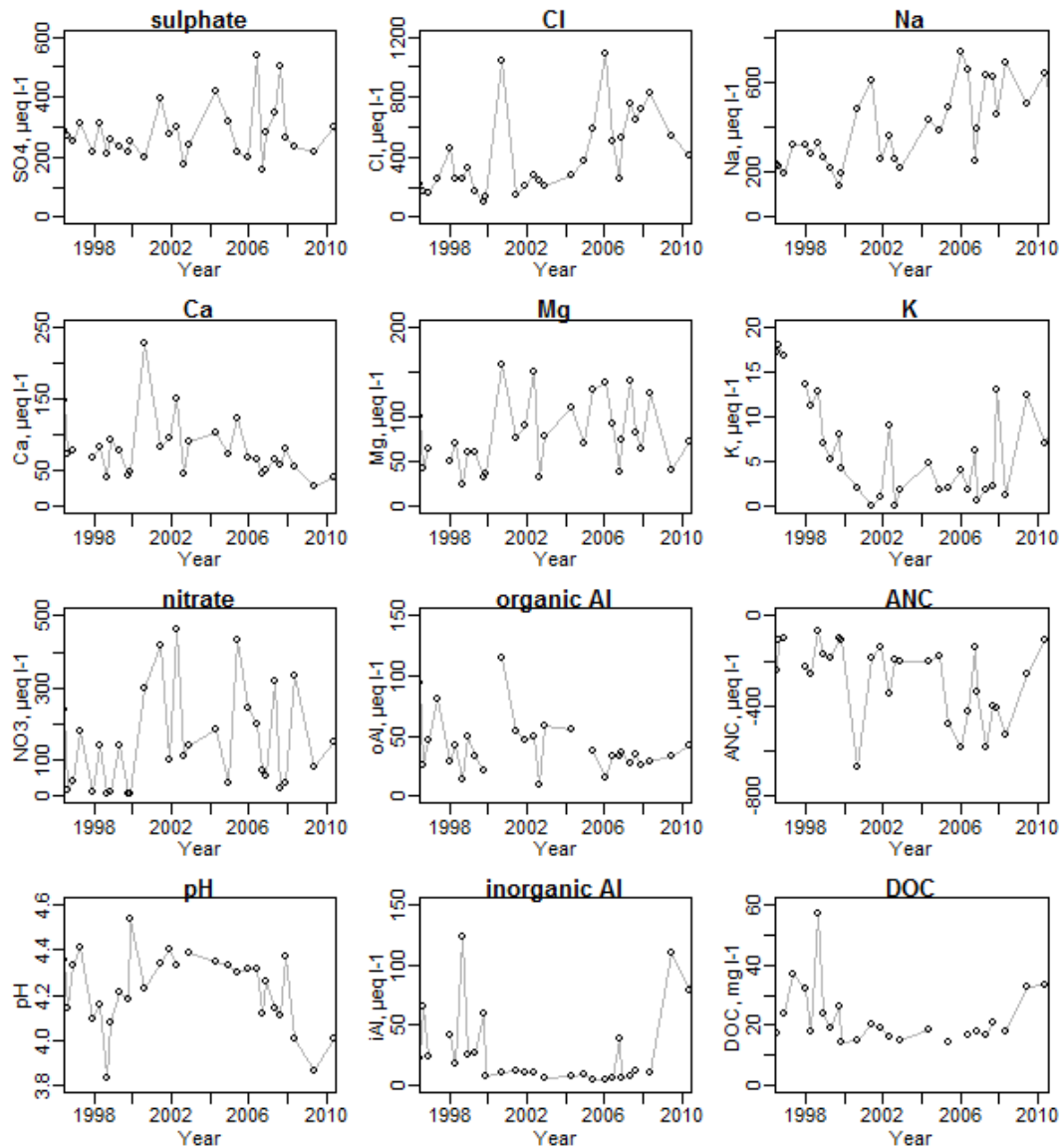
298

299 *3.1.2. Soil solution chemistry 1997–2009*

300 The soil water at the study site over the period 1997–2009 was acidic. The pH ranged from 3.86 to 4.54,  
 301 and the ANC was negative, ranging from about  $-600$  to about  $-100 \mu\text{eq l}^{-1}$ , as can be seen in Figure 3. The  
 302 dominant anion was  $\text{Cl}^-$ , and the dominant cation was  $\text{Na}^+$ . The concentrations of  $\text{Na}^+$  and  $\text{Cl}^-$  increased  
 303 significantly over the whole period monitored ( $\text{Cl}$ ,  $\tau=0.448$ ,  $p<0.001$ ;  $\text{Na}$ ,  $\tau=0.458$ ,  $p<0.001$ ), while  $\text{Ca}^{2+}$   
 304 and ANC decreased ( $\text{Ca}$ ,  $\tau=-0.292$ ,  $p=0.02$ ;  $\text{ANC}$ ,  $\tau=-0.3$ ,  $p=0.02$ ). However, the trend analysis before  
 305 *Lothar* and after *Lothar* gave different results. Before *Lothar*, the nitrate concentration decreased ( $\tau=-$   
 306  $0.491$ ,  $p=0.04$ ). After *Lothar*, the pH and  $\text{Ca}^{2+}$  concentration decreased significantly (pH,  $\tau=-0.602$ ,  
 307  $p<0.001$ ;  $\text{Ca}$ ,  $\tau=-0.568$ ,  $p<0.001$ ), while  $\text{Cl}^-$  and DOC increased significantly ( $\text{Cl}$ ,  $\tau=0.368$ ,  $p=0.03$ ;  $\text{DOC}$ ,  
 308  $\tau=0.459$ ,  $p=0.02$ ). High concentrations of  $\text{Cl}^-$ , nitrate, sulphate,  $\text{Na}^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$  and oAl were noted in  
 309 2000.

310 Over the period monitored, the  $\text{Cl}^-$  concentration was significantly positively correlated with the  
 311 concentrations of  $\text{Na}^+$  and  $\text{Mg}^{2+}$  ( $\text{Na}$ :  $R^2=0.746$ ,  $p<0.001$ ;  $\text{Mg}$ :  $R^2=0.606$ ,  $p<0.001$ ), but not the  $\text{Ca}^{2+}$  and  
 312  $\text{K}^+$  concentrations. Interestingly,  $\text{Cl}^-$  showed the same positive correlations with  $\text{Na}^+$  and  $\text{Mg}^{2+}$  after  
 313 *Lothar* ( $\text{Na}$ :  $R^2=0.615$ ,  $p=0.004$ ;  $\text{Mg}$ :  $R^2=0.522$ ,  $p=0.02$ ), but it was not correlated with any cations before  
 314 *Lothar*. The nitrate concentration was positively correlated with  $\text{Na}^+$ ,  $\text{Mg}^{2+}$  and  $\text{Ca}^{2+}$  ( $\text{Na}$ :  $R^2=0.491$ ,  
 315  $p=0.006$ ;  $\text{Mg}$ :  $R^2=0.800$ ,  $p<0.001$ ;  $\text{Ca}$ :  $R^2=0.526$ ,  $p=0.003$ ) over the period 1997–2009. Both  $\text{Mg}^{2+}$  and  
 316  $\text{Ca}^{2+}$  showed positive correlations with nitrate before and after *Lothar*, but no other correlations with  
 317 nitrate were found before or after *Lothar* for the other elements. The pH was significantly negatively  
 318 correlated with DOC concentration ( $\text{DOC}$ :  $R^2=0.651$ ,  $p<0.001$ ) during the period 1997–2009. The pH was  
 319 also negatively correlated with DOC both before and after *Lothar*.

320



321

**Figure 3.** Soil solution concentrations of major chemical species monitored (sulphate, Cl<sup>-</sup>, nitrate, DOC, Na<sup>+</sup>, oAl, iAl, Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>), ANC and pH at a depth of 50 cm in the Klintaskogen site plot, from 1997–2009.

322

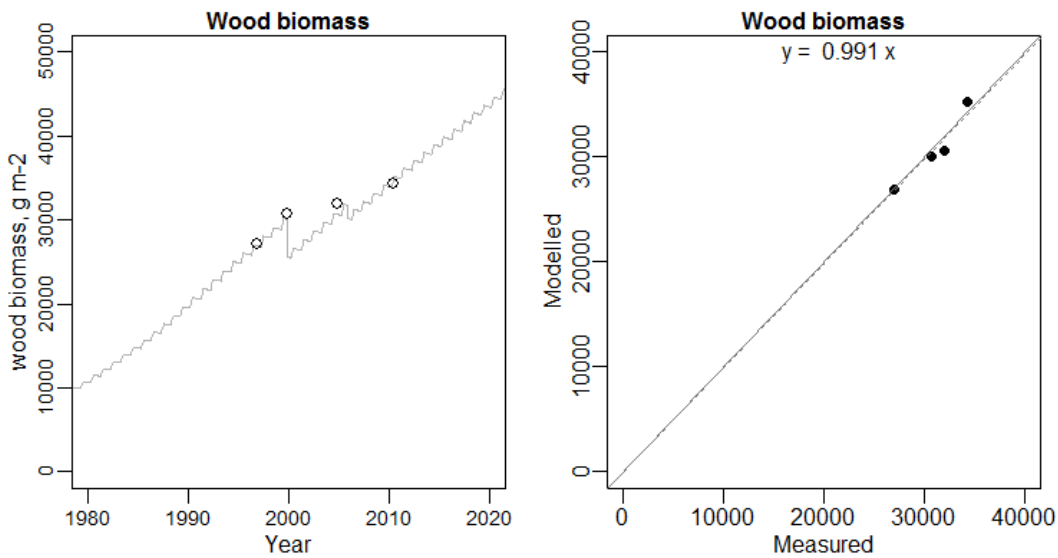
### 323 3.2. Model validation and scenarios

#### 324 3.2.1. Model validation

325 Model validation showed good agreement with the monitored data. The model wood biomass before  
 326 *Lothar*, and the trend of wood biomass change after *Lothar* agreed well with inventory data (Figure 4). For  
 327 the simulation of soil water chemistry over the period 1997–2009, the measured concentrations of all  
 328 chemicals were in the range of modeled concentrations, and ForSAFE also captured the temporal  
 329 dynamics of concentrations for most of chemicals (Figure 5). The model captured two high concentration  
 330 episodes of N after *Lothar* and after *Gudrun* respectively, and the overall modeled N concentration over

331 the period 1997–2009 almost equaled the measured value. The model predicted a decrease in sulphate  
332 concentration, but a few measurements outranged the modeled concentration after *Gudrun*. The model  
333 simulated two high concentration episodes of BC, one after *Lothar* and one after *Gudrun*, which agreed  
334 well with monitored data. But the overall modeled BC concentration was about one-third higher than the  
335 measured value. The model reproduced the temporal dynamics of  $\text{Cl}^-$  concentration, and the overall  
336 modeled  $\text{Cl}^-$  was only 5% more than that of monitored. The modeled  $\text{Na}^+$  concentration was generally  
337 higher than the monitored over the period 1997–2009.

338



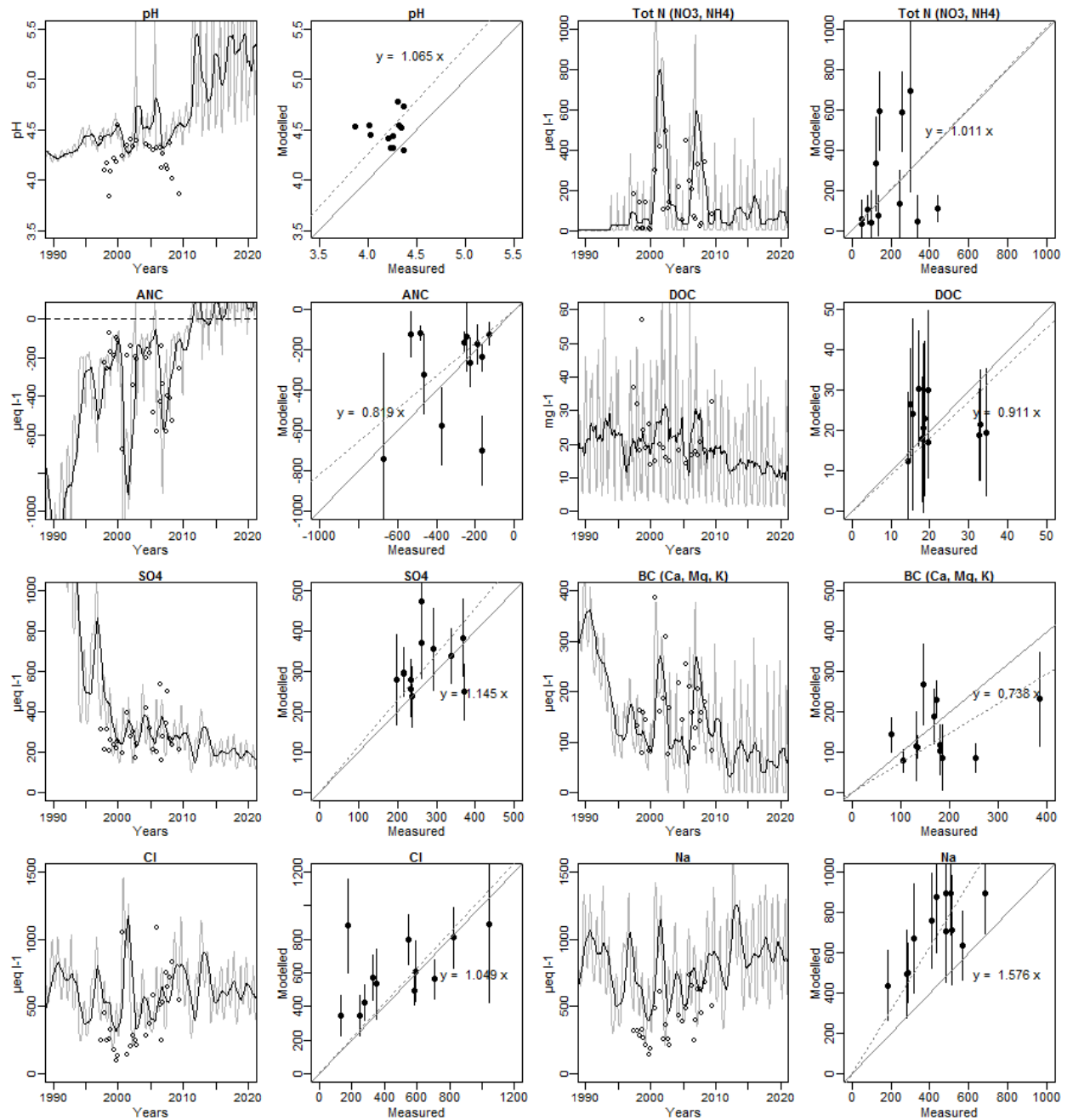
339

**Figure 4.** Measured vs. modelled wood biomass. In the left sub-plot, monthly modelled results (grey line) are compared to measured data (points) over the years; in the right sub-plot, the modelled results and measured data are plotted against each other. The dotted line and its slope reported in the equation ( $y = ax$ ) are an indication of the discrepancy between modelled and measured data.

340

341 However, some discrepancies were observed in the modeling of acidity in soil water. The model  
342 overestimated the pH over the period 1997–2009, and the discrepancies were greater before 2000 and after  
343 2008 (Figure 5). The prediction of ANC, an important indicator of acidity, well presented the temporal  
344 dynamics of measured ANC, but the overall modeled ANC was about 20% higher than the measured value.  
345 The DOC concentration, which has a significant effect on pH, was lower in the model than in the  
346 measured data. Moreover, the shape of the measured DOC concentration curve was significantly different  
347 from the modeled curve.





348

**Figure 5.** Comparison of modeled and measured soil water chemistry data. The grey curves show the modeled monthly values, and the black lines the moving averages (12-month periods) of these values. ○ – measurements, and ● – yearly mean values. The dotted line and its slope reported in the equation ( $y = ax$ ) are an indication of the discrepancy between modeled and measured data. The soil water was collected at a depth of 50 cm.

349

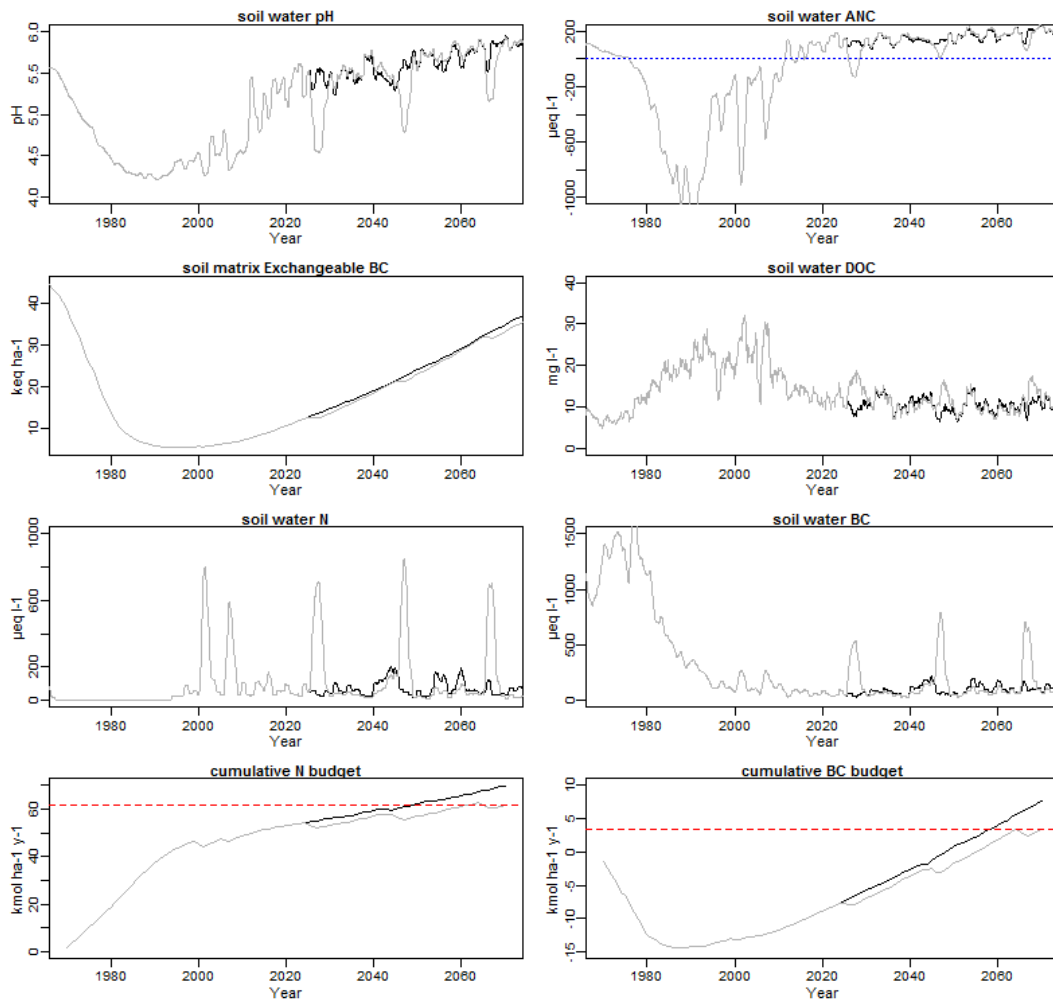
### 350 3.2.2. Model scenarios and soil nutrient budget

351 Clear recovery from the acidification of the 1990s was seen in both the scenarios modelled (Figure 6).

352 This was indicated by a substantial increase in soil solution ANC from  $-1000 \mu\text{eq l}^{-1}$  to  $+100 \mu\text{eq l}^{-1}$ , and

353 by an substantial increase in soil matrix exchangeable base cations from 5 keq ha<sup>-1</sup> to 35 keq ha<sup>-1</sup>. A long-  
 354 term decreasing trend was observed in the modeled BC concentration. The decrease in BC started in the  
 355 1980s, the concentration decreased from about 1500 µeq l<sup>-1</sup> to about 100 µeq l<sup>-1</sup> in the 2000s, by which  
 356 the modeled BC concentration had reached baseline level.

357 Significant differences were predicted in pH, soil solution N concentration and BC concentration between  
 358 the BLS scenario and the FSS scenario. Although a long-term increase in pH was predicted in the FSS  
 359 scenario, strong acidic episodes were seen as the result of storms. Episodes of high concentrations of soil  
 360 solution N and BC were predicted simultaneously in the FSS scenario in response to storms. Higher soil  
 361 solution DOC and lower exchangeable BC in soil matrix were also identified in the FSS scenario due to  
 362 storms. The storm-induced changes in ANC were less profound when comparing the FSS scenario with  
 363 the BLS scenario.



364

**Figure 6.** Effects of future storm disturbances on soil (BLS: black line; FSS: gray line). Soil water pH, ANC, BC and N concentrations are at 50 cm depth. Exchangeable BC of soil matrix is at layer 5 (Table 2). Cumulative N and BC budget is based on section 2.5, the red dotted lines go through the end of grey lines horizontally.

365

366 N accumulated in the soil in both scenarios over the simulation period 1970–2070 (Figure 6). The rates of  
367 accumulation (the slopes of the curve) in most years were positive, but after each storm disturbance, there  
368 were a few years with negative values. The rates started to decrease from 1990s and reached a relative  
369 stable value after 2010s. Base cations were lost from the soil from 1970, and they started to accumulate  
370 from the 1990s, i.e. the rates of accumulation became positive. The rates stabilized after 2010s, and they  
371 also shortly became negative after each storm as N did. Differences of cumulative N and BC input-output  
372 budget between FSS and BLS could be seen after the first future storm (2025), and they increased after  
373 each storm. At 2070, cumulative N and BC budgets were 69.82 and 7.54 kmolc ha<sup>-1</sup> for BLS, and for FSS  
374 they were 61.64 and 3.41 kmolc ha<sup>-1</sup>. The BLS reached FSS's 2070 level in 2050 and 2058 separately, in  
375 other words, the accumulation of N and BC in soil were delayed by storms for about 20 years and 12 years.

## 376 **4. Discussion**

### 377 **4.1. Recovery from acidification at the Klintaskogen site before Lothar?**

378 Klintaskogen is one of the few sites in Sweden which is saturated with both S (Eriksson et al. 1992) and N  
379 (Akselsson et al. 2010; Pihl Karlsson et al. 2011) due to high historical acid deposition. We observed a 50%  
380 decrease of S deposition from 1997 to 2009, but there was no significant change in N deposition over the  
381 period (Figure 2). The change of acid deposition at Klintaskogen over the period 1997–2009 is in line with  
382 the findings in many other sites in Sweden (Pihl Karlsson et al. 2011; Akselsson et al. 2013). One  
383 consequence of the decreasing sulphate deposition is a reduction in the sulphate concentration in soil water.  
384 However, it doesn't necessarily lead to an immediate recovery from the acidification in forest soils (Pihl  
385 Karlsson et al. 2011; Akselsson et al. 2013), probably due to the problem of sulphate adsorption  
386 reversibility (Reuss & Johnson 1986) and the complexity of acidification and alkalization in soils (Van  
387 Breemen et al. 1983). We did not observe significant trends of acidification recovery in soil water before  
388 *Lothar* in monitored pH, ANC or iAl (Figure 3).

389 However, the model simulation clearly showed an increase of pH and a decrease of ANC from 1990s  
390 (Figure 5 and Figure 6). The main driver for the recovery was the decreased sulphate concentration in the  
391 soil water, but another important factor was that no disturbances were simulated between 1990 and 2000.  
392 Therefore we can argue that Klintaskogen should recover from acidification from 1990s if no external  
393 disturbances happen.

### 394 **4.2. The influence of Lothar and Gudrun storms on recovery from acidification during 1997–2009**

#### 395 *4.2.1. Influence of storms on soil chemistry*

396 We observed frequent high-concentration episodes of nitrates, Cl<sup>-</sup>, Na<sup>+</sup> and Mg<sup>2+</sup> after *Lothar* and *Gudrun*,  
397 which could be due to a combination of the sea-salt effect and stimulated mineralization. The temporary

398 changes in pH, Al<sup>3+</sup>, Na<sup>+</sup>, Cl<sup>-</sup> and Mg<sup>2+</sup> measured at the study site after *Lothar* agree well with the  
399 characteristics of the sea-salt effect, which is also supported by the high peak of Cl<sup>-</sup> deposition in 1999.  
400 The continuous high nitrate concentration could probably be explained by an increase in the  
401 mineralization rate due to canopy opening, radiation and temperature change (Vodde et al. 2011).

402 However, soil water chemistry changed slightly after *Gudrun*. This was a storm with high winds but  
403 without heavy precipitation, as could be observed from the monitored throughfall precipitation (data not  
404 shown) and total deposition. Therefore, no evidence of sea-salt effect was found in 2005, whereas  
405 episodes of remarkably high Cl<sup>-</sup>, Na<sup>+</sup> and Mg<sup>2+</sup> concentrations in 2007 and 2008 indicated two new sea-  
406 salt episodes. However, the Ca<sup>2+</sup> concentration did not increase in the two new sea-salt episodes, and we  
407 speculate that this may be due to the greater difficulty for Na<sup>+</sup> to replace Ca<sup>2+</sup> in more acidified soil  
408 (Hindar et al. 1995; Hindar et al. 1994). But the high-nitrate concentration episode had already started in  
409 2005, showing that the increase in mineralization started directly after *Gudrun*.

410 The sulphate seems to decrease before *Lothar* ( $\tau=-0.424$ ,  $p=0.06$ ). Visual inspection of the data after  
411 *Lothar* gives the impression that the amount of sulphate has decreased, but high-concentration episodes  
412 occurred more frequently after *Lothar*, disturbing this trend. The reason for the high-concentration  
413 episodes could be increased sulphur mineralization from organic matter after storms. This suggests that the  
414 long-term decrease in sulphate concentration will be disturbed by future storms.

#### 415 4.2.2. *The influence of storms on soil and water acidity*

416 Higher nitrate leaching was observed after both storms, which are probably due to an increase in organic  
417 matter mineralization and an increase in nitrification. It is known that nitrification, in particular when  
418 excess production of nitrate (not consumed by plants) occurs, has a very strong acidifying effect (Reuss &  
419 Johnson 1986). Meanwhile, nitrate leaching causes base cations leaching.

420 Although high-concentration episodes of sulphate occurred after *Lothar*, sulphate was not correlated with  
421 any cation or ANC, indicating that sulphate is not the main anion regulating ANC or leaching base cations  
422 at Klintaskogen.

423 Sea-salt episodes were observed after *Lothar* and in 2007 and 2008 at Klintaskogen. They are believed to  
424 be of little long-term significance since there is often a recovery in soil water after the sea-salt episode  
425 temporarily acidifies it (Hindar et al. 1995). It is because that the increased H<sup>+</sup> and Al<sup>3+</sup> in soil water are  
426 actually exchanged from soil matrix, i.e. the total acidity of the whole soil does not increase in a sea-salt  
427 episode. Moreover, the recovery from soil acidification may be accelerated due to the additional base  
428 cations from the sea-salt episodes (Beier et al. 2003). We observed high correlation between Cl<sup>-</sup> and Mg<sup>2+</sup>,

429 indicating that sea-salt episodes also brought a lot of  $Mg^{2+}$ . Thus sea-salt episode will probably decrease  
430 the soil acidity of Klintaskogen on long term since it replenishes the base saturation.

431 The changes in soil water acidity between 1997 and 2009 reflect the multiple effects of storms, as well as  
432 other factors, such as DOC. Before *Lothar*, the variation in pH was driven by the DOC concentration, and  
433 indicated by the iAl concentration (Figure 3). The sudden decrease of pH directly after *Lothar* and the  
434 subsequent recovery from acidity up till 2002 was explained by the sea-salt effect (Hindar et al. 1995;  
435 Evans et al. 2001). Similar temporary acidification and recovery were also identified in 2007 and 2008,  
436 when smaller sea-salt episodes were reported also verified by the monitored deposition. The changes in  
437 acidity between 2002 and 2007 were generally related to the changes in ANC, with the exception of 2006,  
438 when an increment in ANC caused a decrease in pH. This was also characterized by very high  $Cl^-$ ,  $Na^+$   
439 and  $Mg^{2+}$  concentrations, indicating a sea-salt episode. However, there was no evidence of a storm in  
440 terms of high precipitation or reports by meteorological institutions, and the reason for this exception is  
441 still unclear. After 2009, the concentration of DOC increased again, causing another acidification episode.

442 Overall, there was no significant acidification recovery over the period 1997–2009. In contrary, the  
443 measured pH decreased significantly due to storms, from 4.54 in Nov. 1999 to 3.86 in May 2009. Given  
444 the fact that sea-salt episode has little long-term influence on acidification, the decreased pH is probably  
445 driven by nitrification and leaching of nitrate. Similar decreasing trends of pH were also observed in two  
446 other southern Swedish Norway spruce forest sites after 2000, which were both very close to the west  
447 coast of Sweden (Akselsson et al. 2013). It indicates that storm disturbances, at sites in close proximity to  
448 sea, would cause delay in the acidification recovery. Meanwhile, the considerable effects of DOC  
449 concentrations on pH suggest that DOC is also a very important factor and should be included in long-  
450 term acidification studies. Visual analysis of the data in this study and the data in Akselsson's study  
451 suggests that soil water that contains  $DOC > 20 \text{ mg l}^{-1}$  would have much higher pH sensitivity to DOC.

#### 452 **4.3. Long-term effects of frequent storms on future acidification recovery and fertility loss**

##### 453 *4.3.1. Reliability of ForSAFE prediction*

454 **Acidity:** The pH curve obtained with the ForSAFE model exhibited a different shape from the  
455 measurements over the monitored period. The model captured the relationship between pH and ANC, but  
456 did not predict the strong impact of DOC on pH very well. Neither was the concentration of DOC well  
457 predicted. In addition, the pH change caused by sea-salt effect is not readily included in the model yet.  
458 Therefore, the difference between the modelled and the measured pH in some cases was larger than half a  
459 unit over the period 1997–2009. However, the modeled ANC was well validated by the measured data  
460 over the period.

461 In the long term, the effects of sea-salt episodes on acidity is little, and the fact that storms will decrease  
462 the ANC in the soil water, regardless the presence of sea-salt episodes, is confirmed by both monitored  
463 data and modeled results. Thus the only uncertainty in the long-term modeled pH is DOC. By assuming  
464 that no significant change would happen in the DOC concentration on long term in southern Sweden, it is  
465 suggested that the modeled long-term trend of acidity change at Klintaskogen is reliable.

466 **Nutrient concentrations:** Both *Lothar* and *Gudrun* caused increases in concentrations of N and BC  
467 (Figure 2 and Figure 3), thus increases in leaching and loss of N and BC from the soil (Figure 6). The  
468 increased leaching of N after both storms was most probably due to the stimulated mineralization (Legout  
469 et al. 2009). The increased leaching of  $Mg^{2+}$  after both storms was most probably due to the sea-salt effect,  
470 but after *Gudrun*, it was caused by two sea-salt episodes in 2007 and 2008, rather than the storm *Gudrun*.  
471 The increased leaching of  $Ca^{2+}$  after *Lothar* was due to the sea-salt effect, but after *Gudrun* the leaching of  
472  $Ca^{2+}$  decreased to the level before *Lothar*, and we speculate that this may be due to the greater difficulty  
473 for  $Na^+$  to replace  $Ca^{2+}$  in more acidified soil (Hindar et al. 1995; Hindar et al. 1994). The leaching of  $K^+$   
474 was very small, compared with those of  $Mg^{2+}$  and  $Ca^{2+}$ .

475 ForSAFE well simulated the concentration and temporal dynamics of soil solution N, but some  
476 discrepancies were seen in soil solution BC. One important reason is that sea-salt effect was not simulated  
477 by the ForSAFE model: Firstly, the base cations are lumped as an equivalent base and are assumed to  
478 react as a divalent component  $BC^{2+}$  in the soil chemistry module of the model. But sea-salt episodes bring  
479 a lot of  $Mg^{2+}$  but not  $Ca^{2+}$  and the vertical reactive transport of these two elements in the soil profile are  
480 very different most especially when the soil contains a high organic matter content (van der Heijden et al.  
481 2014).; And secondly,  $Na^+$  is not currently included in the ion-exchange process in the model. Lumped  
482  $BC^{2+}$  is satisfactory when studying soil acidification. However finer detail is required to better simulate the  
483 nutrient budget, but we believe the model is sufficient for the purposes of this study to obtain an overview  
484 of the long-term trend.

#### 485 4.3.2. Long-term effects of frequent storms on acidification/recovery & fertility

486 ForSAFE predicts that the Klintaskogen site will slowly recover from acidification in the future. Predicted  
487 recovery was slower than prior to 1990s. This recovery is driven by the reduction of S deposition.  
488 However, many studies have reported decreasing base cations deposition like  $Ca^{2+}$  in north America and  
489 Europe over the past decades (Hedin et al. 1994; Watmough et al. 2005; Alewell et al. 2000). This was not  
490 simulated here, because no clear evidence of  $Ca^{2+}$  deposition reductions was available ( $Ca^{2+}$  trend not  
491 significant but  $p=0.06$ ). If  $Ca^{2+}$  deposition also decreases in the future, this may further slowdown or invert  
492 the recovery trend.

493 The simulation of frequent storms has effects on the soil solution chemistry and N and BC leaching fluxes  
494 during the few years after each storm. This caused increased losses of N and BC compared to the BLS  
495 scenario (Figure 6). The leaching of nitrate would have strong acidifying effect on soil water and delayed  
496 N accumulation in soil, but this would not be an issue for the N fertility of the soil because big amounts of  
497 N would accumulated in soils in both scenarios. The leaching of BC would decrease the exchangeable BC  
498 of soil, more problematically, lead to severe loss of BC fertility. The future storms (FSS) lead to loss of 12  
499 years' accumulation of BC fertility comparing to BLS, which is almost 50% of the total accumulation over  
500 the period 1970–2070. However, the recovery indicators (pH, ANC, BC concentration, soil exchangeable  
501 BC) in the FSS scenario are not influenced in the long term compared to the BLS scenario, although a  
502 temporary acidification episode would be seen after each storm. Furthermore, the comparison between the  
503 two scenarios showed that these recovery indicators need about five years or longer to recover from storm  
504 disturbances, which was much longer than the observed time to recover from a sea-salt episode. It  
505 indicates that the influences of mineralization on acidification recovery are stronger than that of sea-salt  
506 episode.

507 Overall, model simulation suggests that storms without sea-salt episodes don't seem to disturb  
508 acidification recovery at Klintaskogen on long term. It is in agreement with the finding of Thiffault et al.  
509 (2007), that storm disturbances appear not to be the long-term driver of soil chemistry. However, storms  
510 raise risks in depleting the BC in the soil. At sites with lower exchangeable pools of  $Mg^{2+}$ ,  $Ca^{2+}$  and  $K^+$ ,  
511 the impact may be more visible.

## 512 **5. Conclusions**

- 513 • The monitoring and the modeled data support the hypothesis that ecosystem disturbances such as  
514 storms may strongly influence acidification recovery processes.
- 515 • There was no monitored evidence proving Klintaskogen was recovering from acidification before  
516 2000, but model simulation implied a trend of recovery. The two storms *Lothar* and *Gudrun* strongly  
517 disturbed the recovery trend, in fact over the period 1997–2009 some indicators (pH, ANC and  $Ca^{2+}$   
518 concentration) suggested that the soil acidification occurred during that period.
- 519 • Storms occurring with both strong winds (windthrow) and high sea-salt inputs (sea-salt episodes) have  
520 an ambiguous effect on acidification recovery. Windthrow and the opening up of the canopy led to  
521 increased mineralization/nitrification, thus nitrate and base cations leaching, causing acidification and  
522 decreased base saturation; the sea-salt episode led to increased soil water acidity and aluminum  
523 concentrations but also possibly replenished the soil cation exchange pool with  $Mg^{2+}$ , thus on long  
524 term increasing the soil pH buffer capacity.

- 525 • No evidence that frequently (20 years) occurring storms with windthrow hamper acidification  
526 recovery but storms may influence soil fertility by losses of base cations. This may be a concern for  
527 the most base cation poor forest soils.

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533 **References**

- 534 Akselsson, C. et al., 2013. Acidification trends in south Swedish forest soils 1986-2008 — slow  
535 recovery and high sensitivity to sea-salt episodes. *The Science of the total environment*, 444,  
536 pp.271–287.
- 537 Akselsson, C. et al., 2010. Assessing the risk of N leaching from forest soils across a steep N  
538 deposition gradient in Sweden. *Environmental pollution*, 158(12), pp.3588–3595.
- 539 Akselsson, C. et al., 2007. Nutrient and carbon budgets in forest soils as decision support in  
540 sustainable forest management. *Forest Ecology and Management*, 238(1-3), pp.167–174.
- 541 Alewell, C. et al., 2000. Effects of reduced atmospheric deposition on soil solution chemistry and  
542 elemental contents of spruce needles in NE—Bavaria, Germany. *Journal of Plant Nutrition  
543 and Soil Science*, 163(5), pp.509–516.
- 544 Attiwill, P.M. & Adams, M.A., 1993. Nutrient cycling in forests. *New Phytologist*, 124(50),  
545 pp.561–582.
- 546 Balland, V., Pollacco, J. a. P. & Arp, P. a., 2008. Modeling soil hydraulic properties for a wide  
547 range of soil conditions. *Ecological Modelling*, 219(3-4), pp.300–316.
- 548 Beier, C., Moldan, F. & Wright, R.F., 2003. Terrestrial ecosystem recovery—modelling the  
549 effects of reduced acidic inputs and increased inputs of sea-salts induced by global change.  
550 *Ambio*, 32(4), pp.275–282.
- 551 Belyazid, S., Westling, O. & Sverdrup, H., 2006. Modelling changes in forest soil chemistry at 16  
552 Swedish coniferous forest sites following deposition reduction. *Environmental pollution*,  
553 144(2), pp.596–609.
- 554 Brandtberg, P.-O. & Olsson, B. a., 2012. Changes in the effects of whole-tree harvesting on soil  
555 chemistry during 10 years of stand development. *Forest Ecology and Management*, 277,  
556 pp.150–162.
- 557 Van Breemen, N., Mulder, J. & Driscoll, C.T., 1983. Acidification and alkalinization of soils.  
558 *Plant and Soil*, 75(3), pp.283–308.
- 559 Dahlgren, R.A. & Driscoll, C.T., 1994. The effects of whole-tree clear-cutting on soil processes  
560 at the Hubbard Brook Experimental Forest, New Hampshire, USA. *Plant and Soil*, 158(2),  
561 pp.239–262.
- 562 DeLuca, T. et al., 2002. Quantifying nitrogen-fixation in feather moss carpets of boreal forests.  
563 *Nature*, 419(October), pp.917–920.
- 564 Eriksson, E., Karlton, E. & Lundmark, J., 1992. Acidification of in Sweden Forest. *Ambio*, 21(2),  
565 pp.150–154.

- 566 Evans, C.D., Monteith, D.T. & Harriman, R., 2001. Long-term variability in the deposition of  
567 marine ions at west coast sites in the UK Acid Waters Monitoring Network: impacts on  
568 surface water chemistry and significance for trend determination. *The Science of the Total*  
569 *Environment*, 265(1-3), pp.115–129.
- 570 Gundersen, P., Schmidt, I.K. & Raulund-Rasmussen, K., 2006. Leaching of nitrate from  
571 temperate forests — effects of air pollution and forest management. *Environmental Reviews*,  
572 14, pp.1–57.
- 573 Hedin, L.O. et al., 1994. Steep declines in atmospheric base cations in regions of Europe and  
574 North America. *Nature*, 367, pp.351–354.
- 575 Van der Heijden, G. et al., 2013. Assessing Mg and Ca depletion from broadleaf forest soils and  
576 potential causes – A case study in the Morvan Mountains. *Forest Ecology and Management*,  
577 293, pp.65–78.
- 578 Van der Heijden, G. et al., 2014. The dynamics of calcium and magnesium inputs by throughfall  
579 in a forest ecosystem on base poor soil are very slow and conservative: evidence from an  
580 isotopic tracing experiment (26Mg and 44Ca). *Biogeochemistry*, 118(1-3), pp.413–442.
- 581 Hindar, A. et al., 1994. Acid water and fish death. *Nature*, 372, pp.327–328.
- 582 Hindar, A. et al., 1995. Extreme acidification in small catchments in southwestern Norway  
583 associated with a sea salt episode. *Water, Air, & Soil Pollution*, 85, pp.547–552.
- 584 Hruška, J. et al., 2012. Long-term forest soil acidification, nutrient leaching and vegetation  
585 development: Linking modelling and surveys of a primeval spruce forest in the Ukrainian  
586 Transcarpathian Mts. *Ecological Modelling*, 244, pp.28–37.
- 587 Iwald, J. et al., 2013. Acidifying effect of removal of tree stumps and logging residues as  
588 compared to atmospheric deposition. *Forest Ecology and Management*, 290, pp.49–58.
- 589 Johns, T.C. et al., 2006. The New Hadley Centre Climate Model ( HadGEM1): Evaluation of  
590 Coupled Simulations. *Journal of Climate*, 19, pp.1327–1353.
- 591 Jonard, M. et al., 2015. Tree mineral nutrition is deteriorating in Europe. *Global Change Biology*,  
592 21, pp.418–430.
- 593 Laudon, H., 2008. Recovery from episodic acidification delayed by drought and high sea salt  
594 deposition. *Hydrology and Earth System Sciences*, 12(2), pp.363–370.
- 595 Legout, A. et al., 2009. Effects of storm Lothar (1999) on the chemical composition of soil  
596 solutions and on herbaceous cover, humus and soils (Fougères, France). *Forest Ecology and*  
597 *Management*, 257(3), pp.800–811.

- 598 Lundell, Y., Johannisson, C. & Ho, P., 2001. Ion leakage after liming or acidifying fertilization of  
599 Swedish forests — a study of lysimeters with and without active tree roots. *Forest Ecology*  
600 *and Management*, 147, pp.151–170.
- 601 Marklund, L.G., 1988. *Biomass functions for pine, spruce and birch in Sweden*.
- 602 McLeod, A.I., 2011. *Kendall: Kendall rank correlation and Mann-Kendall trend test*. R package  
603 version 2.2.
- 604 Milsom, J., 2003. *Field Geophysics* 3rd ed., West Sussex: Wiley.
- 605 Nakicenovic, N. et al., 2000. *IPCC Special Report: Summary for Policymakers Emissions*  
606 *Scenarios*, Geneva.
- 607 Navrátil, T. et al., 2007. Acidification and recovery of soil at a heavily impacted forest catchment  
608 (Lysina, Czech Republic)-SAFE modeling and field results. *Ecological Modelling*, 205(3-4),  
609 pp.464–474.
- 610 Oulehle, F., Hofmeister, J. & Hruška, J., 2007. Modeling of the long-term effect of tree species  
611 (Norway spruce and European beech) on soil acidification in the Ore Mountains. *Ecological*  
612 *Modelling*, 204(3-4), pp.359–371.
- 613 Pedersen, L.B. & Bille-Hansen, J., 1995. Effects of airborne sea salts on soil water acidification  
614 and leaching of aluminium in different forest ecosystems in Denmark. *Plant and Soil*, 168-  
615 169(1), pp.365–372.
- 616 Pihl Karlsson, G. et al., 2011. Reduced European emissions of S and N—effects on air  
617 concentrations, deposition and soil water chemistry in Swedish forests. *Environmental*  
618 *pollution*, 159(12), pp.3571–82.
- 619 Posch, M. & Kurz, D., 2007. A2M—A program to compute all possible mineral modes from  
620 geochemical analyses. *Computers & Geosciences*, 33(4), pp.563–572.
- 621 R Core Team, 2013. *R: A Language and Environment for Statistical Computing*. Vienna, Austria.
- 622 Ranger, J. & Turpault, M.-P., 1999. Input–output nutrient budgets as a diagnostic tool for  
623 sustainable forest management. *Forest Ecology and Management*, 122(1-2), pp.139–154.
- 624 Reuss, J.O. & Johnson, D.W., 1986. *Acid deposition and the Acidification of Soils and Waters*,  
625 New York: Springer-Verlag.
- 626 Schlyter, P. et al., 2006. Assessment of the impacts of climate change and weather extremes on  
627 boreal forests in northern Europe , focusing on Norway spruce. *climate research*, 31, pp.75–  
628 84.

- 629 Simpson, D. et al., 2011. Atmospheric transport and deposition of reactive nitrogen in Europe. In  
630 M. A. Sutton et al., eds. *The European Nitrogen Assessment*. Cambridge: Cambridge  
631 University Press, pp. 298–316.
- 632 Staelens, J. et al., 2008. Calculating Dry Deposition and Canopy Exchange with the Canopy  
633 Budget Model: Review of Assumptions and Application to Two Deciduous Forests. *Water,  
634 Air, and Soil Pollution*, 191(1-4), pp.149–169.
- 635 Sverdrup, H. et al., 2006. Assessing nutrient sustainability of forest production for different tree  
636 species considering Ca, Mg, K, N and P at Björnstorp Estate, Sweden. *Biogeochemistry*,  
637 81(2), pp.219–238.
- 638 Sverdrup, H. et al., 2005. Modeling recovery of Swedish ecosystems from acidification. *Ambio*,  
639 34(1), pp.25–31.
- 640 Sverdrup, H. & Rosen, K., 1998. Long-term base cation mass balances for Swedish forests and  
641 the concept of sustainability. *Forest Ecology and Management*, 110(1-3), pp.221–236.
- 642 Sverdrup, H. & Warfvinge, P., 1993. Calculating field weathering rates using a mechanistic  
643 geochemical model PROFILE. *Applied Geochemistry*, 8(3), pp.273–283.
- 644 Tamm, C.O., 1991. *Nitrogen in Terrestrial Ecosystems* W. D. Billings et al., eds., Berlin:  
645 Springer-Verlag.
- 646 Thiffault, E. et al., 2007. Investigating the soil acid-base status in managed boreal forests using  
647 the SAFE model. *Ecological Modelling*, 206(3-4), pp.301–321.
- 648 Vodde, F. et al., 2011. The influence of storm-induced microsites to tree regeneration patterns in  
649 boreal and hemiboreal forest. *Journal of Forest Research*, 16(3), pp.155–167.
- 650 Vygodskaya, N.N. et al., 2002. Climatic control of stand thinning in unmanaged spruce forests of  
651 the southern taiga in European Russia. *Tellus*, 54B(5), pp.443–461.
- 652 Wallman, P. et al., 2005. ForSAFE—an integrated process-oriented forest model for long-term  
653 sustainability assessments. *Forest Ecology and Management*, 207(1-2), pp.19–36.
- 654 Warfvinge, P. & Sverdrup, H., 1995. Critical loads of acidity to Swedish forest soils. In *Reports  
655 in Environmental Engineering and Ecology*. Lund, Sweden: Lund University.
- 656 Wastenson, L. et al. eds., 1990. *National Atlas of Sweden: The Forest* 1st ed., Enköping: SNA.
- 657 Watmough, S. a et al., 2005. Sulphate, nitrogen and base cation budgets at 21 forested catchments  
658 in Canada, the United States and Europe. *Environmental monitoring and assessment*, 109(1-  
659 3), pp.1–36.

660 Wright, R.F. et al., 1988. Experimental verification of episodic acidification of freshwaters by sea  
661 salts. *Nature*, 334, pp.422–424.

662 Wright, R.F., 2008. The decreasing importance of acidification episodes with recovery from  
663 acidification: an analysis of the 30-year record from Birkenes, Norway. *Hydrology and*  
664 *Earth System Sciences*, 12(2), pp.353–362.

665 Zanchi, G. et al., 2014. Modelling the effects of management intensification on multiple forest  
666 services: a Swedish case study. *Ecological Modelling*, 284, pp.48–59.

667